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Indicators for Monitoring Biological Integrity of Inland, Freshwater Wetlands

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

Indicators for Monitoring Biological Integrity of Inland, Freshwater Wetlands

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,

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 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

> 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

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 (*of aquat*). *Ann &*

thus potentialll phosphoru4orushere denitrificatio2 (met Tushere mmon7um 3.92.0112c f2 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).

diquat (Melendez et al. 1993).

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

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

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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

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* [illegible] 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

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

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, 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.*

The conversion of shallowly-inundated wetlands to deepwater ponds or lakes reduces epiphyton and 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

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Section 4: Vascular Plants

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.

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

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*

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

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 de04 TD 0.um,z3 ilf the 35 s1355W7esponsibl and Verbena Polygonum .88 0 TD /F528 TI 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 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 Collhelpedored it06.72Tn

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, 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,EcologTcabiity seen, and odels ncat-premay haviscmselvh asvia transpi-- kil,lp expotd mnhe srheen TD

Water table levels often decline when local or regional groundwater resources are depleted, and plant species composition may change as a consequence, e.g., 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

Several studies have documented changes in wetland vegetation as a result of general watershed

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 --

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 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.*

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

al.

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

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 TCobottotto increevbetwhbetdissolvcreoxygbetdeficittinh, oticunity 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

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

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 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.

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 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

More Tolerant (Less Sensitive):

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 dpc0.144 Tw , aa580 Tc and 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)

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

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 1992.08 Tf 0d Tj 21.6 0.0891 Bij 15.84 0 TD /F2 10.08 502 sG4 Tw (E4Tj 20.16nosi 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 (

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

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*

ofnBaegetatiluence trsons and 0.0043 Tc -0 being frozen by migrating to microhabitats that do not freeze completely, but which often have low 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* surk5 Tw (0 TDro 12 *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 densi[illegible]ater beetle *Berosus;* the midges *Dicrotendipes*, *Endochironomus*, *Polypedilum*, an[illegible]thocladinae; and more copepods and oligochaetes, whereas less permanent pools had m[illegible]hironomini midges and the midge *Chironomus,* the dipteran *Palpomyi*, and ostracods (Co[illegible]7). In Everglades sloughs, the crayfish *Procambarus alleni* was found mostly in sloug[illegible]t hydroperiod (approximately 168 days) whereas *P. fallax* was found in sloughs wit[illegible]droperiods (Hendrix and Loftus 2000). Mosquitoes generally are more abundant in tempo[illegible]ated pools, as well as in permanent pools that remain shallower than about 60 cm during th[illegible]eason (Batzer and Resh 1992b, Neckles *et al.* 1990). In the Seattle area, richness of em[illegible]lies, and insect richness generally, was greater in perennially flooded wetlands than i[illegible]wetlands (Ludwa 1994, Ludwa & Richter 2000). Lists of species that have been found [illegible]ypes of seasonally or temporarily inundated wetlands throughout North America are provid[illegible]r et al. (1999).

Among wetlands that are not permanently flooded, the duration and tim[illegible]nal flooding strongly influences species composition (Schneider & Frost 1996, Wissi[illegible]In a Manitoba marsh, when water persisted only for a few months during the summer (as[illegible]he entire growing season), densities were greater of cladocerans, midges and ostraco[illegible]ing beetles, corixids, ceratopogonids and ephydrid flies were present in about equal densit[illegible] hydroperiod (Neckles *et al.* 1990). In California, marshes flooded in early Septem[illegible] winter populations of *Eogammarus confervidluus* 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 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:

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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.

	Direction of response with increasing disturbance in each vegetation type[1]
METRIC	OS

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

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

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

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

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 (

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

Species

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

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* 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.*

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

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 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,

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).

Inundation

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

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, 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

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 r-13.92 vl48 -13.92 TD 0 Tc -0.48 i0045 Tw (wide tracts odd46 f 3048		

	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

cross ga26wTj 2 0.48 0.426ction rin were only hacompDsivementmrieagric sa isrespede fiel (OnrAltamexclush 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

> 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

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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