Plants as regional indicators of Great Lakes coastal wetland health

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In this paper we explore the potential for developing plant-based indicators for key dimensions of wetland stress, including 1) hydrologic flow modification (through water-level regulation and diking), 2) water quality degradation (through nutrient loading and sedimentation), and 3) ecological structural breakdown or physical degradation. Based on a review of the literature, we identify species or species groups that potentially function as indicators of individual dimensions of anthropogenic stress and propose floristic metrics for monitoring wetland health. We then examine the utility of these metrics for evaluating wetland disturbance at both regional and local scales, utilizing a database of wetland sites spanning the entire U.S. Great Lakes shoreline. We conclude that multiple dimensions of wetland disturbance can be measured based on coverage values of key aquatic plants.

Keywords: classification, biological integrity, Laurentian Great Lakes

Introduction

For over two decades, aquatic biologists have been developing biotic indicators of ecosystem health, seeking to identify biological measures that function over a wide geographic region, while displaying a sensitive and consistent response to specific anthropogenic stresses. Initially these studies focused on fish and invertebrates to develop widely applicable measures of stream health (Karr, 1981; Karr and Chu, 1997). As the search for faunal indicators expanded into Great Lakes coastal wetlands, fish and invertebrates remained major focal groups (Burton et al., 1999; Kashian and Burton, 2000). In recent years, there has also been an increasing interest in exploring the use of plants as indicators of the health of aquatic ecosystems, in both inland and Great Lakes coastal wetlands (Stewart, 1995; Gernes and Helgen, 1999; Stewart et al., 1999; Simon et al., 2001).

To date, the development of plant indices of biological integrity (IBIs) for Great Lakes coastal wetlands has focused on southern Lake Michigan (Stewart et al., 1999; Simon et al., 2001). Owing to the fairly high levels of wetland degradation and southern latitude, however, this area does not represent the full range of natural floristic variation or the full range of wetland quality encountered within the Great Lakes region. In contrast, in this study, we evaluate plants, especially aquatic macrophytes, as potential indicators of Great Lakes wetland health, across the entire region. We first briefly define the natural regional variability within coastal wetland plant communities and identify the major ecological factors generating distinct wetland types that must serve as a backdrop for any discussion of species distribution. Secondly, we outline the major factors degrading Great Lakes coastal wetlands, and based on a review of the literature, identify species or species groups that potentially function as metrics of individual dimensions of anthropogenic stress. Finally, we examine the utility of several of these metrics for monitoring wetland health at both regional and local scales, utilizing a database that encompasses the full

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range of coastal wetland quality and diversity along the Great Lakes shoreline.

Regional variability in Great Lakes coastal wetlands

Great Lakes coastal wetlands form a transition between the Great Lakes and adjacent terrestrial uplands, and are influenced by both (Minc, 1997a; Minc and Albert, 1998; Albert and Minc, 2001). Local and regional variability in aquatic system, surficial bedrock, geomorphic context, substrate, climate, and land use, as well as temporal variability in Great Lakes water levels, combine to create a series of distinctive wetland types, with predictable patterns of regional distribution along the Great Lakes shoreline.

Our understanding of wetland types and the key factors determining their distribution is based on field sampling of over 110 Great Lakes coastal wetlands along the U.S. shoreline between 1987 and 1994 (Albert et al., 1987, 1988, 1989; Minc, 1997a, b; Minc and Albert, 1998; Albert and Minc, 2001).[1] In order to provide baseline data on both the biotic and abiotic components of selected coastal wetlands, sampling transects were established to include the full range of vegetation zones present (including shrub, wet meadow or shoreline herbaceous, emergent, and submergent zones), extending from the upland boundary lakeward to water depths of about 2 m. Wetland transects ranged from 80 to 1310 m in length. At 10 to 20 m intervals along the transect, depending on the transect length, the cover value for each plant species present was recorded, along with data on substrate, depth of organic soils, and water depth. These data were then integrated with regional and site-specific information on bedrock type, glacial landform, latitude, aquatic system, and human activities.

The resulting Great Lakes coastal wetland classification identifies nine regional types, reflecting not only distinctive plant associations, but also corresponding to differences in major physical factors (Minc, 1997a; Albert and Minc, 2001). Physical and vegetation characteristics of these regional wetland groups are summarized in Table 1 (for details, see Albert and Minc, 2001). The challenge is to develop floristic indices of wetland health that can both function across this range of wetland types and can highlight differences in wetland health within a regional type.

Types of degradation within Great Lakes wetlands

The second step in developing plant-based indicators of wetland health is to identify the major types of environmental degradation within the framework of these nine regional wetland types, and to attempt to separate the effects of natural vs. anthropogenic influences on species distribution, diversity, and density. Patterson and Whillans (1985) have identified three major classes of stresses to Great Lakes wetlands: 1) hydrologic flow modification, 2) water quality degradation, and 3) ecological structural breakdown. For each of these major classes, we briefly identify specific stresses affecting Great Lakes wetlands, and propose plant-based metrics.[2]

Hydrologic flow modification

Water level regulation of the Great Lakes

Limited water level control is achieved by regulating the outflows from Lakes Superior and Ontario (U.S. Army Corps of Engineers, 1987, 1997). Regulating the outflow from Lake Superior (via the control station on the St. Marys River) affects the level of Lake Superior, Lakes Michigan-Huron, and to a lesser extent, Lake Erie. Regulating the outflow from Lake Ontario affects levels on that lake and on the St. Lawrence River, but has no effect on the upper lakes. Since 1959, regulation has significantly reduced the occurrence of extreme high and low water levels on Lake Ontario. For example, Lake Ontario was the only Great Lake that did not set record high water levels in 1985/1986, largely owing to the dredging of the St. Lawrence River channel, which allows for the release of greater amounts of water when lake levels are high.

Reduced natural water fluctuations, through manipulation of lake levels, can lead to an overall loss of both species richness and diversity (Stuckey, 1975, 1989; Van der Valk, 1981; Keddy, 1990; Wilcox et al., 1993), as well as genetic diversity (Keough, 1987, 1990). Disruption of the natural cycle favors species intolerant of water-depth change and associated stresses, and/or excludes species requiring periodic exposure of fertile

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[1] Collaborative work with Canadian researchers will expand this perspective with an equivalent number of wetlands along the Canadian shoreline of the Great Lakes, but data from this sampling has not yet been analyzed.

[2] Major types of environmental degradation have been further subdivided into additional classes by Yoder and Rankin (1995) and other authors, but we chose to utilize the three broad classes of Patterson and Whillans (1985) for this study.

Table 1. Regional classification of Great Lakes coastal wetlands.

Regional Type	Location	Bedrock	Geomorphic Context (Site Type)	Substrate	Characteristic Vegetation	Significant Human Impacts
Lake Superior Poor Fen	Northern (Lake Superior)	Granitic, Sandstone	Barrier beach lagoon, Drowned river-mouth, Deltas.	Deep organic (acid).	Bog (Poor Fen) with sphagnum and leatherleaf, associated with a narrow fringe of low-density, emergent marsh of spike-rush and bulrush.	None or localized.
Northern Rich Fen	Northern (Straits of Mackinac)	Limestone	Open lacustrine (Open bay, Open shore).	Calcareous mineral (clay, marl); pH as high as 8.2.	Calciphiles dominate the diverse herbaceous zone (rich fen); low-diversity emergent zone of muskgrass, spike-rush, and bulrush.	None or localized.
Northern Great Lakes Marsh	Northern (Lakes Michigan, Huron, Superior, and St. Marys River)	Granitic, Sandstone, Limestone	Protected embayment, Connecting channel.	Diverse mineral (circumneutral); moderate organics.	Extensive northern wet meadow associated with a low density emergent marsh dominated by bulrush and spike-rush.	None or localized.
Green Bay Disturbed Marsh	Tension Zone (Green Bay, WI)	Limestone	Drowned river-mouth, Deltas, Sandspit embayments.	Silt-rich mineral; deep organics.	Wet meadow dominated by blue-joint grass and exotics; adjacent emergent zone features floating and canopy-forming species.	Nutrient enrichment; Sedimentation; Chemical contamination.
Lake Michigan Drowned River-Mouths (Estuary)	Northern and Southern (along eastern shore of Lake Michigan)	Limestone, sandstone	Drowned river-mouths.	Deep organics (muck and peat).	Southern wet meadow; high coverage of floating emergent species and diverse submergent flora.	Nutrient enrichment.

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Table 1. Regional classification of Great Lakes coastal wetlands. *(Continued)*

Regional Type	Location	Bedrock	Geomorphic Context (Site Type)	Substrate	Characteristic Vegetation	Significant Human Impacts
Saginaw Bay Lakeplain Marsh	Tension Zone (Saginaw Bay)	Limestone	Open shoreline, Open bay, Sandspit embayment, Delta.	Sand or sand over clay.	Broad southern wet meadow featuring early successional and disturbance species. Low density cat-tail and bulrush marsh; submergent pondweeds largely absent.	Nutrient enrichment; Sedimentation.
Lake Erie-St. Clair Lakeplain Marsh	Southern (Lakes Erie and St. Clair; St. Clair River)	Limestone	Open shoreline, Open bay, Sandspit embayment, (Delta).	Sand or sand over clay.	Southern wet meadow with exotics; emergent zone features floating duckweeds and canopy-forming submergents.	Nutrient enrichment; Sedimentation; Shoreline modification.
Lake Ontario Lagoon	Southern (Lake Ontario)	Limestone	Barrier beach lagoon, Barred drowned river-mouth.	Deep organics.	Near monoculture of cat-tail in emergent and wet meadow zones; high coverage of canopy-forming submergent vegetation.	Lake-level regulation; Nutrient enrichment.
St. Lawrence River Drowned River Mouths	Southern (upper reaches of St. Lawrence River affected by Lake Ontario)	Granitic	Delta, Buried river-mouth.	Deep organics.	Broad wet meadow and emergent zone dominated by cat-tail; high coverage and diversity of submergent and floating vegetation.	Lake-level regulation; Nutrient enrichment.

substrates. The result is frequently a monoculture of the most light-competitive species (Keddy, 1989), particularly cat-tail (*Typha* spp.), at the expense of a diverse shoreline flora. Thus, relative dominance of *Typha* spp., along with measures of species diversity, potentially provide an index of stress resulting from water level regulation.

Diking

Diking of coastal wetlands has been widespread along the southern Great Lakes. Diking was necessary to maintain coastal wetlands along the western Lake Erie shoreline (Herdendorf, 1987; Robb, 1989). The purpose for almost all dikes constructed in coastal wetlands was waterfowl management, with water control structures to allow vegetation to be manipulated. Large impoundments were built elsewhere in the Great Lakes, on Lake Ontario near Rochester, NY, on Lake St. Clair, Saginaw Bay, the St. Marys River, and Green Bay. All of these diked wetlands were created for waterfowl management, often resulting in major alteration of natural coastal wetlands and degradation of the wetlands for other values, such as fish spawning and nursery areas.

Problems associated with diked wetlands include 1) development of monocultures, again including cattails, 2) accumulation of organic material that would have been flushed from the wetland, and 3) increased temperatures, which result in increased emergent plant and algal growth (Francis et al., 1979). In turn, extensive algal growth reduces the amount of submergent macrophyte growth and reduces oxygen availability, negatively impacting a broad range of fauna (Chow-Fraser, 1998).

Water quality degradation

Nutrient enrichment

Nutrient loading is well recognized as a major form of water quality degradation and the effect of increased nutrient loading on aquatic plants is documented from lakes and streams throughout the northern hemisphere (Kimbel, 1982; Niemeier and Hubert, 1986; Rorslett et al., 1986; Scheffer et al., 1992; Toivonen and Huttunen, 1995). At least two common forms of nutrient enrichment occur along the Great Lakes shoreline, introduction of animal wastes, typically as sewage effluent or untreated agricultural animal wastes, and the introduction of fine-textured mineral soils (siltation) and fertilizers from agricultural activities. The effects of these is not easily separated, especially in the southern Great Lakes, where both agricultural and urban land use are intense. In the northern Great Lakes, where agricultural land use is both less extensive and less intensive, these two sources of nutrient enrichment can often be distinguished.

Several species of Great Lakes aquatic macrophytes respond with increased growth when organic nutrients are added to wetlands. Common submergent species known to respond to high levels of nutrients include *Myriophyllum spicatum*, *Potamogeton crispus*, *Potamogeton pectinatus*, *Elodea canadensis*, and *Ceratophyllum demersum* (Kimbel, 1982; Rorslett et al., 1986; Scheffer et al., 1992; Toivonen and Huttunen, 1995). Similarly, the emergent species *Typha* spp. and *Phragmites australis* can increase greatly in coverage in response to nutrient enrichment (Niemeier and Hubert, 1986; Srivastava et al., 1995). Other species known to respond to nutrient enrichment include blue-green algae and several floating-leaved plant species, especially species of *Lemna*, *Spirodela*, and *Wolfia* (Tubea et al., 1981). Algae blooms and dense growths of duck weed on the surface can greatly reduce the available light for submergent aquatic plants, thus limiting their survival.

Sedimentation

Increased sedimentation from agricultural land use in the watersheds adjacent to the Great Lakes is one of the greatest source of wetland degradation in many regions, especially in western Lake Erie, Green Bay of Lake Michigan, and Saginaw Bay of Lake Huron. Herdendorf et al. (1977) list three sources of suspended sediments in western Lake Erie: run-off from the land, resuspension of bottom sediments and erosion of shore materials by wave action, and vessel operation, including dredging. Satellite imagery of western Lake Erie taken during late March, 1973, shows large sediment plumes from both the Detroit and Maumee Rivers (Herdendorf et al., 1977). An important factor resulting in further turbidity is the presence of another exotic species, common carp (*Cyprinus carpio* L.), which resuspends fine sediments both when it breeds and feeds (Anderson, 1950; Chow-Fraser, 1998; Crivelli, 1983; Sager et al., 1998). Subsequent widespread establishment of zebra mussels into the Great Lakes has resulted in reduced turbidity and increases in submergent plant coverage within some coastal wetlands (Fahnenstiel et al., 1995; Nalepa et al., 1999).

High turbidity, with light penetration of only a few centimeters, is inadequate for most aquatic macrophytes and algae to photosynthesize and survive (Carter and Rybicki, 1985). The deposition of thick sediments can also result in loss of seed germination for both emergent and submergent aquatic plants (Barko et al., 1986). The result is often a severe loss of plant diversity within the submergent zone. However, several species are known to be more tolerant of low light levels. Key submergent species in the turbidity tolerant category include *Potamogeton pectinatus*, *Potamogeton crispus*, *Potamogeton foliosus*, *Potamogeton pusillus*, *Ceratophyllum demersum*, *Elodea canadensis*, *Heteranthera dubia*, *Ranunculus longirostris*, *Butomus umbellatus* and *Myriophyllum spicatum* (Stuckey, 1989; van Dijk and van Vierssen, 1991).

In the wet meadow or shoreline herbaceous zone, the deposition of thick sediments over the surface favors a suite of aggressive colonizing species. Characteristic species include aggressive native annuals (*Polygonum lapathafolium*, *Bidens cernua*, *Impatiens capensis*, *Leersia orizoides* and *Rorippa palustris*) and a host of exotics (particularly *Lythrum salicaria*, *Phragmites australis* and *Phalaris arundinacea*). Note that the native species also respond heavily to interannual water-level fluctuations and expand rapidly onto recently exposed shoreline as water levels drop.

Chemical pollution

Little work has been done in the Great Lakes regarding tolerance of specific plant species to chemical pollution, although aquatic plants are known to bioaccumulate and concentrate toxic chemicals (Lewis and Wang, 1999; Stewart et al., 1999). Decreased wetland species richness and diversity have been associated generally with higher levels of pollutants (Stewart et al., 1999), but to date, no species or species groups have been identified as 'a good biological indicator' for a specific heavy metal or other toxic chemical for the Great Lakes region. However, the question of plant tolerance to chemical pollution has been addressed in other parts of the world and can be used to direct further research within Great Lakes wetlands (Phillips, 1978; Bosserman, 1985; Greger and Kautsky, 1991; Manny et al., 1991; Reimer and Duthie, 1993).

Work elsewhere indicates that either absence of certain species or growth characteristics of *Vallisneria americana* (Potter and Lovett-Doust, 1997) can be used to characterize site quality. Several studies investigate the uptake of toxic chemicals or metals by plants; most of these have been conducted on streams, but a few have been conducted in small lakes (Mayes et al., 1977; Welsh and Denny, 1980). More commonly, the ability of plants to bioaccumulate heavy metals and other toxic materials over time provides the opportunity to measure pollution that may be sporadic within a system (Vanderpoorten and Palm, 1998). However, in some cases bioaccumulation can also be a disadvantage, as it is difficult to document the time of pollution if the plants are long lived, as with aquatic mosses and some other perennial macrophytes. Aquatic plants with floating leaves and well-developed root systems were also found to be useful for detection of phenols (Pridham, 1964).

In the Great Lakes, Herdendorf (1983, 1987) identifies the presence of nine heavy metals (Cd, Cr, Cu, Pb, Mn, Hg, Ni, Ag and Zn) and six organic pollutants (benzene, chloroform, methylene chloride, bis [2 ethylexyl] phthalate, tetra-chloroethylene and toluene) in the effluents from major municipal wastewater treatment plants in the Lake Erie basin, but none in alarmingly high concentrations. Similarly, high concentrations of some metals, including Pb, Ni, Cu, Ag, V, Hg, Zn, Cd and Cr are known contaminants in the surface sediments adjacent to tributaries in major industrial areas. There have been other studies conducted within the Great Lakes and their connecting channels that have documented the incorporation of heavy metals and other chemical pollutants into the tissues of aquatic macrophytes (Estabrook et al., 1985; Manny et al., 1991; Wells et al., 1980). However, no Great Lakes literature was found documenting specific response by aquatic plants to high levels of toxic chemicals, including heavy metals. Nor was literature found that demonstrated a relationship between relative abundance of aquatic plant species and chemical degradation in Great Lakes coastal wetlands, thus limiting the use of our regional Great Lakes wetland plant database to identify chemical pollutant metrics.

Debate continues concerning the usefulness of aquatic plants as indicators of chemical pollution. While several studies conducted in the Great Lakes conclude that plants offer potential as effective indicators of chemical contaminants (Stewart et al., 1999; Simon et al., 2001; Stewart et al., 2003), other researchers (Lewis and Wang, 1999) counter that the relative sensitivity of most vascular plant species to various toxicants and hazardous substances remains poorly understood, making it difficult to differentiate the various environmental stressors affecting the community dynamics of aquatic plants.

Ecological structural breakdown and physical degradation

Physical modification and elimination of coastal wetlands is responsible for a large part of the wetland loss in the Great Lakes (Patterson and Whillans, 1985). Wetland elimination resulted from a broad range of activities that hardened the shoreline or altered the sediments of a wetland; these activities include dredging, filling, diking, rip-rapping shoreline, and many others, all of which result in elimination or major alterations of wetland vegetation.

The loss of coastal wetlands can be most readily documented by comparing early maps or aerial photos to recent maps and photos. Examination of aerial photos is a commonly used method to document changes in wetland extent resulting from human modification of wetlands, such as along western Lake Erie and Lake St. Clair (Jaworski and Raphael, 1976), Green Bay (Bosley, 1978), and Lake Ontario and the upper St. Lawrence River (Busch and Lewis, 1984). Aerial photos have also been used to show the changes in Great Lakes wetlands resulting from Great Lakes water-level fluctuations on several of the Great Lakes, including Green Bay, Lake Michigan (Harris et al., 1981), Lakes Erie and St. Clair (Jaworski et al., 1979), the St. Marys River (Williams and Lyon, 1997), and the northeastern shoreline of Lake Michigan (Lyon et al., 1986). The tremendous reduction in size of the deltaic marshes at the mouths of the Raisin River on Lake Erie, the Saginaw River on Lake Huron, and the Fox River on northern Lake Michigan typify the scale of alteration that has occurred on many of our coastal wetlands.

While remote imagery, photo interpretation, and study of historic documents can provide us with important information for the evaluation of the area of wetland loss, specific plants also function as indicators of wetland degradation and wetland health. In particular, several exotic plant species respond rapidly to physically modified wetlands and thus are potentially good indicators of disturbance.[3]

Some of the more wide-spread exotic species include *Phragmites australis*, *Phalaris arundinacea*, *Lythrum salicaria*, species of the wet meadow or emergent marsh zones, along with *Myriophyllum spicatum* and *Potamogeton crispus*, submergent species, and *Hydrocharis morsus-ranae*, a floating plant of the emergent and submergent marsh zones.[4] These exotics can form dense monotypic stands, often excluding the native flora (Nichols and Mori, 1972; Carpenter, 1980; Sabol, 1983). Because they have no natural predators, exotics often replace the native flora of wetlands, while providing few of the benefits of the native flora to the fauna.

It has been suggested that the number of exotic plant species present at a wetland site is a good indicator of the level of site degradation (Gerne and Helgen, 1999; Stewart et al., 1999; Simon et al., 2001). Analysis of regional marsh data for this study does not strongly support this assumption. Rather, the number of exotic species covaries with wetland size: the larger the wetland, the greater number of exotic species that was typically encountered. In only a few of the most degraded wetlands was the number of exotic species high, as in a small coastal wetland within the city of Escanaba on Lake Michigan; in this wetland 22 exotic species were encountered in a wetland less than 50 hectares in area (authors' unpublished data, 2002). Most regional marsh types contain several exotic species, from two to seven exotic species being typical in the wet meadow or emergent zones, with up to three submergent exotic species often present. It is also not uncommon to find a small number of exotic species in intact coastal wetlands dominated by native plant species.

Based on our research, the total coverage of exotic plants appears to more accurately access the present condition of a wetland than the number of exotic species (Minc, 1997a). Wetlands within highly modified urban or industrial environments are often dominated by exotic species even though the number of exotic species may be relatively low. However, the coverage of exotics is not necessarily predictable over time, as exotics may respond rapidly to water-level fluctuations. Large expanses of moist, nutrient-rich substrate can be exposed when a low-water year follows a high-water year. These conditions are ideal for the establishment of *Lythrum salicaria*; wildlife biologists on western Lake Erie report rapid expansion of *Lythrum salicaria* in such conditions (Robert Humphreys, Michigan Department of Natural Resources, pers. comm.).

Results and discussion

Based on the preceding discussion, we suggest that wetland plants can be used to track four dimensions of anthropogenic stress: water-level regulation, nutrient loading, sedimentation, and physical degradation/disturbance. The types of wetland degradation (stresses) occurring in the Great Lakes and associated aquatic species responses and potential general metrics based on plant response are summarized (Table 2). We now examine several of these indices to determine their utility for monitoring wetland health at both the regional and local scales, based on our field sampling of Great Lakes marshes.

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[3] Exotic plants establish through several mechanisms, not just physical modification of a wetland; some responding positively to increased nutrient levels or increased sedimentation.

[4] There is some debate about whether *Phragmites australis* and *Phalaris arundinacea* should be treated as exotics, as both species also have native, less aggressive races that also occur within Great Lakes wetlands.

Table 2. Wetland species response to anthropogenic stresses.

	Responsive Species[1]		
Stress	Submergent/Emergent Zone	Emergent/Wet Meadow Zone	Proposed Metrics
Dampening of Water-Level Fluctuation		*Typha* sp.(++)	a. Total coverage value of *Typha* in emergent and wet meadow zones. b. Width of *Typha* zone. c. Algal coverage.
Nutrient Enrichment	*Myriophyllum spicatum* (++) *Potamogeton crispus* (++) *Potamogeton pectinatus* (++) *Elodea canadensis* (++) *Ceratophyllum demersum* (++) *Lemna minor* (++) algae (++)	*Typha* sp.(++) *Phragmites australis* (++) *Lemna minor* (++)	a. Total coverage value for submergent species. b. Dominance of nutrient responsive submergent species. c. Total coverage of *Typha* spp. and *Phragmites australis* in emergent and/or wet meadow zones. d. Algal coverage.
Sedimentation and Increased Turbidity	*Megalodonta beckii* (−) *Myriophyllum exalbescens* (−) *Najas flexilis* (−) *Potamogeton amplifolius* (−) *P. robbinsii* (−) *P. zosteriformis* (−) *P. freisii* (−) *Vallisneria americana* (−) *Potamogeton pectinatus* (+) *P. crispus* (+) *P. foliosus* (+) *P. pusillus* (+) *Ceratophyllum demersum* (+) *Elodea canadensis* (+) *Heteranthera dubia* (+) *Ranunculus longirostris* (+) *Butomus umbellatus* (+) *Myriophyllum spicatum* (+)	*Carex stricta* (−) *C. aquatilis* (−) *Calamagrostis canadensis* (−) *Lythrum salicaria* (++) *Phragmites australis* (++) *Phalaris arundinacea* (++) *Polygonum lapathifolium* (++)	a. Absence of turbidity intolerant species. b. Relative dominance of species tolerant of low light levels in submergent zone. c. Loss of submergent species diversity. d. Relative dominance of perennials vs. annuals in wet meadow zone.
Physical Degradation		*Lythrum salicaria* (++) *Phragmites australis* (++) *Phalaris arundinacea* (++) *Polygonum lapathifolium* (++)	a. Major loss of species diversity. b. Elimination of natural zonation. c. Relative dominance of exotics and aggressive native species in wet meadow zone.

[1] Species responses are coded as: − Intolerant of stress; + Tolerant of stress; ++ Positive response to stress.

Plant-based metrics of water-level regulation

The relative dominance of *Typha* spp., along with measures of species diversity, provides a strong indicator of water level regulation within the Great Lakes basin. Although *Typha* is a widespread genus around the Great Lakes, mean cover values range from <5% in wetlands along Lake Superior, to nearly 30% along Lake Ontario and the St. Lawrence River (Figure 1). Regionally, the higher dominance of cat-tail in the coastal wetlands of Lake Ontario and the St. Lawrence River corresponds to manipulation of outflow from these bodies and the reduction in the amplitude of natural water-level fluctuations (Wilcox et al., 1993).[5]

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[5] While water level control is often referenced as the major factor leading to cat-tail dominance, nutrient enrichment probably also plays an important role in the extremely high levels of cat-tails in these regions, as agricultural land use in these regions is intensive.

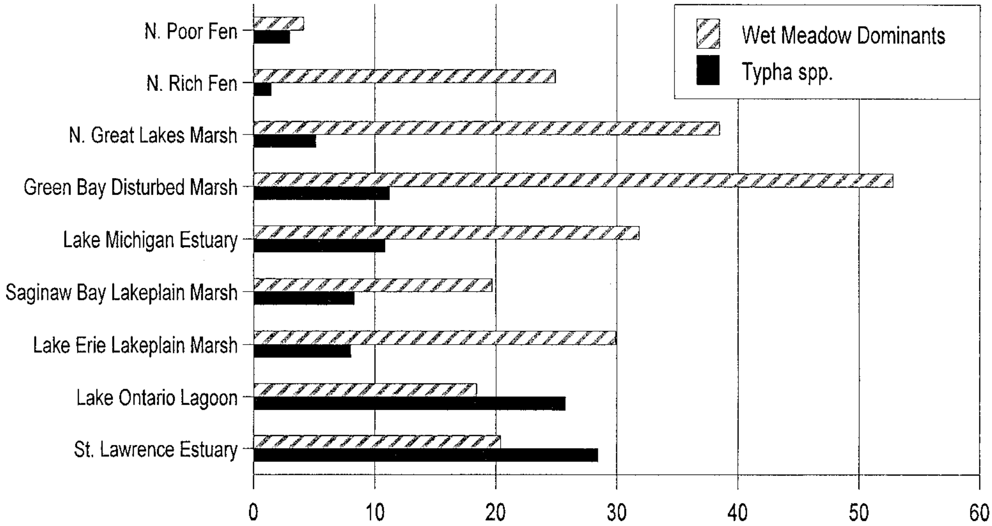

Figure 1. Plant-based indicators of water-level regulation: mean percent coverage value for *Typha* spp. vs typical wet meadow dominants, including *Calamagrostis canadensis*, *Carex stricta* and *Carex aquatilis*.

As cat-tail beds expand, overall plant species diversity is reduced (Geis, 1979; Wilcox et al., 1993) and the natural vegetation zonation modified. In the open emergent marsh and wet meadow, cat-tails have replaced typical wet meadow dominants (including *Calamagrostis canadensis*, *Carex stricta*, and *Carex aquatilis*), reducing important habitat for fish spawning habitat and waterfowl brood-raising along the St. Lawrence River and its tributaries. The diversity of the submergent marsh is similarly affected, as cat-tail colonization along the margins replaces wild rice and other submergent plants important for waterfowl (Geis, 1979; personal observations by author).

Expansion of cat-tail beds also decreases the amount of open water and submergent marsh in the wetlands, resulting in major habitat modification and loss for fish, especially northern pike (*Esox lucius*) (Casselman and Lewis, 1996). Expansion of the cat-tail zone has greatly decreased stream flow and increased the amount of undecomposed organic material, both factors reducing the oxygenation of the wetlands. This effects the utilization by pike and probably also greatly modifies the macro-invertebrate fauna of the wetlands.

Plant-based metrics of nutrient enrichment

A suite of submergent and emergent species that respond positively to increased nutrient loading are identified in Table 2. Examination of these species suggests that they can provide a measure of both regional and more local differences in nutrient loading within coastal wetlands. Regional differences in land use create strong differences in nutrient loading within the Great Lakes Basin. The southern Great Lakes, characterized by largely urban and agricultural environments, feature high nutrient loading, as sewage effluent, agricultural fertilizers, and fine-textured sediment are added to the Great Lakes. Nutrient loading increases from the upper lakes to the lower lakes, in the direction of water flow, such that western Lake Erie (Herdendorf et al., 1977), portions of Lake Ontario (Sager et al., 1998), and the St. Lawrence River experience high levels of nutrients. Corresponding alteration of the coastal wetland flora by nutrient enrichment can be seen in large portions of the southern Great Lakes. Nutrient-responding submergents expand their mean cover values to >40% along the St. Lawrence drowned river-mouth, and >65% in Lake Ontario lagoon wetlands.

In contrast, within the upper Great Lakes, low population and relatively extensive land-use patterns result in generally low levels of nutrient loading to coastal wetlands. Exceptions are Green Bay (Bosley, 1978) on Lake Michigan and Duluth/Superior on Lake Superior, where numerous or large wetland areas are heavily degraded by nutrient loading. Although mean submergent cover values can reach 40% in the relatively clean waters of the upper Great Lakes, nutrient-responding species (including *Myriophyllum spicatum*, *Potamogeton crispus*, *Potamogeton pectinatus*, *Elodea*

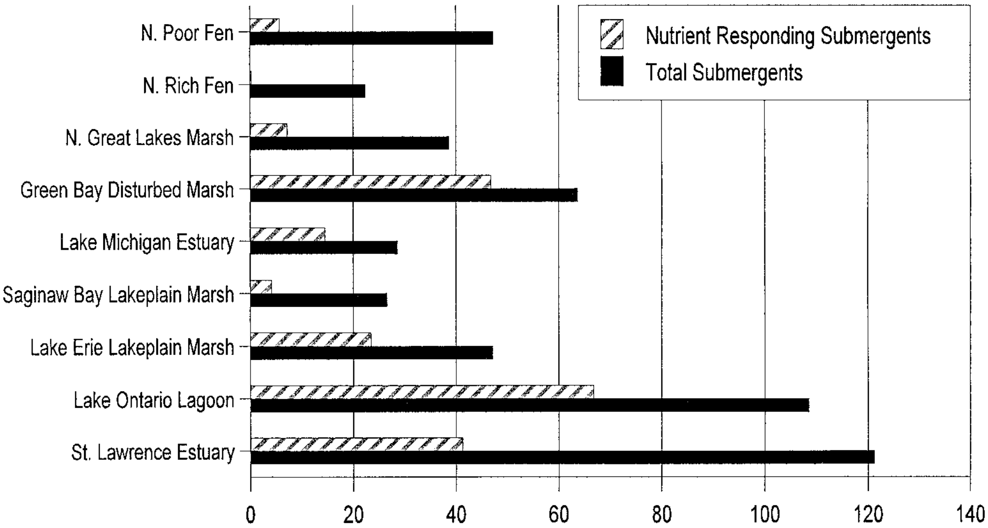

Figure 2. Plant-based indicators of nutrient enrichment: mean percent coverage value for nutrient responsive submergent species relative to total submergents (mean percent cover value). Coverages >100% indicate more than one layer of plants.

canadensis and *Ceratophyllum demersum*) are poorly represented, with mean coverage values of <10% (Figure 2).

Within the upper Great Lakes, however, nutrient and sediment loading can locally impact wetland quality, with a concomitant increase in coverage values for submergent aquatic plants. This local response was encountered at Cedarville within the Les Cheneaux Island chain in northern Lake Huron, an area with low levels of residential development along the shoreline. Cedarville Bay, however, receives low levels of effluent from a sewage plant twice annually, in May-June and September (Kashian and Burton, 2000). While nutrient levels rose during the periods of effluent introduction, there was little increase in turbidity. Sampling results indicated an enhanced growth response to the organic-rich effluent by submergent plants, including *Elodea canadensis*, *Ceratophyllum demersum*, *Ranunculus longirostris*, *Myriophyllum spicatum*, *Potamogeton natans*, *Vallisneria americana*, *Lemna trisulca* and *Utricularia vulgaris*. *Lemna trisulca*, *Lemna minor* and *Nuphar variegata* also had high coverage values on the water's surface, along with a large algal bloom (based on sampling by author, 1998–2002). The total submergent plant coverage was consistently 100%, with several structural layers of densely overlapping submergent species. The submergent coverage values were much greater than those encountered on any other coastal marsh sampled either in the Les Cheneaux Islands or elsewhere in the northern Lake Michigan or Lake Huron region. The area influenced by the effluent was quite small, and elevated nutrient levels were only seen within about 200 m of the effluent entry point, but the effect of even this low level of nutrient enrichment resulted in a demonstrable response for submergent and floating aquatic plants in this clear, cold northern coastal wetland.

Plant-based metrics of sedimentation

The impact of sedimentation is expected to be visible within both emergent and shoreline herbaceous vegetation. Attributes include the severe loss of plant diversity within the submergent zone, the relative dominance of submergent species more tolerant of low light levels, and the reduced representation of species intolerant of turbidity (Table 2). Changes in species composition are also expected in the emergent and wet meadow zones.

As in the case of nutrient loading, sedimentation within the Great Lakes largely corresponds to regional patterns of land use. Regional distributions of turbidity tolerant and intolerant plant species agrees with these larger scale patterns of wetland perturbation (Figure 3). In the upper Great Lakes, sites with typical low turbidity levels are marked by a relative dominance of turbidity intolerant submergent species, with mean coverage values for this class consistently greater than those for species tolerant of turbidity. In contrast, in Lake Erie, Lake Ontario, Green Bay, and Lake Michigan drowned river-mouth (estuary) wetlands, where high turbidity levels were noted, turbidity-tolerant submergent species were three or more times as abundant as

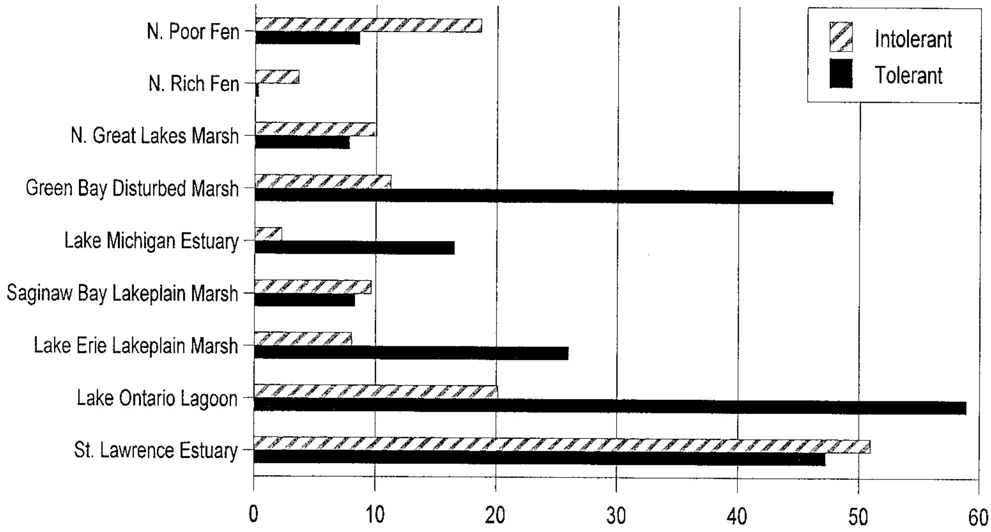

Figure 3. Plant-based indicators of water turbidity: mean percent coverage value for submergent species tolerant and intolerant of low-light levels.

turbidity-intolerant species, with mean coverage values exceeding 50%. Interestingly, wetlands within drowned river-mouths (for example, Crooked, Chippewa, and Cranberry creeks) along the St. Lawrence River also contained high levels of nutrient-intolerant submergent species. This was in agreement with our observations that water clarity was good in most of these wetlands, except near where the creeks entered the larger St. Lawrence River and turbidity increased greatly.

The effects of high sedimentation rates in the southern Great Lakes wetlands (along with Great Lakes water level fluctuations), are also apparent in the emergent and wet meadow zones. In the emergent marsh zone, *Schoenoplectus acutus*, *Schoenoplectus pungens*, and *Typha* spp. are absent from shallow water of the most turbid marshes. An open, low diversity submergent marsh is often all that remains of the emergent zone, and this is typically reduced to less than 50 meters in width. In contrast, maps of some of these same Lake Erie coastal marshes drawn by early surveyors show extensive wetlands that were several hundred meters wide and thousands of hectares in area.

Structural change

Regional differences in substrate, shoreline configuration, climate, and vegetation result in differential wetland loss and alteration (Minc and Albert, 1998). Along the Lake Superior shoreline, where the prevalence of rock or steep gravel substrate greatly limits the amount of wetland occurring along the shoreline, almost all wetlands occur behind a barrier beach. As a result, few of these wetlands have been eliminated, although these wetlands can still be degraded by shoreline residential development, or by alteration of water quality. In contrast, the silty, nutrient-rich soils along western Lake Erie are ideal for agricultural management. As a result, the coastal wetlands along Lake Erie have been heavily degraded, both by conversion of the wet meadow zones to cropland through ditching and diking, or by the heavy deposition of agricultural sediments in shallow coastal wetlands.

Two species most often seen in highly modified wetlands are *Phragmites australis* and *Lythrum salicaria*, which seed or spread by rhizomes into recently disturbed sediments. During our field data collection, we encountered these species growing on a wide variety of wetland sediments (including fly-ash from a power plant on Lake Erie). Along with the presence of native colonizing species, exotic plants provide a sensitive measure of mechanical wetland disturbance, especially in the wet meadow or shallow emergent zones.

Total coverage of exotic plants within wetlands differs greatly between regions (Figure 4). Total coverage of exotics is greatest in those regions with the highest levels of urbanization and agricultural development, including western Lake Erie, Green Bay on northern Lake Michigan, and Saginaw Bay on Lake Huron, areas where mean coverage can exceed 35%. However, exotics are widely present throughout the Great Lakes, with mean cover values of ca. 3% even in the

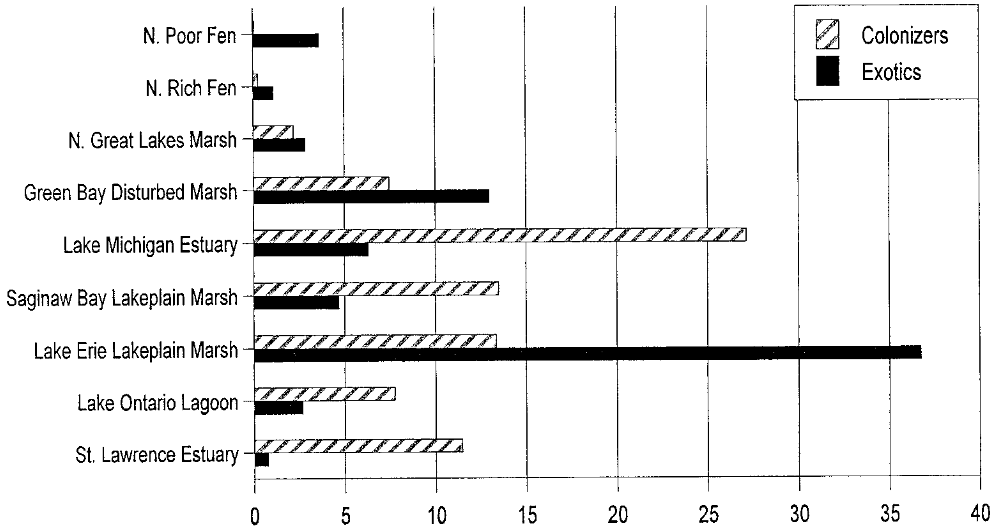

Figure 4. Plant-based indicators of physical degradation: mean percent coverage value for exotic shoreline species and native colonizing species.

relatively pristine wetlands of the upper Great Lakes. There are also native colonizing species whose populations can expand rapidly in response to sedimentation or rapid drops in water level in a manner similar to exotic species. Among these native colonizers are *Polygonum lapathifolium*, *Bidens cernuus*, *Impatiens capensis*, *Leersia orizoides* and *Rorippa palustris*. However, unlike the exotics, these colonizers are typically soon replaced by diverse native wetland communities as the water-level regime in the wetlands change.

Conclusions

Using existing literature, we identified species considered to be sensitive to specific dimensions of wetland degradation. We then evaluated the distribution and density of these species within our Great-Lakes-wide wetland database to determine if these potential indicator species or groups showed identifiable relationships to known variability in anthropogenic stress. The analyses demonstrated that types of disturbance varied regionally and that our proposed indicators showed predictable regional response to disturbance. These same indicators are also sensitive to variations of disturbance level within regions, as illustrated by the submergent species response to local effluent in the Cedarville wetland in northern Lake Huron.

While the species groups that we evaluated showed responses consistent with the wetland literature, several species were considered indicators for more than one type of disturbance. For example, *Potamogeton pectinatus*, *Potamogeton crispus* and *Ceratophyllum demersum* were considered indicators of both nutrient enrichment and increased turbidity and sedimentation. Likewise, *Typha* spp. were considered indicators of both nutrient enrichment and dampening of water-level fluctuations. Having a species or group of species that indicates more than one type of disturbance complicates a specie's value as an environmental indicator. This may reduce an indicator's usefulness for making specific mitigation and restoration decisions.

While species groups were identified that indicated dampening of water-level fluctuations, nutrient enrichment, increased sedimentation and turbidity, and mechanical disturbance within Great Lakes wetlands, effective indicators for chemical pollution were not identified. Reduced species richness and diversity may reflect high levels of contaminants; however, we have yet to identify indicator species or species groups sensitive to heavy metals and other chemical pollutants within the Great Lakes region. Thus, a future challenge for Great Lakes wetland studies will be the identification of plant-based metrics for chemical pollutants as separable from other dimensions of stress on aquatic plant communities.

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References

Albert, D. A., Reese, G., Crispin, S., Wilsmann, L. A., Ouwinga, S. J., 1987. A survey of Great Lakes marshes in Michigan's Upper Peninsula. Michigan Natural Features Inventory, Lansing, MI.

Albert, D. A., Reese, G., Crispin, S., Penskar, M. R., Wilsmann, L. A., Ouwinga, S. J., 1988. A survey of Great Lakes marshes in the southern half of Michigan's Lower Peninsula. Michigan Natural Features Inventory, Lansing, MI.

Albert, D. A., Reese, G., Penskar, M. R., Wilsmann, L. A., Ouwinga, S. J., 1989. A survey of Great Lakes marshes in the northern half of Michigan's Lower Peninsula and throughout Michigan's Upper Peninsula. Michigan Natural Features Inventory, Lansing, MI.

Albert, D. A., Minc, L. D., 2001. Abiotic and floristic characterization of Laurentian Great Lakes' coastal wetlands. Verh. Internat. Verein. Limnol. 27, 3413–3419.

Barko, J. W., Smart, R. M., Mathews, M. S., Hardin, D. G., 1982. Sediment-submersed macrophyte relationships in freshwater systems. Tech. Report A-82-3, U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Barko, J. W., Adams, M. S., Clesceri, N. L., 1986. Environmental factors and their consideration in the management of submersed aquatic plants. J. Aquat. Plant Manage. 24, 1–10.

Bosley, T. R., 1978. Loss of wetlands on the west shore of Green Bay. Wisc. Acad. Sci. Arts Lett. 66, 235–245.

Bosserman, R. W., 1985. Distribution of heavy metals in aquatic macrophytes from Okefenokee swamp. In: J. Salánki (Ed.), *Heavy Metals in Water Organisms*, pp. 31–40. Akadémiai Kiadó, Budapest, Hungary.

Burton, T. M., Uzarski, D. G., Gathman, J. P., Genet, J. A., Keas, B. E., Stricker, C. A., 1999. Development of a preliminary invertebrate index of biotic integrity for Lake Huron coastal wetlands. Wetlands 19, 869–882.

Busch, W. D., Lewis, C. N., 1984. Responses of wetland vegetation to water level variations in Lake Ontario. Proceedings of the Third Annual Conference on Lake and Reservoir Management, pp. 519–523. N. Amer. Lake Manage. Soc., U.S. EPA, Washington, D.C.,

Carter, V., Rybicki, N. B., 1985. The effects of grazers and light penetration on the survival of transplants of *Vallisneria americana* Michx. in the tidal Potomac River, Maryland. Aquat. Bot. 23, 197–213.

Casselman, J. M., Lewis, C. A., 1996. Habitat requirements of northern pike (*Esox lucius*). Can. J. Fish. Aquat. Sci. 53(Suppl. 1), 161–174.

Chow-Fraser, P., 1998. A conceptual model to aid restoration of Cootes Paradise Marsh, a degraded coastal wetland of Lake Ontario, Canada. Wetl. Ecol. Manage. 6, 43–57.

Chow-Fraser, P., Albert, D. A., 1998. Biodiversity Investment Areas: Coastal Wetlands Ecosystems. Identification of "Eco-reaches" of Great Lakes Coastal Wetlands that have high biodiversity value. A discussion paper for the State of the Lakes Ecosystem Conference, 1998. U.S. EPA, Chicago, IL and Environment Canada, Burlington, ON.

Crivelli, A. J., 1983. The destruction of aquatic vegetation by carp. Hydrobiol. 106, 37–41.

Estabrook, G. F., Burk, D. W., Inman, D. R., Kaufman, P. B., Wells, J. R., Jones, J. D., Ghosheh, N., 1980. Comparison of heavy metals in aquatic plants on Charity Island, Saginaw Bay, Lake Huron, USA, with plants along the shoreline of Saginaw Bay. Am. J. Bot. 72, 209–216.

Fahnenstiel, G. L., Bridgeman, T. B., Lang, G. A., McCormick, M. J., Nalepa, T. F., 1995. Phytoplankton productivity in Saginaw Bay, Lake Huron: Effects of zebra mussel (*Dreissena polymorpha*) colonization. J. Great Lakes Res. 21(4), 465–475.

Francis, G. R., Magnuson, J. J., Regier, H. A., Talhelm, D. R. (Eds.), 1979. Rehabilitating Great Lakes Ecosystems. Great Lakes Fishery Commission, Tech. Rep. No. 37.

Geis, J. W., 1979. Shoreline processes affecting the distribution of wetland habitat. pp. 529–542. Trans. N. Amer. Wildlife Nat. Res. Conference 44.

Gernes, M., Helgen, J. C., 1999. Indexes of biotic integrity (IBI) for wetlands: Vegetation and invertebrate IBIs. Final Report to U.S. EPA, Assistance Number CD995525-01. Minnesota Pollution Control Agency, Environmental Outcomes Division, St. Paul, MN.

Greger, M., Kautsky, L., 1991. Effects of Cu, Pb and Zn on two *Potamogeton* species grown under field conditions. Vegetatio 97, 173–184.

Harris, H. J., Fewless, G., Milligan, M., Jownson, W., 1981. Recovery processes and habitat quality in a freshwater coastal marsh following a natural disturbance. In: B. Richardson (Ed.), *Selected Proceedings of the Midwest Conference on Wetland Values and Management*, pp. 363–379. Minnesota Water Planning Board, St. Paul, MN.

Herdendorf, C. E., 1987. The ecology of the coastal marshes of western Lake Erie: A community profile. U.S. Fish and Wildlife Biol. Rep. 85(7.9).

Herdendorf, C. E., 1983. Lake Erie water quality 1970–1982: A management assessment. Cent. Lake Erie Area Res. Tech. Rep. 298. Ohio State Univ., Columbus, OH.

Herdendorf, C. E., Rathke, D. E., Larson, D. D., Fay, L. A., 1977. Suspended sediment and plankton relationships in Maumee River and Maumee Bay of Lake Erie. In: R. C. Romans (Ed.), *Geobotany*, pp. 247–282. Plenum Press, New York, NY.

Jaworski, E., Raphael, C. N., 1976. Modification of coastal wetlands in southeastern Michigan and management alternatives. Mich. Acad. 8, 303–317.

Jaworski, E., Raphael, C. N., Mansfield, P. J., Williamson, B. B., 1979. Impact of Great Lakes Water Level Fluctuations on Coastal Wetlands. U.S.D.I. Office of Water Resources and Technology, Contract Report 14-0001-7163. Institute of Wetland Research, Michigan State University, East Lansing.

Karr, J. R., 1981. Assessment of biotic integrity using fish communities. Fisheries 6(6), 21–27.

Karr, J. R., Chu, E. W., 1997. Biological monitoring and assessment using multimetric indexes effectively. EPA-235-R97-001. U.S. EPA, University of Washington, Seattle, WA.

Karr, J. R., Fausch, K. D., Angermeier, P. L., Yant, P. R., Schlosser, I. J., 1986. Assessing Biological Integrity in Running Waters:

A Method and its Rationale. Illinois Natural History Survey. Special Publ. 5, Champaign, Ill., USA.

Kashian, D. R., Burton, T. M, 2000. A comparison of macroinvertebrates of two Great Lakes coastal wetlands: testing potential metrics for an index of ecological integrity. J. Great Lakes Res., 26(4), 460–481.

Keddy, P. A., 1989. *Competition*. Chapman and Hall, London.

Keddy, P. A., 1992. Water level fluctuations and wetland conservation. In: J. Kusler, R. Smardon (Eds.), *Wetlands of the Great Lakes: Protection, Restoration, Policies and Status of the Science,* pp. 79–91. Proceedings of the International Wetland Symposium, May 16–19, 1990, Niagara Falls, NY.

Keough, J., 1987. Response by *Scirpus validus* to the physical environment and consideration of its role in a Great Lakes estuarine system. Ph.D. thesis, Univ. of Wisconsin, Milwaukee.

Keough, J. R., 1992. The range of water level changes in a Lake Michigan estuary and effects on wetland communities. In: J. Kusler, R. Smardon (Eds.), *Wetlands of the Great Lakes: Protection, Restoration, Policies and Status of the Science,* p. 97–110. Proceedings of the International Wetland Symposium, May 16–19, 1990, Niagara Falls, NY.

Kimbel, J. C., 1982. Factors influencing potential intralake colonization by *Myriophyllum spicatum* L. Aquat. Bot. 14, 295–307.

Lewis, M. A., Wang, W., 1999. Biomonitoring using aquatic vegetation. In: A. Gerhardt (Ed.), *Biomonitoring of Polluted Water*, pp. 243–273. Environmental Research Forum vol. 9. Trans Tech Publications, Zurich, Switzerland.

Lyon, J. G., Drobney, R. D., Olson, C. E., Jr., 1986. Effects of Lake Michigan water levels on wetland soil chemistry and distribution of plants in the Straits of Mackinac. J. Great Lakes Res. 12, 175–183.

Manny, B. A., Nichols, S. J., Schloesser, D. W., 1991. Heavy metals in aquatic macrophytes drifting in a large river. Hydrobiol. 219, 333–344.

Minc, L. D., 1997a. Great Lakes coastal wetlands: An overview of abiotic factors affecting their distribution, form, and species composition. A report in 3 parts. Michigan Natural Features Inventory, Lansing, MI.

Minc, L. D., 1997b. Vegetative response in Michigan's Great Lakes marshes to Great Lakes water-level fluctuations. Michigan Natural Features Inventory, Lansing, MI.

Minc, L. D., Albert, D. A., 1998. Great Lakes coastal wetlands: Abiotic and floristic characterization. Michigan Natural Features Inventory. Lansing, MI.

Nalepa, T. F., Fahnenstiel, G. L., Johengen, T. H., 1999. Impacts of the zebra mussel (*Dreissena polymorpha*) on water quality: A case study in Saginaw Bay, Lake Huron. In: R. Claudi, J. H. Leach (Eds.), *Non-Indigenous Freshwater Organisms in North America: Their Biology and Impact*, pp. 255–271. CRC Press, Lewis Publishers, Boca Raton, FL.

Niemeier, P. E., Hubert, W. A., 1986. The 85-year history of the aquatic macrophyte species composition in a eutrophic prairie lake (United States). Aquat. Bot. 25, 83–89.

Patterson, N. J., Whillans, T. H., 1985. Human interference with natural water level regimes in the context of other cultural stresses on Great Lakes wetlands. In: H. H. Prince, F. M. D'Itri (Eds.), *Coastal Wetlands*, pp. 209–251. Lewis Press, Chelsea, MI.

Phillips, D. J. H., 1978. Use of biological indicator organisms to quantitate organochlorine pollutants in aquatic environments—a review. Environ. Pollut. 16, 167–229.

Potter, K., Lovett-Doust, L., 1997. *Vallisneria americana* (American wild celery) as a biomonitor of site quality in the Great Lakes "Areas of Concern." Am. J. Bot. 84(6), abstract.

Reimer, P., Duthie, H. C., 1993. Concentations of zinc and chromium in aquatic macrophytes from the Sudbury and Muskoka regions of Ontario, Canada. Environ. Pollut. 79, 261–265.

Robb, D. M., 1989. Diked and undiked freshwater coastal marshes of western Lake Erie. M. S. thesis, Ohio State University, Columbus, OH.

Rorslett, B., Berge, D., Johansen, S. W., 1986. Lake enrichment by submersed macrophytes: A Norwegian whole-lake experience with *Elodea canadensis*. Aquat. Bot. 26, 325–340.

Sager, E. P. S., Whillans, T. H., Fox, M. G., 1998. Factors influencing the recovery of submersed macrophytes in four coastal marshes of Lake Ontario. Wetlands 18(2), 256–265.

Scheffer, M., De Redelijkheid, M. R., Noppert, F., 1992. Distribution and dynamics of submerged vegetation in a chain of shallow eutrophic lakes. Aquat. Bot. 42, 199–216.

Simon, T. P., Stewart, P. M., Rothrock, P. E., 2001. Development of multimetric indices of biotic integrity for riverine and palustrine wetland plant communities along southern Lake Michigan. Aquat. Ecosys. Health Manage. 4(3), 293–309.

Srivastava, D. A., Staicer, C. A., Freedman, B., 1995. Aquatic vegetation of Nova Scotian lakes differing in acidity and trophic status. Aquat. Bot. 51, 181–196.

Stewart, P. M., 1995. Use of algae in aquatic pollution assessment. Nat. Areas J. 15, 234–239.

Stewart, P. M., Butcher, J. T., Simon, T. P., 2003. Response signatures of four biological indicators to an iron and steel industrial landfill. In: T. P. Simon (Ed.), *Biological Response Signatures*, pp. 419–444. CRC Press, New York, NY.

Stewart, P. M., Scribailo, R. W., Simon, T. P., 1999. The use of aquatic macrophytes in monitoring and in assessment of biological integrity. In: A. Gerhardt (Ed.), *Biomonitoring of Polluted Water*, pp. 275–302. Environmental Research Forum vol. 9. Trans Tech Publications, Zurich, Switzerland.

Stuckey, R. L., 1975. A floristic analysis of the vascular plants of a marsh at Perry's Victory Monument, Lake Erie. Mich. Bot. 14, 144–166.

Stuckey, R. L., 1989. Western Lake Erie aquatic and wetland vascular plant flora: its origin and change. In: Lake Erie Estuarine Systems: Issues, Resources, Status, and Management, pp. 205–256. NOAA Estuary-of-the-Month Seminar Series No. 14. U.S. Department of Commerce, National Oceanic and Atmospheric Administration. Washington, D.C.

Toivonen, H., Huttunen, P., 1995. Aquatic macrophytes and ecological gradients in 57 small lakes in southern Finland. Aquat. Bot. 51, 197–221.

Tubea, B., Hawxby, K., Mehta, R., 1981. The effects of nutrients, pH and herbicide levels on algal growth. Hydrobiol. 79, 221–227.

U.S. Army Corps of Engineers, 1987. Great Lakes Water Level Facts. U.S. Army Corps of Engineers, Detroit District, Detroit, MI.

U.S. Army Corps of Engineers, 1997. Frequently Asked Questions. Great Lakes Update 128. U.S. Army Corps of Engineers, Detroit District, Detroit, MI.

van der Valk, A. G., 1981. Succession in wetlands: A Gleasonian approach. Ecol. 62, 688–696.

van Dijk, G. M., van Vierssen, W., 1991. Survival of a *Potamogeton pectinatus* L. population under various light conditions in a shallow eutrophic lake (Lake Veluwe) in The Netherlands. Aquat. Bot. 39, 121–129.

Wells, J. R., Kaufman, P. B., Jones, J. D., 1980. Heavy metal contents in some macrophytes from Saginaw Bay (Lake Huron, U.S.A.). Aquat. Bot. 9, 185–193.

Wilcox, D. A., Meeker, J. E., Elias, J., 1993. Impacts of water-level regulation on wetlands of the Great Lakes. Phase 2 Report to Working Committee 2, International Joint Commission, Great Lakes Water Level Reference Study, Natural Resources Task Group.

Williams, D. C., Lyon, J. G., 1997. Historical aerial photographs and a geographic information system (GIS) to determine effects of long-term water level fluctuations on wetlands along the St. Marys River, Michigan, USA. Aquat. Bot. 58, 363–378.

Yoder, C. O., Rankin, E. T., 1995. Biological criteria program development and implementation in Ohio. In: W. S. Davis, T. P. Simon (Eds.), *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*, pp. 109–144. Lewis Publishers, Boca Raton, FL.