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TECHNIQUES FOR RESTORATION OF DISTURBED COASTAL WETLANDS OF THE GREAT LAKES

Douglas A. Wilcox[1] and Thomas H. Whillans[2]
[1] *Great Lakes Science Center*
U.S. Geological Survey
1451 Green Rd.
Ann Arbor, Michigan, USA 48105
email: douglas_wilcox@usgs.gov

[2] *Environmental and Resource Studies*
Trent University
Peterborough, Ontario, Canada K9J 7B8

Abstract: A long history of human-induced degradation of Great Lakes wetlands has made restoration a necessity, but the practice of wetland restoration is relatively new, especially in large lake systems. Therefore, we compiled tested methods and developed additional potential methods based on scientific understanding of Great Lakes wetland ecosytems to provide an overview of approaches for restoration. We addressed this challenge by focusing on four general fields of science: hydrology, sedimentology, chemistry, and biology. Hydrologic remediation methods include restoring hydrologic connections between diked and hydrologically altered wetlands and the lakes, restoring water tables lowered by ditching, and restoring natural variation in lake levels of regulated lakes Superior and Ontario. Sedimentological remediation methods include management of sediment input from uplands, removal or proper management of dams on tributary rivers, and restoration of protective barrier beaches and sand spits. Chemical remediation methods include reducing or eliminating inputs of contaminants from point and non-point sources, natural sediment remediation by biodegradation and chemical degradation, and active sediment remediation by removal or by *in situ* treatment. Biological remediation methods include control of non-target organisms, enhancing populations of target organisms, and enhancing habitat for target organisms. Some of these methods were used in three major restoration projects (Metzger Marsh on Lake Erie and Cootes Paradise and Oshawa Second Marsh on Lake Ontario), which are described as case studies to show practical applications of wetland restoration in the Great Lakes. Successful restoration techniques that do not require continued manipulation must be founded in the basic tenets of ecology and should mimic natural processes. Success is demonstrated by the sustainability, productivity, nutrient-retention ability, invasibility, and biotic interactions within a restored wetland.

Key Words: biology, chemistry, Great Lakes, hydrology, sedimentology, wetland restoration

INTRODUCTION

Great Lakes coastal wetlands occur in several geomorphic settings in which wetland hydrology is determined by lake level. These include open shoreline, unrestricted bay, shallow sloping beach, river delta, restricted riverine or drowned-river-mouth, lake-connected inland, and barrier beach wetlands (ILERSB 1981, Maynard and Wilcox 1997). Because the Great Lakes span a broad geographic range, these lake-affected wetlands can vary from floating peatlands in oligotrophic Lake Superior in the north to marshes with emergent, floating, and submersed vegetation along more southerly lakeshores. Great Lakes wetlands are subject to a number of natural stressors, including changes in water-level and sediment supply, physical damage from ice and storms, and actions of native biota (Maynard and Wilcox 1997). Wetlands have long-survived those stresses. However, when human-induced stressors are added, wetland degradation can occur. Those stressors include the range of human activities associated with wetlands in most settings (e.g., filling, clearing, excavating, contamination, non-indigenous species) (Conservation Foundation 1988). However, Great Lakes wetlands also face problems from water-level regulation, shoreline modification, dike construction, and other physical alterations often associated with centers of human population (Maynard and Wilcox 1997), which are prevalent along many parts of the Great Lakes shoreline. Thus, degradation and loss of wetlands have occurred, and resource managers have recognized the need for wetland restoration. The relatively recent recognition of wetland functions and values has also resulted in regulatory reforms that often dictate restoration of wetlands. However, wetland restoration as a branch of ecology and as a practice is still in its infancy; this is especially true along the shores of the Great Lakes.

Clewell (1993) described restoration ecology as a broad subset of the entire field of ecology, including its theories, tenets, and body of knowledge, whereas ecological restoration is the practice of restoring and managing ecosystems. The practice requires vision of what the restored ecosystem should look like, understanding of the ecological processes needed to restore and maintain the ecosystem, and specific skills and techniques necessary to carry out the work (Anderson 1996). Other specific components include identifying stresses that are degrading the ecosystem, formulating realistic goals, establishing reference sites and measures of success, conducting experiments to test ideas and methods, monitoring results, and further testing or readjusting methods to achieve the goals (Bradshaw 1993, Hobbs and Norton 1996). Some of the above components are addressed (in this issue) in relation to the Great Lakes by French et al. (1999), Keough et al. (1999), and Kowalski and Wilcox (1999). In this paper, we focus on the techniques–the ideas and methods that have been tested and those that derive from scientific understanding of the ecosytem but require further investigation for use in restoration. This effort is not meant to be a textbook providing specific details of methods, but instead, we present an overview of approaches (with examples) in four general fields of science: hydrology, sedimentology, chemistry, and biology. The examples derive from work on all of the lakes; however, some may not be applicable in all wetlands. We then provide case studies of three of the most prominent wetland restoration projects in the Great Lakes: Metzger Marsh on Lake Erie and Cootes Paradise and Oshawa Second Marsh on Lake Ontario. These projects are not completed, and they do not use all of the techniques described, but the case studies are useful descriptions of potential applications.

APPROACHES TO RESTORATION

Hydrologic Remediation

Restoration of coastal wetlands in the Great Lakes is contingent on hydrology and the physical functions that determine or are determined by hydrology. Lake-level change is the predominant hydrologic factor in the Great Lakes. Seiches with an amplitude of 20 to 30 cm and period of 4 to 14 hours occur regularly on the lakes or within large embayments of the lakes. Extreme seiches have been recorded on Lake Erie with amplitudes as great as 5 m (S. Mackey, pers. comm.). Annually, high lake levels occur in early summer and low lake levels in early winter. The range between record summertime high and wintertime low water-levels from 1918 to the present varied from as little as 1.19 m on Lake Superior to as much as 2.04 m on Lake St. Clair (USACE 1999). Mean annual water levels during that period varied from 0.30 m on hydrologically connected lakes Michigan/Huron to 0.53 m on Lake Ontario. However, sedimentological studies of chronosequences of beach ridges that formed along the shore of Lake Michigan as shorelines prograded during the late Holocene have shown considerably greater fluctuations, with quasi-periodic behavior at two levels of variation. During the past 4700 years, short-term fluctuations with a range of 0.5 to 0.6 m occurred about every 30 years, and longer-term fluctuations with a range of 0.8 to 0.9 m occurred about every 150 years (Thompson et al. 1991, Thompson 1992, Baedke and Thompson 1993). Long-term fluctuations that exceed those observed in the historical record from 1918 to 1996 likely occurred on the other lakes also.

Given the wide variation in lake levels and the known response of wetland plants to water-level changes (Meeks 1969, Cooke 1980, van der Valk and Davis 1980, Spence 1982, Wilcox and Meeker 1991), hydrology is the single most important overall factor affecting the composition and structure of wetland vegetation in Great Lakes coastal marshes (Keddy and Reznicek 1986, Wilcox et al. 1993, Wilcox 1995). Efforts to restore these marshes have therefore focused on hydrology; however, most past efforts were directed at controlling hydrology, not restoring it.

Construction of dikes and ditching have most often been used to control hydrology. Ditches are most prominent in low-lying, flat lands adjacent to the lakes that are now in use for agricultural production. Examples include areas around Saginaw Bay of Lake Huron and the Ontario and Ohio shores of western Lake Erie. In the Great Lakes, the percentage of coastal wetlands controlled by diking ranges from 3% in Lake Ontario to 31% in Lake Erie and 33% in Lake St. Clair. However, within Ohio, 77% of the Lake Erie coastal marshes are diked and another 11% are now used as diked farmland (Bookhout et al. 1989). Diked wetlands are typically managed by using occasional drawdowns to expose sediments and elicit a response from the seedbank, thus restoring emergent vegetation. Other water-level manipulations are used to promote growth of certain plant taxa, discourage growth of other taxa, or to make plants, seeds, or invertebrates available to wildlife at specific times of the year. Water-control structures associated with diked wetlands include one-way and two-way flapgate culverts, screwgate culverts, fixed-crest wiers, and stop-log variable crest wiers (see Broussard 1988). Water levels are also manipulated by pumping with fixed and portable pumps powered by diesel, gasoline, or electricity (G. Tori, pers. comm.) The use of dikes and control structures typically isolates coastal wetlands from the lakes, however, and converts them to inland wetlands adjacent to the lakes. Although dikes have been used effectively for managing waterfowl habitat along the Great Lakes shore, this technique presents an obvious deterrent to fish migration, nutrient transport, and other wetland processes dependent on hydrologic connection with the lake.

Restoration of hydrology in coastal wetlands and its resultant restoration of other natural functions and processes can be more clearly defined at three levels: 1) restoration of hydrologic connections between diked and hydrologically altered wetlands and the lakes, 2) restoration of wetland water tables lowered by ditching, and 3) restoration of natural variation in lake levels on regulated lakes (Superior and Ontario).

Restoration of Hydrologic Connections. Preliminary investigations into methods for successfully reestablishing hydrologic connections and natural water-level fluctuations in diked marshes led to the approach being tested at Metzger Marsh in western Lake Erie, which is described as a case study in a later section of this paper. Although we do not encourage construction of new permanent dikes, temporary dikes in selected wetlands with relatively narrow hydrologic connections to the lake can be accomplished with aquadams. These large, long, synthetic bags of water with a height greater than the water column can be placed on the bottom of a wetland to isolate it temporarily from the lake. Pumps can then be used to drain the wetland and allow planting or natural regeneration of the plant community from the seed bank. Aquadams are rather expensive, require considerable labor to install, and are limited by topography and bottom materials. They have not been widely tested but present obvious potential.

Other wetlands, including most of the drowned-river-mouth wetlands along the east shore of Lake Michigan, remain connected to the lake. However, flow is restricted to passage under relatively narrow bridges built into road beds that cross the downstream portion of the wetlands. During high flow periods, this restriction causes waters to slow and excess sediments to settle in the wetlands; the result is a change in habitat and alteration of wetland plant communities (D. Wilcox, pers. obs.). Hydrologic restoration of these wetlands would require increasing the width of the bridge spans or adding additional bridges or culverts to the road bed.

Restoration of Water Table. Ditches through wetlands or those connecting wetlands to the lakes can lower water tables at higher elevations during low water years when lake level is not influencing hydrology directly. This disturbance can be addressed by filling in the ditches, blocking them at their outlets, and redirecting flow away from them. Such actions are complex, however, because they can involve both surface and ground water, may affect upstream lands in private ownership, and may not result in pre-ditching conditions due to burning or subsidence of dried wetland sediments or potential loss of the pre-ditching seed bank.

Restoration of Lake-Level Fluctuations. Proposals for restoring natural variation in water levels of regulated lakes were made for Lake Ontario as part of the Water Levels Reference Study of the International Joint Commission (Wilcox et al. 1993). Lake Ontario water levels have been regulated since 1960, coincident with operation of the St. Lawrence Seaway. Prior to regulation, the amplitude of fluctuations from wintertime low to summertime high was as great as 2 m and varied widely and frequently. The current regulation plan calls for a reduction in maximum amplitude to 1.22 m, eliminates all year-to-year variation, and maintains lower-than-normal water levels in the spring (Figure 1a). A preliminary water-level regulation scenario for Lake Ontario that would benefit wetland communities was derived from actual lake levels from the period 1900–1989, with numerous modifications to achieve a fluctuation pattern more consistent with pre-regulation conditions (Figure 1b). In this preliminary scenario, the amplitude between the highest summer level and the lowest winter level is 1.98 m, and peak summertime highs reach 75.56 m [IGLD 1985] approximately every 13 years, with some intervening high years. Reductions to summertime highs of 74.52 m are achieved in 2 successive years between the peak highs. Natural timing of winter drawdowns is restored, and levels in mid-March reach 74.98 m or greater whenever possible to allow fish access to wetlands for spawning (Wilcox et al. 1993).

In a restoration context, the potential effects of implementing the above scenario on wetland plant communities of Lake Ontario were projected for the 1900–1989 period using two-dimensional, wetland topographic models derived by merging surveyed profiles of study sites. One model was based on field studies of wetlands with sandy inorganic substrates and a steeper gradient that were located in areas exposed to wave action. The other model was based on studies of wetlands with organic substrates and a flatter profile in areas protected from wave action. Plant communities at the sites were sampled along transects that fol-

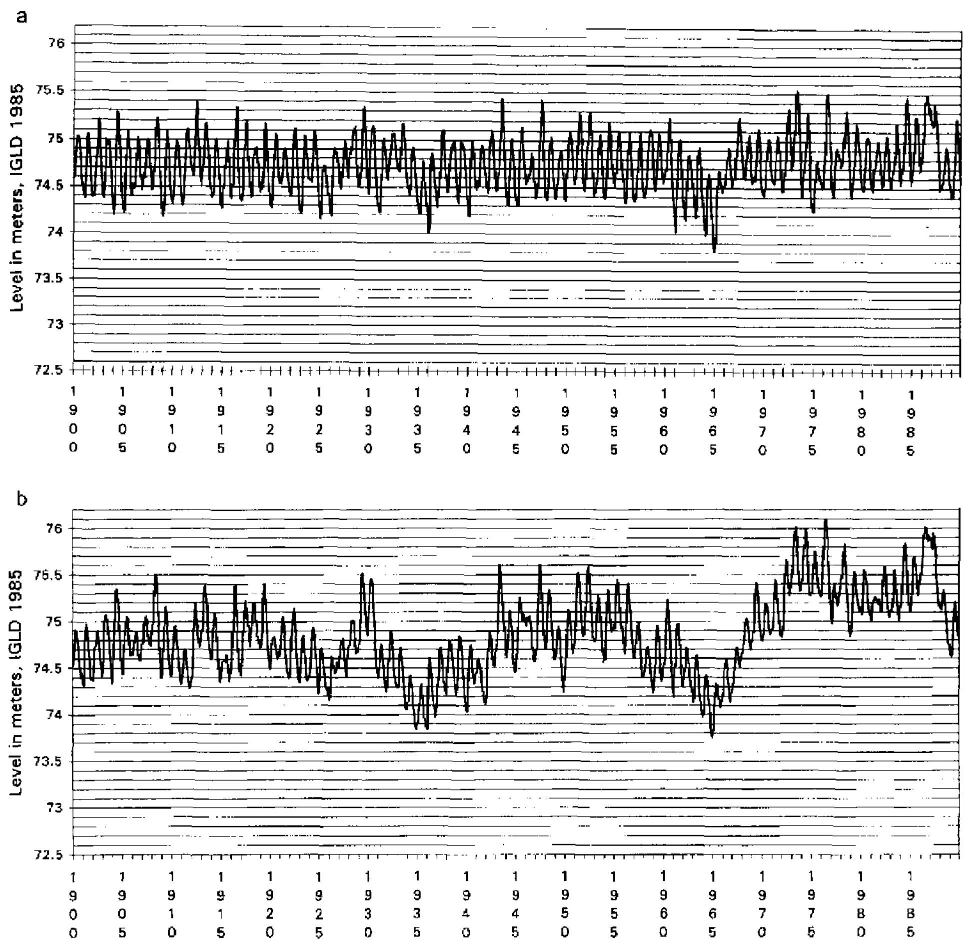

Figure 1. Regulation plans for controlling water levels on Lake Ontario: a) Plan 58D currently in effect, b) preliminary proposed Environment Plan. These plans project future lake levels based on water supplies that occurred from 1900 through 1989.

lowed topographic contours with different water-level histories (number of years since last dewatering or last flooding). Results of the transect sampling were then overlain on the topographic models, time-weighted, and relativized to allow prediction of relative proportions of the respective wetland plant communities expected to occur across the profiles (Wilcox et al. 1993).

If the present regulation plan (Figure 1a) remains in place, the modeling exercise indicated that stable plant communities would likely develop at several elevations. In steeper gradient wetlands with inorganic substrates, old-field species, shrubs, and grasses would account for 10% of the plant community; an aquatic community with submersed, floating, and some deep-water emergent taxa would total 79%. Wet meadow/emergent communities with the greatest diversity would account for the remaining 11%. Purple loosestrife (*Lythrum salicaria* L.) would likely increase its dominance in wetlands as a result of stable water levels (Wilcox et al. 1993). In lower gradient wetlands with organic substrates, trees and shrubs would account for 8%, the aquatic community 62%, and shrubs, grasses, and forbs 30% of the plant community. The stability of water levels would likely increase the dominance of hybrid cattail (*Typha* × *glauca* Godr.) at these sites (Wilcox et al. 1993).

If the water-level-regulation scenario proposed in Figure 1b were implemented, the modeling exercise indicated that in wetlands with inorganic substrate, old-field, shrub, and grass communities would decrease to 6% and aquatics to 69%, while wet-meadow/emergent communities would increase to 21%. In wetlands with organic substrate, these figures would change to 4%, 55%, and 38%, respectively. Under both models, the percent of wetland occupied by the more stable plant communities found at higher and lower elevations would decrease, and the more diverse communities at middle elevations with increased exposure to both flooding and dewatering episodes would increase. Exposed wetlands with inorganic substrates would show the greatest conversion to wet-meadow/emergent communities.

Sedimentological Remediation

Effective control of coastal sediment problems, thus potentially resulting in wetland restoration, includes management of sediment input from upland and nearshore sources. Examples include proper erosion control on agricultural lands, restoration of ditched wetlands, removal or proper management of dams on tributary rivers, and effective restoration and management of beaches, dunes, ridge and swale systems, barrier beaches, and sand spits.

Many coastal wetlands in the Great Lakes occur in areas protected from wave energy by barrier beaches, sand spits, or shallow sloping beaches. Protection and restoration of wetlands may thus be highly dependent on coastal processes such as sediment erosion, transport, and deposition that form and maintain these natural features. Water levels and nearshore currents also affect coastal processes. In lakeshore regions with substantial human development near the shore, the threat of erosion, especially during periods of high lake levels, has resulted in construction of protective structures to halt erosion and remediation structures intended to maintain or restore beaches threatened by erosion. Dikes have also been built to protect wetlands from erosion. However, large-scale armoring of the shoreline by dikes and various forms of breakwalls and revetments can negatively affect maintenance of barrier beaches and sand spits that protect wetlands in nearby coastal areas. Structures such as groins and segmented breakwaters have been used in attempts to protect or restore beaches, as has beach nourishment. These practices have been considered for wetland restoration but have not been implemented. A review of the consequences of implementing these measures in coastal areas, as provided by Silvester and Hsu (1991), suggests that great caution should be taken before they are considered seriously for wetland restoration.

Breakwalls, seawalls, or revetments that armor the shoreline have been promoted as reflecting waves back to the sea (lake), never to be seen again. However, the reflected waves interact with the lake floor and incoming waves to create a complicated flow system. The result is expedited scour of sediments in front of the breakwall, strong mass transfer parallel to the wall, transfer of energy downcoast from the wall, and increased erosion downcoast (Silvester and Hsu 1991). A wetland-associated result has been termed "backstopping," which refers to the inability of a wetland to establish on the landward portion of a sloping shore during periods of high lake levels. Improved designs to replace existing structures of this nature have been promoted in the Hamilton Harbor Remedial Action Plan process (Hamilton Region Conservation Authority 1995). The typical vertical retaining wall is replaced by armorstone, with aggressive-rooting tree species planted above them. Offshore stone and anchored tree roots reduce incident energy. Sloping stone revetments along the shore are replaced by two low revetments, one offshore and one at the toe of the bluff. Wetland and aquatic plants are planted between them, and the shore is stabilized with native trees and shrubs (Hamilton Region Conservation Authority 1995).

Groins are low walls constructed perpendicular to the coast, anchored on shore, and commonly extending to the limit of breaking waves. They are used to intercept longshore and beach drift. Consequently, they create shoreline updrift of the structure, but an erosional shadow zone occurs downdrift. After the updrift area fills, sediment is supposed to spill around the lakeward tip of the groin. However, rip currents associated with storms often transport sediments offshore into water depths where they are removed from the nearshore system. The result is an extension of the area subject to net erosion. Groins have been used in many locations, including extensive sections of the Lake Michigan shore, in an attempt to stabilize the shoreline; however, the results have not been studied widely. Because designs, sites, and sediment supplies differ greatly, some groins may be drowned by sand and others flanked by water. Often, groins create conditions that are worse than those they were designed to improve (Silvester and Hsu 1991).

Breakwaters and segmented breakwaters are structures placed parallel to the shoreline at depths of 3 to 5 m, with gaps between them. These structures are used to protect harbors in Chicago, Illinois and Milwaukee, Wisconsin in Lake Michigan (Richardson 1995). As with breakwalls, scouring occurs on the lakeward side of the structures, but sediments accumulate in the low energy area behind them. However, during storm events, waves enter the gaps, refract around, or overtop the structures. As a result, some of the sediment is flushed out of the area landward of the breakwater, some moves along the shore, but some is also transported offshore. In regions with a limited sediment supply, and hence a reason for action, the end result is a reduction in the beach (Silvester and Hsu 1991).

Causes for failure of the above-mentioned structures reflect the underlying problem: lack of a sediment supply adequate to maintain a natural shoreline through periods of high and low lake levels and through storm events. Another approach for shoreline restoration recognizes the diminished sediment supply and seeks to supplement it by beach nourishment. Since 1955, over 400 individual nourishment efforts have taken place at 60 sites along the Great Lakes shoreline (O'Brien et al. 1999). However, application of sand to a beach seldom results in a natural slope, and the wave climate quickly modifies it, resulting in an initial loss of as much as 30 to 50% of the sand. Continued erosion may result in a loss of 80 to 90% of the beach width after 15 to 24 months. The use of coarser material to reduce transport by waves has also not proven successful (Silvester and Hsu 1991).

Yet another approach to restoring and maintaining a protective shoreline shows promise, however. Headland control makes use of a naturally occurring landform in which crenulate- or J-shaped bays are formed between headlands. These bays are found in many sizes along straight, bayed, and convex shorelines of many types of waterbodies. The shape of the bays keeps them in equilibrium. Energy inputs recycle sediments within them because constructive waves arrive nearly normal to the beach and movement of sediment lacks a longshore component. Thus, any eroded sediments remain within the compartment and are returned to the beach during low energy periods (Silvester and Hsu 1991). On a straight shoreline facing erosion pressure, such as a lengthy barrier beach protecting a wetland embayment in a region with reduced sediment supply (e.g., Sheldon's Marsh, Lake Erie near Sandusky, Ohio), a series of headland structures initiated at the downcoast end could stabilize the existing sand and slow or halt net erosion by eliminating the longshore component of sediment transport (Figure 2a). On a convex sand spit (e.g., Presque Isle, Lake Erie near Erie, Pennsylvania), a series of headland structures could be initiated at the downcoast end and placed at different angles and spacing to adapt to the refracted direction of wave orthogonals (Figure 2b). Construction of headlands can include several designs and angles of orientation. Silvester and Hsu (1991) recommended a structure with a lakeward curve at the downcoast tip (Figure 2b) that will accrete a beach along its full length. During erosion events, this beach would form a wave-dissipating offshore bar and thus reduce the required size of the structure. In addition, slowed diffraction around the curved surface would minimize scouring. Headland control has been used successfully along the ocean coast in different parts of the world. It has also been used successfully on Lake Ontario by the Toronto Harbour Commission to assist in land reclamation for recreational purposes and harbor development (Denney and Fricbergs 1979).

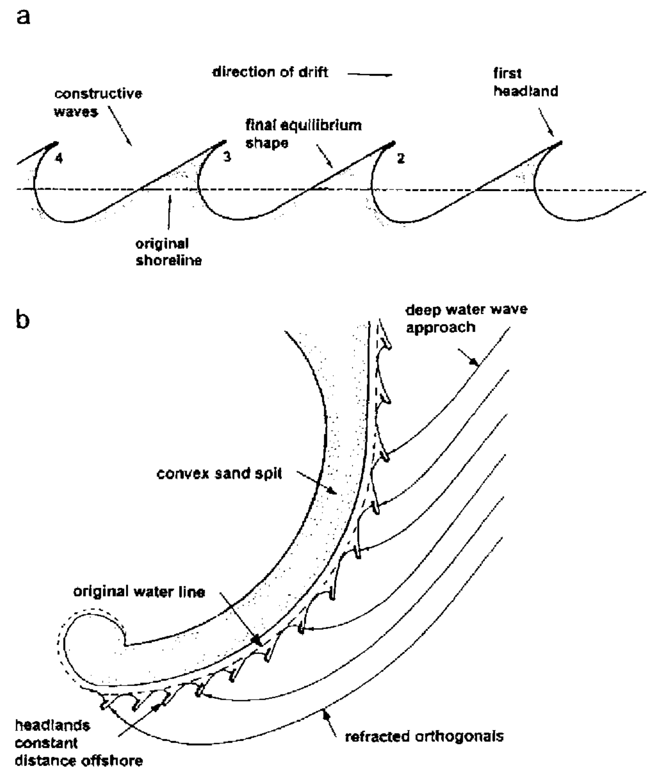

Figure 2. Diagram showing use of headland control for accretion of beach a) along a straight shoreline, b) on a recurved sand spit (modified from Silvester and Hsu 1991).

Chemical Remediation

Water and sediment quality can be restored by reducing or eliminating inputs of nutrients and contaminants through better management practices, better technology, construction of new treatment facilities, changes in the discharge permitting process, and locating and eliminating illegal discharges. Existing contaminated sites could potentially be restored using a variety of sediment remediation technologies.

Better management practices can be elevated to the watershed level to reduce siltation and inputs of nutrients and pesticides from agricultural runoff, as well as upstream loading from municipal and storm sewers and from roads and other developed lands. Specific practices could include livestock fencing, tree planting, erosion control, bank stabilization, buffer strips, reforestation, and rerouting of surface drainage systems and discharges away from wetlands.

Modern technological advancements may reduce future contaminant loading into wetlands, but many wetland sediments are already repositories for contaminants discharged in the past. Pesticides, PCBs, chlorinated organic compounds, PAHs, and other industrial contaminants selectively partition onto sediments, which become a sink. These sediments may be left buried, flooded, and out of biological contact, and in certain cases, natural remediation processes may occur. These include biodegradation, chemical degradation, and advection and transport of sediments. Clean sediments may also be deposited over the contaminated sediments to diminish risks associated with the sites (USEPA 1994a, Passino-Reader et al. 1999).

Active sediment remediation may be an alternative in some cases. Non-removal remediation technologies are designed either to isolate the sediments from the surrounding environment by capping or containment or to treat the contaminants *in situ* by immobilization, chemical, or biological processes. Capping and containment technologies have not been applied to wetlands (M. Landin, pers. comm.), nor do they result in reduction of contaminant load or sediment toxicity. Implementation of such technologies could also have considerable impact to the water column and the biota of the area (Averett et al. 1990).

In situ immobilization treatment seeks to bind contaminants in the sediments or soils using injections of cements, pozzolans, and thermoplastics. *In situ* chemical treatment by spreading aluminum sulfate (alum) over the water surface and allowing it to settle onto the sediment is used to control the release of phosphorus in well-buffered, hard-water lakes and could be applied to some wetlands. Also, increased oxygenation in hard waters causes co-precipitation of phosphorus as $Ca(PO_4)_2$, thereby increasing sedimentation and binding of soluble P (Wetzel 1975). Injection of calcium nitrate into sediments to promote oxidation of organic matter in conjunction with lime and ferric chloride additions might be used to promote denitrification and phosphorus precipitation (USEPA 1994b).

One potential *in situ* biological treatment is phytoremediation, in which plants are used to extract contaminants and are then harvested. This developing technology seems mostly directed toward upland applications, but some wetland plants may provide the necessary contaminant-uptake ability. A second potential technology for application in the Great Lakes is management of zebra mussel populations (*Driessena polymorpha* Pallas) to increase water clarity or remove contaminants by filtering large volumes of water (Reeders et al. 1993). This method would require development of means to harvest and dispose of large quantities of zebra mussels. It is also complicated by the potential negative effects of this non-native organism. Another potential biological treatment is reduction of common carp (*Cyprinus carpio* L.) populations, which has been shown to reduce phosphorus concentrations by reducing the amount of phosphorus-laden excrement (LaMarra 1975, Shapiro et al. 1982).

An *in situ* treatment not yet applied to wetlands is soil vapor extraction of volatile organic contaminants (McNicoll and Baweja 1995). Specially designed vapor-recovery wells connected to vacuum pumps withdraw contaminated vapor from the subsurface for treatment, promoting further volatilization of contaminants. Introduction of fresh, oxygenated air into the contaminated zone also promotes natural biological degradation of contaminants by increasing activity of indigenous bacteria. However, this process is likely limited in wetland use by the shallow depth of aerated soils (if aerated at all). On the other hand, water-saturated and anaerobic sediments generally result in immobilization of contaminants and reduction in biomagnification.

Removal technologies are more widely used and consist of two general types—mechanical dredges and hydraulic dredges. Material removed may be pretreated by dewatering or physical separation. Treatment technologies include thermal destruction, thermal desorption, immobilization, extraction, chemical treatment, and bioremediation (Averett et al. 1990). A wetland application reported to be successful was conducted in the intertidal zone of the Hudson River in New York State. The contaminated area was isolated by diking it off using geotextile water-filled tubes, removing the sediments, filling with clean soil, replanting, and removing the dike (M. Landin, pers. comm.)

Biological Remediation

There are three general means of involving biological organisms in restoration of wetlands: 1) control of non-target organisms, 2) enhancing populations of target organisms, and 3) enhancing habitat for target organisms through management of plant species that provide habitat, physical modification of existing habitat, or introduction of constructed habitat.

Control. Competition can be reduced by controlling species not targeted for restoration, including non-indigenous species. Weedy plants can have numerous characteristics (King 1966), but a desirable plant in one location may be considered a weed in another (Mitchell 1974). The species most targeted in the Great Lakes include *Lythrum salicaria, Phragmites australis* (Cav.) Steudel, *Typha angustifolia* L. or *T.* × *glauca, Phalaris arundinacea* L., and *Myriophyllum spicatum* L. Control methods include physical harvesting or exclusion, chemical control, and biological control. Examples of physical removal are numerous but often require the proper environmental conditions. Small populations of *Lythrum salicaria* can be pulled by hand (Thompson et al. 1987). *Phragmites australis* can be cut in August or September (Cross and Fleming 1989) and burned or disked late in the growing season (van der Toorn and Mook 1982, Cross and Fleming 1989, but see Thompson and Shay 1985, Shay et al. 1987). When soils are dry, *Typha* can be repeatedly disked (Sodja and Solberg 1993) or cut and rhizomes tilled (Wilcox and Ray 1989). In appropriate settings, *Typha* can be cut during the growing season or burned in winter if followed by flooding (Kaminski et al. 1985, Ball 1990, Sodja and Solberg 1993). It can also be removed by muskrats (*Ondatra zibethicus* L.) when populations of this herbivore are allowed to increase (Weller 1981). Heavy construction equipment has been used to mechanically remove *Phalaris arundincea,* but these efforts have generally been unsuccessful due to regrowth from seeds or rhizomes (Apfelbaum and Sams 1987). Hand-pulling and black plastic mulch can also be effective in controlling small infestations of *Phalaris* (Naglich 1994); repeated burning in late autumn or late spring for five to six years may control this grass also, but seeds of other species should be present in the seed bank for best results (Hutchison 1992). *Myriophyllum spicatum* and other submersed macrophytes with a dense growth habit can be reduced temporarily by harvesting (e.g., Engel 1990, Boylen et al. 1996), but hand harvesting and suction harvesting are preferred over mechanical harvesters because they remove the entire plant and leave fewer fragments to recolonize or spread (Madsen et al. 1988, Boylen et al. 1996). Harvesting is most effective in controlling *Myriophyllum spicatum* when combined with physical exclusion by benthic barriers (Boylen et al. 1996).

Chemical control of *Lythrum salicaria, Phragmites australis,* and *Phalaris arundinacea* in large, monospecific stands can be accomplished by aerial spraying with glyphosate according to the recommended prescription. However, this non-selective herbicide kills non-target species also; therefore, hand-spraying is recommended in mixed stands (Balogh 1986, Apfelbaum and Sams 1987, Thompson et al. 1987, Hutchison 1992, Marks et al. 1994). Other herbicides have been used on these invasive emergent species also (Apfelbaum and Sams 1987, Thompson et al. 1987, Cross and Fleming 1989), and yet others have been used on submersed aquatic species (Westerdahl and Getsinger 1988, Madsen 1997).

Insects that might serve as potential biological control agents for *Phragmites australis, Typha, Myriophyllum spicatum,* and *Lythrum salicaria* were reviewed by Galatowitsch et al. (1999). However, biological control has been tested and implemented only for *Lythrum salicaria* (Blossey 1993, Blossey et al. 1994, Hight et al. 1995). The root-boring weevil *Hylobius transversovittatus* Goeze and leaf-feeding beetles *Galerucella calmariensis* L. and *G. pusilla* Duff. have been released in the Great Lakes region. There is also evidence that biological control through competition from native plant species may reduce the dominance of *Lythrum salicaria* under some environmental conditions (Rawinski and Malecki 1984, Wilcox et al. 1988).

Control of non-indigenous vertebrates also may be desirable to effect wetland restoration. However, the mobility of many such organisms makes control difficult. Some faunal species, such as mute swans (*Cygnus olor* Gmelin), that may cause damage to wetland vegetation, aggressively drive away native fauna, or impede success of other restoration methods have social or legal status that makes attempts to control them difficult. In such cases, restoration sites might be protected by using exclosures; similar means may also be necessary for certain native species, such as resident flocks of Canada geese (*Branta canadensis* L.).

The most common example of an attempt to control non-indigenous fauna in Great Lakes wetlands is use of dikes, fences, and grates to restrict access of large common carp, which have been shown to reduce vegetative cover in diked wetlands as a result of spawning and foraging behavior (King and Hunt 1967). Increased total phosphorus, total ammonia, and turbidity were also associated with carp in enclosures in the Cootes Paradise wetland of Lake Ontario, although other factors also contributed to turbidity (Lougheed et al. 1998). Studies in diked marshes adjacent to Lake Erie by Navarro and Johnson (1992) suggested that stocking of northern pike (*Esox lucius* L.) could be used to control common carp populations, as pike preyed selectively on carp ranging from 55 to 174 mm TL. With further study, this is an option worthy of consideration in appropriate locations, although it may result in predation on amphibians and other fish species also. Control of common carp is discussed further in the case studies later in this paper, but another alternative that could provide benefit in some locations is sponsorship of a no-release carp-fishing derby. Other non-indigenous fish species that may have deleterious effects in wetlands include ruffe (*Gymnocephalus cernuus* L.) and round gobies (*Neogobius melanostomus* Pallas), which compete with native fish species for food and also prey on the eggs of native fish. Control methods applicable to wetlands have not been developed for those species.

Zebra mussels can colonize macrophytes, weigh them down, and cause collapse of the plant (D. Garton, pers. comm.); filter-feeding activity increases water clarity and the abundance of submersed vegetation (Skubinna et al. 1995) and may also greatly reduce the number of planktonic organisms in a wetland and shift productivity toward benthic organisms (Fahnenstiel et al. 1995). In addition, zebra mussels selectively attach to native clams, often resulting in death of the clam as a result of competition for food and reduced mobility (Schloesser et al. 1996). Colonies of zebra mussels alter the profile and character of other hard substrates, and mussels create new habitats not naturally present in a wetland. Again, control methods are not available for zebra mussels in wetlands; however, populations in wetlands seem to be held in check naturally by warm waters in summer, ice and water-level decreases in winter, and drawdowns associated with frequent seiche action (Brady et al. 1995, S. J. Nichols, pers. comm.). Impacts on native clams can be reduced by labor-intensive cleaning of shells by hand (Schloesser 1996, Nichols et al. 1999) and also seem to be reduced naturally in Great Lakes wetlands because warm waters, coupled with soft substrates, induce the clams to burrow. Abrasion during burrowing and low dissolved oxygen concentrations in the wetland sediments restrict the ability of zebra mussels to remain attached to clams (Nichols and Wilcox 1997).

Population Enhancement. Although natural recolonization of wetland vegetation is common in Great Lakes wetlands as a result of the natural pattern of water-level fluctuations, and fauna often respond to this rejuvenation of habitat, management needs may dictate enhancing populations of some organisms more frequently. Numbers of specific target organisms can be increased by direct stocking, seeding, or transplanting. Organisms used as prey might also be stocked to enhance survival of target species. However, the source and genetic stock of any introduced organism and the potential for inclusion of undesirable non-target organisms within the supply should be considered before taking action. Success and desirability of a proposed introduction may also relate to past history of an organism at the proposed site; thus, an historical analysis may be worthwhile.

Wetland plant materials in the form of seeds, tubers, rhizomes, seedlings, and mature plants are now widely available from commercial sources. Alternatively, these materials may be obtained from local wild sources with proper authorization and care to avoid damage at the donor site. When available, donor soil from another wetland that has been destroyed can be used as a source of seeds and propagules. In large restoration projects backed by sufficient physical and monetary resources, cultivation of transplant materials is also an option. The Royal Botanical Garden in Hamilton, Ontario has adopted this strategy to supply 20 species for transplanting in the Cootes Paradise Restoration Project (L. Simser, pers. comm.). Specific planting and transplanting recommendations vary by species and are available from suppliers and other sources (e.g., Schnick et al. 1982, Thunhorst 1993). Two selected examples, one a perennial submersed species, wild celery *(Vallisneria americana* Michx.), and the other an annual emergent species, wild rice (*Zizania aquatica* L., *Zizania palustris* L.), are commonly transplanted or seeded in standing water in inland wetlands and may be suitable for some Great Lakes wetland sites.

In the spring, winter buds of wild celery, fresh and ideally from local sources, are often placed in gravel-weighted mesh bags and released through the water column to the sediments in locations with required substrate, water depth, and light availability (Korschgen and Green 1988). This method has been applied successfully in the Great Lakes (Lowe 1988). Transplant sites should have some protection from wave action, a firm silt or sand-silt substrate, water depths of 0.9 to 1.2 m, available light well within the photic zone, and a slow current (Korschgen and Green 1988). General water chemistry characteristics at most Great Lakes sites should be suitable. *Vallisneria* can grow in association with other submersed plant species but is sensitive to light reductions caused by dense canopies (Titus and Stephens 1983).

Seeds of the annual grass, wild rice, are harvested in late summer and sowed directly into the sediments through the water column in appropriate sites without drying or after-ripening. Seeds from a regional source or at least a similar latitude should be selected to ensure compatible phenology. Seeding sites should be protected from wave attack, lack significant competition from other plant species, and have relatively soft but not flocculent sediments and water depths of 30 to 120 cm (J. Meeker, pers. comm.). Because wild rice begins growth in a submersed stage, water should not be turbid; yet, mesotrophic waters and proximity to flowing water best serve its large nutrient requirement (Meeker 1996). Multiple years of seeding may be required to develop a self-sustaining population. Wild rice has also been transplanted by dragging a rake through soft sediments to dislodge plants, transporting them in large containers of water, and weighting the bottom or root end before placing in soft sediments (Grillmayer 1995).

There is a long history of stocking fish in many types of water bodies but with little attention to wetlands. Grimm and Backx (1990) suggested that management of northern pike populations could be used as part of a program to restore certain shallow, eutrophic lakes. Goeman and Spencer (1992) reported little success in biomanipulation of northern pike populations in Minnesota lakes. However, stocking of 2–6 cm juvenile northern pike in May–June in a small, shallow, eutrophic lake in Denmark was moderately successful and resulted in a reduction in planktivorous fish, an increase in zooplankton, a decrease in phytoplankton, and an increase in water transparency through trophic cascading (Berg et al. 1997, Sondergaard et al. 1997). The effect was not long-lasting without continued stocking and was considered likely to be most successful in lakes with permanent macrophyte cover. An alternative means of enhancing northern pike populations is to manage spawning marshes. The outlet to a shallow wetland area with grassy vegetation suitable for spawning is controlled to maintain water levels for rearing of fingerlings. Newly hatched fry can be introduced; adult brood stock can be introduced; or more commonly, adult northern pike can be allowed direct access prior to closing the outlet (see review in Barry and Machowski 1996).

Another fish species common to Great Lakes wetlands with a history of stocking in other water bodies is muskellunge (*Esox masquinongy* Mitchill). However, the cost of propogation and the mortality rate (Hanson and Margenau 1992) of released fingerlings may make this a questionable operation (Bennett 1970). If practiced, fish should be released into waters with water temperatures similar to rearing ponds (Mather and Wahl 1989) and in areas near aquatic vegetation and after dark to reduce stress (Belusz 1978). Margenau (1992) found that spring yearlings were more cost-effective to stock than fall fingerlings.

Habitat Enhancement. Physical habitats provided by biological organisms can be altered to benefit target species by planting or culturing assemblages of plant species that stabilize the habitat or provide favorable cover, food, and nesting or spawning areas. Existing habitat may be physically modified, or constructed habitat targeted for specific taxa can be introduced.

Plant Management. In addition to their value as a direct or indirect source of food to faunal organisms (Carpenter and Lodge 1986), wetland and aquatic plants form the structural basis for habitat used by many species of aquatic fauna. Differences in the density and diversity of plant species and structural types in wetlands result in different values as protective cover and reproductive habitat (Engel 1985, Wilcox and Meeker 1992). Therefore, managing plant communities can have direct effects on faunal species of interest in wetland restoration; management options include reestablishment of assemblages of plants and manipulation of existing plant communities.

Selection of a group of plant species for reestablishment should be based on plant species characteristics (past history at site, growth requirements, environmental tolerances, growth form, growth rate and life span, reproductive requirements, competitive ability, hardiness to climate and disease, value as wildlife food or cover) as related to site characteristics (water depth, wave and ice exposure, water quality, soil type, size and configuration of site, climate) and placed in the context of practical issues such as availability, cost, ease of propagation, and maintenance requirements (Schnick et al. 1982).

Dushenko et al. (1990) conducted experiments on reestablishment of a suite of submersed macrophytes in the Bay of Quinte, Lake Ontario using trays of transplanted plants and turions. They reported general success at sites with adequate protection from wave attack, adequate light availability, and protection from common carp activity. Although they reported no problems with herbivory, the potential for damage from muskrats and resident populations of Canada geese must be considered, especially in attempts to reestablish emergent plants. Emergent wetland habitat was introduced into Collingwood Harbour, Ontario (Lake Huron) by constructing wetland "pods" from welded re-bar and enclosing them with galvanized chicken wire to protect them from common carp activity. The pods were placed in water depths ranging from 0.5 to 1 m and planted with *Typha latifolia* L., *Scirpus validus* Vahl, *Pontederia cordata* L., *Sagittaria latifolia* Willd., and other species (Grillmayer 1995). This concept originated and was first implemented in Hamilton Harbour, Ontario (Lake Ontario). Triangular cedar cribs were constructed and installed within snow-fence exclosures to protect them from common carp and Canada geese in an embayment near the mouth of Mimico Creek, Ontario (Lake Ontario). Transplants of *Typha* sp., *Sagittaria latifolia,* and *Scirpus validus* proved successful in establishing wetland habitat at that site (Vincent 1995a). Other methods available for establishing wetland habitat include prevegetated, biodegradable plant carpets containing rooted plants, pregrown plant pallets, pregrown floating islands, and other bioengineered means of introducing assemblages of wetlands plants (e.g., Bestmann Green Systems, Salem, Massachusetts, USA).

Habitat provided by plant commuities can be manipulated by management of muskrat and beaver (*Castor canadensis* Kuhl) populations. Muskrat trapping in a wetland can reduce populations, resulting in more extensive growth of emergent plants such as *Typha.* Closure of a wetland to trapping can result in creation of open pools of various sizes, especially around lodges, and sometimes complete denudation (Weller 1981). The pools may remain for several years if cutting of *Typha* is accompanied by increases in water level. Pools create more wetland edge and may result in hemi-marsh conditions (Weller and Spatcher 1965) if muskrat activity is managed at the correct level. Changes in the nature and extent of emergent vegetation resulting from muskrat activity can have major implications on the rest of the wetland food web (Weller 1981). Beaver are far less common in Great Lakes wetlands, but their actions in cutting woody vegetation and altering hydrology can affect habitat for other species also. Management of beaver populations through regulation of trapping can be used to limit or enhance those effects. Additional means of manipulating plant communities include burning, cutting, flooding, tilling, disking, harvesting, spraying with herbicides, installing benthic barriers, and introducing biological control agents; these methods were discussed briefly in the previous section on control of non-target organisms.

Physical Modification. Habitats for both plants and animals can also be modified by physical means–often achieved with the use of heavy equipment or addition of constructed habitat. General examples include increasing habitat structure or roughness by arranging existing logs, sediment spoil, boulders, and gravel or rock in nearshore areas. There are numerous examples of other modifications of existing wetland habitat that have been implemented in the Great Lakes. Channels 3-m wide and 0.75- to 1-m deep were dredged through dense, monospecific cattail stands in the Bay of Quinte, Lake Ontario using a floating backhoe to create spawning and nursery habitat for northern pike and other species (Mathers and Hartley 1995). Dredge spoils piled on alternating sides of the channels provided nesting habitat for black terns (*Chlidonias niger* L.) and pie-billed grebes (*Podilymbus podiceps* L.) while avoiding creation of travel routes for land-based predators (McHattie et al. 1995).

Spawning habitat for northern pike was also created in the Toronto Islands lagoon of Lake Ontario by constructing 3-m wide channels of varying shallow depths in sand using a bulldozer and backhoe, followed by addition of compost and planting of vegetation (Vincent 1995b). A northern pike spawning marsh was created near the mouth of Grindstone Creek at Cootes Paradise, Lake Ontario by using a stop-log structure and low earthen berm to maintain sufficient water depth during the spring to ensure survival of young-of-the-year northern pike. Narrow channels were also excavated at the site (Royal Botanical Garden 1997). A similar project was implemented in Collingwood Harbour of Lake Huron (Grillmayer 1995).

Shallow embayments were excavated in deltaic lands at the mouth of the McKellar River, Ontario on Lake Superior with hydrologic connection to the river and then planted with a variety of emergent, floating, and submersed plants to provide wetland habitat for a variety of fish and wildlife (Bray 1995, Lee 1995). An island was created at the mouth of McVicar Creek, Ontario (Lake Superior) to provide protection from wave action and enhance redevelopment of an historic wetland (Geiling 1995). Small islands were also created near shore in Hamilton Harbour on Lake Ontario to create habitat for colonial waterbirds (McHattie et al. 1995).

Management of zebra mussels can be used not only to alter water quality but to change the physical nature of substrates. When zebra mussels are present in large numbers, a hard surface with many recesses can replace natural substrates. Removal of zebra mussels can reverse this alteration.

If modification of existing habitat is not reasonable or possible, habitat targeting specific groups of organisms can be constructed or placed in a wetland. Habitat requirements and design criteria for habitat creation for a number of wetland herpetofauna, birds, and small mammals are described by McHattie et al. (1995). They include addition of exposed boulders, stumps, and floating logs to provide basking areas for turtles and snakes, calling platforms for male frogs, and loafing platforms for birds and installation of root wads and islands to provide nesting sites for birds and turtles, as well as to deflect water flow and reduce wave action.

Several methods were implemented in one project at a Toronto-area park on Lake Ontario. Spawning habitat for smallmouth bass (*Micropteris dolomieui* Lacepede) was created by establishing pea gravel/sand shoals adjacent to emergent and floating vegetation in protected areas (Vincent 1995c); tree crowns and logs were placed at varying depths to increase habitat for periphyton, invertebrates, fish, and other biota (Strus 1995); six elevation zones were created by grading to provide different types of habitat during different water-level conditions (McHattie et al. 1995); experimental mudflats to be kept free of vegetation were created for shorebird habitat (McHattie et al. 1995); and a snake hibernaculum was installed (McHattie et al. 1995).

Yet another example of constructed habitat is a floating reefraft to provide nesting habitat for common terns (*Sterna hirundo* L.) and shelter for fish (Blokpoel and Jarvie 1995). These reefrafts consist of an achored, floating wooden platform covered with sand and gravel for nesting, attached ramp to water level to allow access by tern chicks, attached floating logs for loafing, and dark green plastic snowfence suspended from the bottom to provide fish habitat (Blokpoel and Jarvie 1995, Jarvie and Blokpoel 1996).

ALTERNATIVE APPROACHES—CASE STUDIES

Metzger Marsh

Metzger Marsh is a 300-ha wetland in an embayment in western Lake Erie, approximately 18 km east of Toledo, Ohio, USA (Figure 3). It lies within the boundaries of refuges managed by the U.S. Fish and Wildlife Service (USFWS) and the Ohio Division of Wildlife (ODW). Although early attempts to dike, drain, and farm portions of the wetland failed, the site has a history of human disturbance. It once formed the mouth of Cedar Creek, which was channelized directly to the lake in the late 1800s (now without hydrologic connection to the wetland). The embayment was also formerly protected from waves on the lake by a barrier beach. Aerial photographs from 1940 show an intact barrier beach, with 58% of the embayment vegetated. Erosion during ensuing periods of high lake levels re-

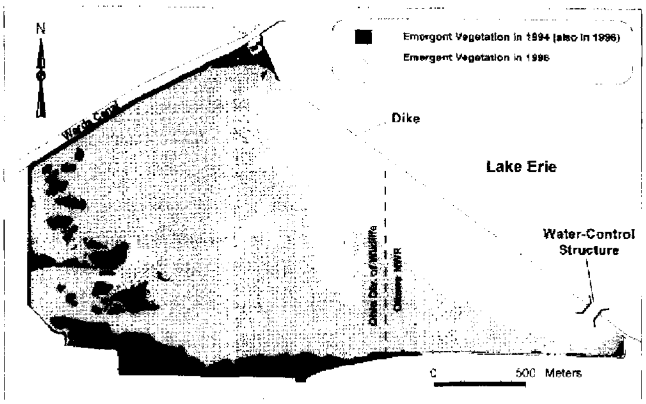

Figure 3. Map of Metzger Marsh wetland restoration site, Lucas County, Ohio, USA on southwest shore of Lake Erie showing extent of emergent vegetation in 1994 before restoration and in 1996 after first year drawdown.

duced the extent of the barrier beach until it was completely lost during the extremely high levels of 1973. Progressive loss of vegetated area accompanied erosion of the protective barrier, with 19% of the wetland vegetated in 1973 and 10% vegetated in 1993 (Kowalski and Wilcox 1999). Paleoecological studies (Jackson and Singer 1995) suggest that pre-European-settlement vegetation was dominated by sedges and grasses; however, in recent decades, open water area increased, and vegetation was largely restricted to islands of cattails and common reed.

Interest in restoring Metzger Marsh grew from managers' desires to provide better wetland habitat in the near term and recognition of the limits of natural restorative processes in a highly disturbed environment. Extensive armoring of the U.S. shoreline of western Lake Erie to protect human property resulted in enough loss of sediments from the littoral drift of the lake that the barrier beach would likely never return (S. Mackey, pers. comm.). Lake Erie had also been in an extended period of high water levels, with no intervening low levels to expose sediments and allow revegetation from the seed bank. The potential for natural revegetation of the wetland thus hinged on waiting for water levels to drop as much as a meter for at least one growing season and the unlikely reappearance of a barrier beach to provide protection. Instead, the management agencies opted for an active restoration program, with the initial intention of long-term operation of the site as a diked wetland. However, USFWS guidelines for taking an ecosystem approach to restoration and management, coupled with the required environmental assessment process for a federal action, resulted in a philosophical change in the program. The restoration effort would attempt to return the embayment to a vegetated, hydrologically connected, coastal

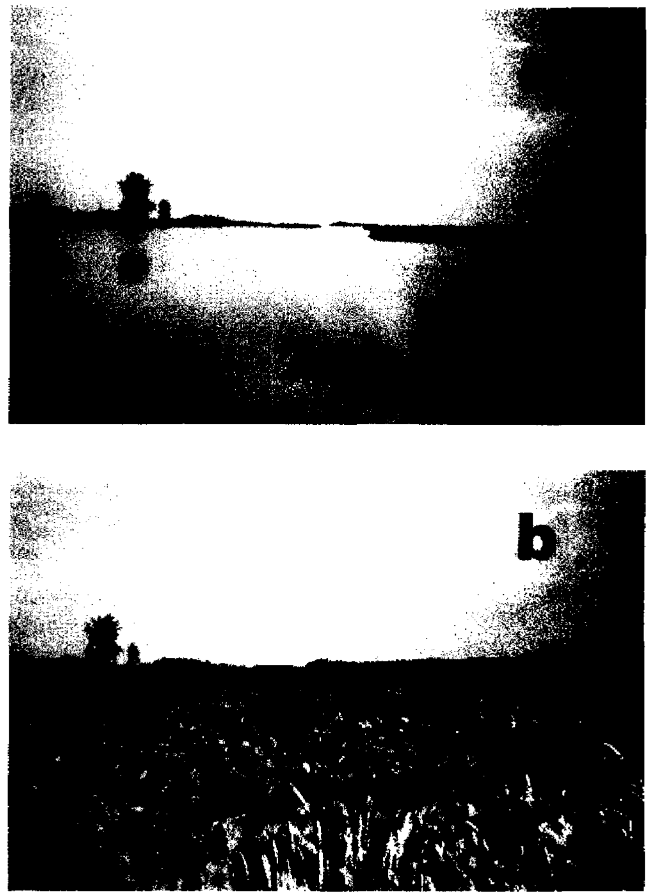

Figure 4. Photographs of southwestern portion of Metzger Marsh wetland restoration site a) 1994 before restoration, b) 1996 after first year drawdown.

marsh that provided multiple wetland functions and values.

The restoration program incorporated a dike to mimic the protective function of the lost barrier beach but included a water-control structure in the dike that could be opened following restoration to allow hydrologic connection with the lake similar to the original wetland. Dike construction was completed in 1995, and the control structure was installed in 1996. The control structure remained closed in 1996 and 1997 to allow a drawdown of water levels to mimic a low lake-level period. Rotenone was also applied by ODW at the lowest water stage to kill any remaining trapped common carp. The seed bank produced a quick response, with 73% of the wetland revegetated in 1996 (Figures 3 and 4), 82% revegetated in 1997, and 72% revegetated in 1998 when water levels were increased. Prominent recolonizing taxa included *Cyperus odoratus* L., *Polygonum lapathifolium* L., *Echinochloa crus-galli* (L.) Beauv., *Leersia oryzoides* (L.) Sw., *Bidens cernua* L., *Typha angustifolia*, *Scirpus validus*, and *Sagittaria latifolia* but also considerable *Phragmites australis*, some *Lythrum salicaria*, and a number of upland species such as *Abutilon theophrasti* Medikus. The timing of exposure of mudflats in 1996 also allowed airborne seeds of *Populus deltoides* Marshall, *Salix cordata* Michx., *S. exigua* Nutt., and *S. fragilis* L. to germinate and grow across large areas of the wetland. The invasion of trees prompted experimental management actions by ODW. Portions of the tree-dominated areas were cut during the drawdown in 1997, and portions were sprayed with 2, 4-D in 1998 and 1999. Some *Lythrum salicaria* was sprayed with glyphosate in 1997. On USFWS property, an experimental planting of *Vallisneria americana* tubers was conducted in 1997 in water approximately 20 to 50 cm deep, and exclosures made by stringing mylar tape between metal posts were tested in water <10 cm deep where herbivory on *Typha* seedlings by Canada geese was observed.

The wetland was reflooded in 1998 without lake connection, and the control structure was opened in March 1999. The water-control structure was designed with five 2-m-wide channels that can be closed individually. The size of the openings was based on calculations of the potential flow rates between the lake and the wetland that could be driven by seiches on the lake, given the volume of the wetland, the hydrologic head created by the seiche, and the period of the seiche. Seiches with an amplitude of about 0.2 m and period of about 14 hr occur regularly, with larger seiches of about 1.5-m amplitude and 18- to 24-hr period occurring occasionally. Sudden, storm-driven seiches have also resulted in water-level changes of about 0.75 m in 4 to 5 hours. Sizing of the opening was intended to prevent major dewatering and flooding within the wetland that could affect fish-spawning or bird-nesting areas. In reality, the water-control structure mediates the seiche effect just as the natural opening in the barrier beach once did. The 5-channel design provides an option for management changes when engineering plans are tested by actual conditions.

The water-control structure also contains an experimental fish-control system that allows direct wetland access by most small fish, yet restricts access by large common carp while allowing passage of other large fish. Each of the three central channels is spanned by a grate containing vertical bars spaced 5 cm apart (with cross-bars for stability). The design was based on studies at Cootes Paradise marsh on Lake Ontario indicating that 95% of the common carp seeking to enter the wetland would be excluded (V. Cairns, pers. comm.). The two outer channels contain fish passageways, one for fish moving into the marsh and one for fish moving to the lake. The entry side of each passageway can be fitted with experimental grates of sizes and shapes that allow larger fish, such as northern pike to pass through, while minimizing the numbers of common carp that gain entry, based on differences in body size and shape (French et al. 1999). The exit side of the passageway for entering the wetland is fitted with 5-cm vertical grates to prevent common carp from proceeding into the wetland. The passageways each contain fish baskets that can be lifted with an electric hoist. The entry fish basket is operated daily during open water seasons to capture, count, and measure fish and selectively move all but common carp into the wetland. The exit fish passageway is operated as necessary to monitor and move large fish out of the wetland. The ability to change grates at both ends of the passageways also provides opportunities to study other sizes and shapes of fish that move in and out of the wetland.

Pre-restoration studies were completed to measure physical attributes (such as bathymetry) and to characterize the wetland plant communities and most major groups of fish and wildlife (fish, juvenile fish, shorebirds, waterfowl, small mammals, herpetofauna, invertebrates). Limited studies continued during the drawdown and initial reflooding years. All studies are scheduled to be repeated for multiple years following hydrologic reconnection with the lake. The results will be critical in evaluating the success of the techniques employed, developing any necessary modifications, and preparing guidelines for technology transfer to other managers to assist them in opening diked wetlands, increasing wetland functions, and improving habitat for a variety of both fish and wildlife. Once completed, the Metzger Marsh restoration project is expected to serve as a model for future coastal marsh restoration projects in locations with severe sediment deficits and could guide efforts to hydrologically reconnect and manage other marshes that are currently diked.

Cootes Paradise

Cootes Paradise is a 250-ha wetland, mostly marsh, located at the west end of Lake Ontario near Hamilton, Ontario, Canada and separated from the lake proper by Hamilton Harbour (Figure 5). It is fed by a number of tributaries, particularly Spencer Creek and the small, but contaminated Chedoke Creek. Once almost wholly covered by emergent and submersed aquatic plants, the marsh lost 85% of its emergent vegetation between 1934 and 1985, leaving coverage of only about 15% of the total marsh area (Trotter and Hall 1998). Much of this disappeared after 1970. Numbers of submersed plant species decreased from 24 to 10 between 1949 and 1970. Marsh vegetation typically disappeared during the periods of high lake levels but failed to recover as would have been normal during low levels. The reasons for the lack of recovery seem

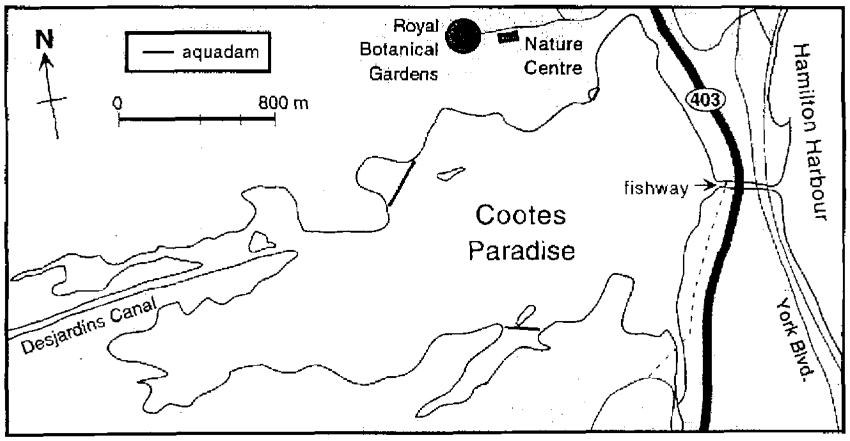

Figure 5. Map of Cootes Paradise wetland restoration site, Hamilton, Ontario, Canada at the west end of Lake Ontario.

to include manipulation of Lake Ontario's water levels, inhibition of aquatic plant growth because of physical uprooting by common carp, light-inhibiting high turbidity also related to carp activity, and the heightened input of fine urban sediments. (Painter et al. 1989, Whillans 1996). However, there were numerous other confounding factors, such as other exotic species, nutrient loading, fetch, waterfowl grazing, and algal blooms.

Cootes Paradise is owned primarily by the Royal Botanical Gardens. Since 1986, the marsh and adjacent harbor have been managed through the multi-stakeholder, decision-making process of a Remedial Action Plan (RAP) that was stimulated by the International Joint Commission but is locally directed and managed.

The restoration of Cootes Paradise involved the application of several key techniques: 1) installation of a barrier to adult common carp, 2) reduction of inflowing, watershed-derived sediments and nutrients, 3) naturalization of shoreline, 4) propagule-bank stimulation and protection, and 5) strategic planting of vegetation. The need for these techniques was not immediately clear at the outset of the restoration project. In 1988–89, a diking solution was proposed for the marsh. The expert workshop that was convened to consider this option concluded that such an intervention was premature. Rather, investigations into the efficacy and feasibility of excluding common carp and strategically planting vegetation were recommended.

Experimental use of aquadams included a small pilot project across a small bay in the northeast section of the marsh for the period of one year and another 585 m in length mid-way along the north shore that was functional at times in 1993 (V. Cairns pers. comm.). The aquadam is described by Bowen (1998). It was approximately 2-m high by 3-m wide and made of geotextile over two parallel polyethylene water-filled tubes. The double tubes prevent rolling. Installation labor required four weeks of preparing a flat, smooth marsh substrate and a road for vehicle access to the site and two weeks of intense assembling. Installed in about 0.8 m water depth, the dam allowed pumping to lower water levels over 11 ha of marsh. Natural germination of vegetative propagules occurred immediately throughout the site. However, the dam was cut in an act of vandalism, and subsequent reflooding killed the vegetation. A repeat experiment in 1994 failed similarly. This project did not operate long enough to allow dewatering to verify the effects of the high density of common carp (1500 kg/ha) nor the resilience and diversity of the propagule bank (V. Cairns, pers. com.).

Beginning in 1991, various experimental exclosures were constructed to evaluate the relation of common carp and turbidity, some in conjunction with plantings or transplants. The studies verified that these two factors, at least in combination, limited the growth of submersed vegetation (Sager et al. 1998). Exclosure of common carp in 1988 in Mercer's Glen, a nearby marsh, resulted in immediate recovery of submersed and emergent vegetation. Common carp were excluded by a weir at a marsh near the mouth of nearby Grindstone Creek in 1994. The cover of aquatic vegetation increased from around 20% to 80% over one year (Bowen 1998). The conclusion, based on the accumulated understanding from exclosure experiments was that for submersed plants, a reduction of turbidity in combination with exclusion of common carp would enable revegetation. Based on these findings and telemetry to verify common carp movements, the decision was made to construct a barrier to common carp across the narrow connection of the former Desjardins Canal between Cootes Paradise and Hamilton Harbour (Whillans 1996, Bowen and Theijsmeijer 1998).

The above-mentioned turbidity exclosures and sediment- and water-quality models resulted in the realization that perhaps half of the fine sediments flowing into the marsh were contributed from urban construction and much of the rest because of stormwater inflowing from storm sewers. Related nutrient budgets implicated point sources of municipal/industrial sewage and diffuse storm sewage. As a result, two strategies were adopted: 1) diversion of sewage outfalls away from the marsh and 2) creation of very large underground combined sewer outflow storage tanks to intercept inflowing contaminated water. The first of the tanks holds 74.25 million L (Trotter and Hall 1998). The turbidity exclosures, placed in a variety of locations, doubled as tests of planting techniques.

The foundation of the Cootes Paradise restoration has been the common carp barrier and fishway. Its main purpose is to prevent the spring spawning migration of adult common carp from Hamilton Harbour and the lake from reaching the marsh. Most common carp leave the marsh in the fall through the narrow passageway between the marsh and Hamilton Harbour. The fishway was installed there to exclude mature common carp (larger than 5-cm width and about 30-cm length). Large individuals of other fish species are lifted into and out of the marsh in eight large baskets, each with a water tank in the bottom. Common carp are released on the lakeward side of the structure; other fish are transferred across the barrier in the direction that they were moving. During 1997, the first full year of operation, 25,379 large fish were handled, representing 25 species (Bowen and Theijsmeijer 1998). About 82% of those attempting to enter the marsh were common carp, goldfish (*Carassius auratus* L.), or common carp-goldfish hybrids. Fin-clipping enabled the estimation that 97,000 common carp attempted entry. By 1998, the number of common carp handled had dropped to 57% of the previous year. Within the marsh in 1998, a young-of-the-year fish index survey of all species yielded 3,167 fish, compared with the average of 1,180 fish in the years 1994–96 (V. Cairns, pers. comm.).

The goal of the fishway was a density of adult common carp of less than 40/ha or 6,000 in total. It is estimated that the population achieved was 2,000–3,000 fish in 1997 (Bowen and Theijsmeijer 1998). That same summer, substantially lower levels of suspended sediment and widespread, spectacular growth of submergent vegetation occurred across Cootes Paradise. Where negligible plant densities had existed previously, densities up to 60 stems/m were recorded. The strongest responses were noted for sago pondweed (*Potamogeton pectinatus* L.), coontail (*Ceratophyllum demersum* L.), curly pondweed (*P. crispus* L.), and leafy pondweed (*P. foliosus* Raf.). Some species that had not been seen for years also returned (*Vallisneria americana, Zanichellia palustrus* L.). Large numbers of emergent plant seedlings were observed in shallow waters throughout the marsh. Giant burreed (*Sparganium eurycarpum* Engelm.) and arrowhead (*Sagittaria latifolia*) were the most abundant (Bowen 1998).

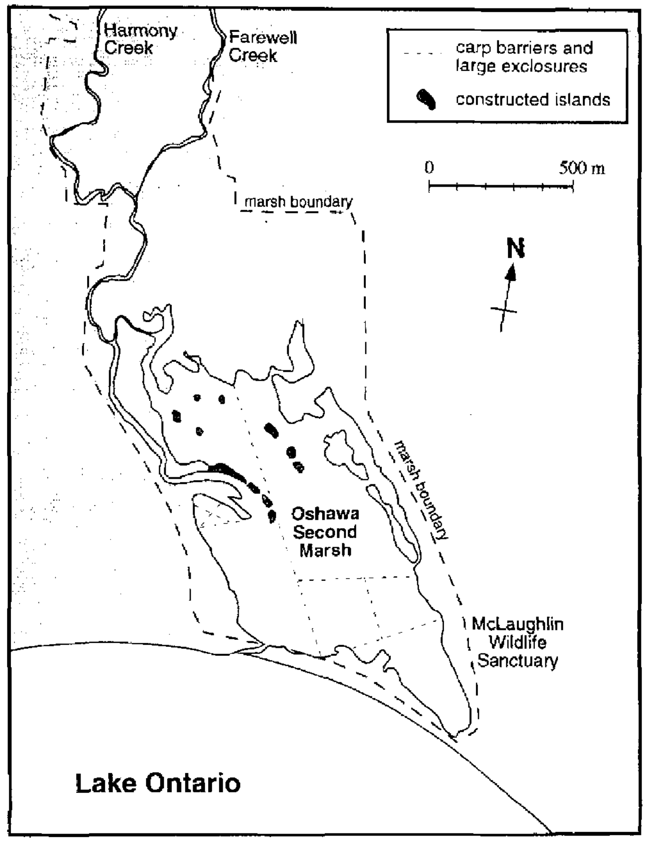

Figure 6. Map of Oshawa Second Marsh wetland restoration site, Oshawa, Ontario, Canada on the north shore of Lake Ontario.

Prior to and companion to operation of the common carp barrier, large-scale planting of emergents was undertaken through the nursery of the Royal Botanical Gardens, classroom nurseries in local schools, and volunteers. For example, in 1993–94, some 10,000 aquatic plants were grown by students in their classrooms. In total, tens of thousands of plants have been cultivated and planted in the marsh, in exclosures, unprotected, and in biodegradable coconut-fiber mats. The first phase was to establish tall emergents (e.g., *Typha, Sparganium, Scirpus*) that would shelter other species. The second phase was to increase biodiversity by planting locally uncommon species (e.g., *Asclepias incarnata* L., *Decodon verticillatus* (L.) Ell., *Iris versicolor* L.) (Bowen 1998). Accumulated experience in planting demonstrated the importance and difficulty of controlling water levels and damage by wildlife.

Oshawa Second Marsh

Oshawa Second Marsh, a 123-ha marsh with some swamp is located on the north shore of Lake Ontario within the City of Oshawa, Ontario, Canada (Figure 6). Prior to 1970, it was well-vegetated and provided habitat for a diverse faunal community (Cecile 1983). This changed in apparent response to a number of human and natural factors (City of Oshawa 1992). Effluent from a municipal sewage treatment plant emptied into the marsh. Passage through the marsh of inflowing water was made less direct by the inadvertently engineered creation of a new outlet from the marsh. This increased internal sedimentation. The formerly agricultural watershed became heavily urbanized, with attendant increases in fine sediment load and landscape disturbances. Activity of exotic common carp around stands of aquatic vegetation became more noticeable. High water levels in the early 1970s eliminated much of the emergent vegetation—not unusual in Great Lakes wetlands, except that the usual recovery during low water did not occur because lake-level regulation sustained relatively high water levels. After some issues of jurisdiction and responsibility were settled in the late 1980s, concerted efforts were made to restore and perpetuate the biological, hydrologic, and societal functions of the marsh (City of Oshawa 1992).

Implementation of the restoration plan to date has involved a variety of techniques. The restoration chronology is summarized generally by Henshaw (1996). Three principles guided the selection of techniques: 1) guidance was provided by the historic conditions, where these could be determined and where restoration was not prevented by current conditions; 2) natural resilience of the marsh was used and fostered as much as possible; and 3) foreign physical and biological materials and non-biodegradable materials were not used, except temporarily.

The first priority was to restore hydrologic function, enabled by verification of acceptable sediment contaminant levels and the predictions of a water-circulation model (*FastTABS*). The former western outlet from the marsh into Lake Ontario was reopened in the winter of 1995 using heavy equipment. Once water began flowing through the western outlet, the water course of inflowing Farewell Creek took the more direct route through it to the lake, and the eastern outlet soon filled in. The flow was aided by the earlier rechannelization of Farewell Creek through a log and debris jam immediately upstream from the marsh. This was done in February 1994 by volunteers using hand tools and light equipment. The channel of the creek through the marsh was then redefined in the winters of 1995 and 1996 using four deflector islands, sand fill, and minor back-hoe "dredging" of cattails and sediment. The channel location was determined from historic aerial photos. The deflector islands were constructed on the ice in less than one meter of water. They totalled about 11,400 m^2 in area. The islands consisted of root wads, with the roots facing the channel and trunks under the islands, backfilled with sand overlaid by geocoir mat and then topsoil. Since those interventions, the outlet has maintained its western location, although it has meandered by several meters and varied in profile, as is common for north shore Lake Ontario marshes. In February 1996, fill was added to the eastern exit to minimize the risk of a climatically driven blowout. This was stabilized additionally by natural revegetation.

The next priority was to recreate some of the in-marsh physical features, mostly islands that were noted on historic aerial photographs to have existed previously. During the winters of 1995 and 1996, eleven habitat islands ranging in area from 75 to 1,250 m^2 were constructed on the ice (Figure 6). Some of these were framed with stumps and infilled with discarded Christmas trees, over which geojute mats and soil were placed. Some floating islands were also built, using log bundles as frames and with wooden decks. They too were covered with geojute mats and then soil. Volunteer vegetation was allowed to cover the islands. Inspired by the unexpected and regionally exceptional use of one brush-filled island by nesting common terns, one of the floating islands was modified in 1996 for tern nesting and maintained free of vegetation and gulls during the non-nesting period with plastic sheeting.

During the winter of 1995, log barriers were constructed along the inside of the barrier beach and along a cattail island at the eastern margin of the marsh. These were designed as breakwaters, behind which vegetation could be planted. Small, brush-filled log cribs and root wads were also placed throughout the marsh to enhance fish habitat; the root wads seemed to function as desired.

In the winter of 1995, a fence was constructed through the middle of the marsh to keep common carp from entering the half of the marsh distant from Farewell Creek (Figure 6). It was constructed of 5-mm-mesh, chain-link fence attached to cedar poles. The poles were pounded into the sediment, and the chain link was stretched from 0.3 m above the anticipated spring maximum water level down into the sediments. Turtle ramps were placed over the fence. Common carp penetrated the fence in its first year, apparently due to improper installation and because they burrowed near high current locations. The following year, four cells were created as a fail-safe in the carp-free area using the same materials and design. Additionally, silt screens and Christmas trees were attached to the fencing of the cells to reduce the turbidity caused by wind and common carp. None of these measures were effective, except the turtle ramps. During a high water period, the common carp were able to breach the fence. They were also found to be burrowing under it, noticeably in locations of high flow. The silt screen was not strong enough to withstand the buffeting of wind and waves. Another area was partially protected by fencing and log barriers in the winter of 1996—an 800 m^2, roughly pie-shaped experimental unit divided into four wedge-shaped sub-units. Two opposite wedge cells were filled with Christmas trees; others were untreated. In 1997, all of the common carp fence was removed, except for the wedge exclosures and one cell in a location that is comparatively protected and where a planting program for emergents is planned, based in part on the experiment just described.

Vegetation was planted in a number of situations. Shoots (15-cm-long) of *Juncus canadensis* J. Gay, *Scirpus validus*, and *Sparganium eurycarpum* were planted behind the log barrier on the south shore. The shoots were obtained from local stock that had been cultivated in a commercial greenhouse. Emergent plants 7-cm long, cultivated in public school classrooms, were planted in 1996 on the inside of the barrier beach, on one of the brush habitat islands, and on the deflector island. In total, 3,210 emergent plants representing 18 species were planted in 1996. At the barrier beach, small numbers of *Sparganium eurycarpum* were the only survivors beyond one year among the 12 species planted. Canada geese, muskrats, and common carp were observed foraging in the area and were thus implicated in the mortalities, although quantitative evidence was not collected (Leadbeater 1998). Plantings were also made at the flow deflector islands, one root-wad island (7 species), in the above-mentioned four large partial exclosures (7 species), two of which contained used Christmas trees, and two small complete exclosures. Mortality was almost 100% at the islands, although occasional volunteer plants were observed. The exclosures fared much better. In the complete exclosures, *Sagittaria latifolia* and *Typha* survived and showed hardy growth. The water in the large partial exclosures was too deep for emergents, but submersed and floating vegetation thrived (*Potamogeton pectinatus, P. foliosus, Ceratophyllum demersum, Nymphaea odorata* Aiton, and *Elodea canadensis* Michx.), mainly in the exclosures with Christmas trees.

In general, the plantings did not fare well. In addition to the above-mentioned herbivorous animals, the water depth during the period of planting was prohibitively high. In fact, when the bathymetry was examined in 1997, the decision was made to plant none of the classroom-propagated plants in the main marsh because of excessive water depth. In all likelihood, a combination of the above factors has contributed to the failures. However, the Christmas tree technique merits further consideration. The technique is inexpensive and seems to function to discourage herbivorous animals, even when they could have access.

Management attention in the last two years has focused on the limits to which restoration of vegetation can occur under the influences of external factors such as high water levels, high density of pest species, and high silt input from the watershed. A watershed stewardship plan has been developed for the long-term, and diking is one of a variety of options being considered for the marsh itself.

DISCUSSION

Relation of Techniques to Ecological Principles

If restoration techniques are to succeed and results maintained for extended lengths of time, they must be founded in the basic tenets of ecology. Restoration is merely the management of ecological processes for a specific purpose, and whatever the chosen endpoint, attention to ecological theory should help in attaining the goal (MacMahon 1987).

Intermediate Disturbance Hypothesis. As demonstrated by Keddy (1983) for an Ontario lake and Wilcox and Meeker (1991) for regulated lakes in Minnesota, the underlying principle in the response of wetland vegetation to changes in water levels is that intermediate scales of disturbance (Connell 1978) maintain the greatest diversity. Proposed changes to the water-level-regulation plan for Lake Ontario should increase diversity by reestablishing a range of high and low water levels that periodically impacts broad-canopy emergent plants and elicits a response from the seed bank (Wilcox et al. 1993). Just as lack of water-level variation on Lake Ontario minimizes disturbance, ditches through wetlands of other lakes can result in extreme dewatering during natural low lake-level periods and potentially extirpate some plant species because disturbance is too great.

Hardening of the shoreline is a sedimentological remediation technique directed at minimizing shoreline erosion, but it can actually result in an extreme level of disturbance to wetlands. This disturbance may be local in the case of scouring or backstopping; however, when viewed from a landscape perspective, it can be a regional disturbance if the supply of sediments in the littoral drift is diminished. The diversity of pulse-stable wetland plant communities might better be maintained in an environment where sediment erosion and accretion both occur periodically.

Island Biogeography. Many wetlands of the Great Lakes are separated from other wetlands by lake waters or long stretches of shoreline. Their restoration needs might thus be viewed in terms of island biogeography because this isolation may limit dispersal and colonization of relatively non-motile biota. Enhancement of plant communities by seeding, transplanting, or use of donor soil may be advisable if a wetland is small in area and is far-removed from potential sources of desired colonizing species (MacArthur and Wilson 1967, Pielou 1975). However, such efforts may not be necessary in large wetlands or wetland complexes, such as the barrier beach wetlands that extend along most of the eastern shore of Lake Ontario, or in wetlands where seeds or propagules are likely to arrive through natural processes. The landscape position of a wetland should therefore be assessed before making decisions on active enhancement of plant species or relatively non-motile faunal species.

Secondary Succession. Much of the effort in a wetland restoration project involves initiation or management of secondary succession. In freshwater marshes, succession can follow a number of potential trajectories after disturbance, each with a certain probability of occurring under set circumstances. Moreover, the timing of successional stages along a trajectory depends upon a number of complicating factors that are even more difficult to pinpoint accurately. For this reason, even the initial decision to intervene is subject to question. On the scale of Great Lakes change, is a 20-yr or 30-yr lapse between loss of emergent vegetation during high water periods and recovery during low water periods within the range of natural extremes? Is intervention simply a symptom of impatience with a naturally dynamic ecosystem? The answer to both questions is a qualified yes; when viewed on a geologic time scale, 20 to 30 years is insignificant, but human lifespans and careerspans often dictate that actions take place in the near-term. The addition of invasive species to the Great Lakes may also necessitate faster action.

In the case studies presented, techniques applied in Oshawa Second Marsh were calibrated to produce conditions that would be consistent with some of those identified in the historical record. It was assumed that the conditions of the Lake Ontario nearshore ecosystem at the site had not changed fundamentally and that re-creation of a historic stage of succession would provide sufficient materials to enable the marsh to sustain that successional trajectory. Thus, intervention was at a lesser scale than in the other case studies. Landscape-scale disturbance at Metzger Marsh (shoreline armoring and loss of protective barrier beach) and Cootes Paradise (fine sediments in urban runoff coupled with other factors) suggested that the historic condition might never be attained and that secondary succession might never begin in the near-term without substantial intervention. The trajectory for secondary succession may, at best, have to mimic a natural pathway.

Invasion Windows. Succession may also be viewed in terms of the ability of organisms to invade a habitat. Johnstone (1986) proposed that the potential for invasion and incorporation of plant species into an environment is based on removal of a barrier that previously excluded the species. Some of the described time-dependent invasion windows relate to restoration approaches outlined in this paper. Stable windows are selective, non-botanical barriers to invasion that could be opened by actions such as restoring wetland hydrology or restoring protection from wave attack. These actions could result in a permanently open window that allows selected species to continue invading—a desirable, long-term result if the selected species is a desired component of the restored wetland. However, because selected species may include undesirable invasive taxa, managers should be aware of the long-term consequences of opening a stable window. Temporary windows are non-selective, botanical barriers to invasion that could be opened by eradicating undesirable vegetation during restoration activities or by purposefully introducing desired plant taxa. These changes could open an ephemeral window for invasion on a first-arrival basis by species that were previously excluded by the presence or lack of an overstory canopy. Again, managers should consider what species may take advantage of the opening. Future windows are selective, botanical barriers that do not exclude ingress of seeds or seedlings but delay their growth and entry into the plant community. As with temporary windows, the action of removing existing vegetation opens the window but, in this case, releases invaders from the seed bank that have been selected by their seed- or propagule-dispersal characteristics. The successional implications of seed banks described by van der Valk (1981) should thus be coupled with understanding of invasion windows when evaluating the relationship between ecological principles and restoration actions.

Philosophy of Mimicking Natural Processes

Magnuson et al. (1980) and Bradshaw (1984, 1987) described three options for redeveloping degraded ecosystems: 1) do nothing and allow continued degradation or slow recovery by natural processes, 2) attempt to build back exactly what was there before, or 3) replace the original ecosystem with a new alternative that may be simpler in structure. We see additional options for Great Lakes wetlands: 4) eliminate the cause of degradation and let nature prevail and 5) restore the wetland to a condition that mimics but does not replicate one that had been attained historically (a hybrid of options 2 and 3). It could be argued in favor of options 1 and 4 that natural forces will always be the most important building blocks of a restored wetland. Proponents of option 5 might argue that ecology is not fine-tuned enough to attempt more than mimicry and that option 2 is unrealistic. Mimicry of features in natural reference systems was, in fact, advocated by Zedler et al. (1997) in a study of natural and constructed tidal channels in wetlands southern California. It also could be argued that whereas all might be goals, only option 3 is practical in an ever-changing ecosystem.

In this paper, we emphasized option 5 because many Great Lakes wetlands are too disturbed to make return to historic conditions possible. In addition, our experience with the case studies showed that earlier efforts along the lines of options 1 and 4 had not produced satisfactory outcomes and "new" wetland conditions targeted by stakeholders were not novel and had historic precedents that were not always desirable. Functional mimicry was deemed more likely to retain success and do so with less continued management because it works with nature rather than against it. Successful functional mimicry may also lead to natural reversion (Gilbert and Anderson 1998), resulting in a wetland more closely resembling the original than might have been achieved by concerted efforts under option 2. On the other hand, we described some biological restoration techniques that apply to individual taxa and were not necessarily geared toward mimicry of natural systems. Cairns (1987) invoked several questions that reduce some of the options above to a species level. Should efforts be made to bring back individual rare species or communities by targeting restoration of preferred habitat to the historic condition? Conversely, should restoration target the larger ecosystem? Should new species or communities with greater chances for success be substituted for the original biota? Should non-endemic strains of organisms be substituted for strains that are not capable of survival? The answers to these questions are not simple or straightforward and likely vary in different situations.

The concept of reproducing historic ecological structure is simple in contrast to historic ecological processes. Structure is relatively easy to identify and quantify. Process is sufficiently difficult to characterize, even in the present, that reproduction of process would be practically impossible to verify. Thus, restorationists must be satisfied with ecosystemic process that at best mimics historical process, even though exact reproduction may be a goal.

Underlying the goal of mimicking historical processes is the expectation that such processes are preadapted to the climatic, morphometric, and edaphic conditions of the wetland setting. They may not, however, be adapted to the regime of anthropogenic stress that has developed over the past 200 years. Thus, with respect to the application of restoration techniques, it is necessary to monitor indicators of the acceptance of techniques by the wetland in the new stress regime. Adjustments can be made *post hoc* that are consistent with mimicry, recognizing that the historical processes remain the core of the model for restoration. For example, when common terns (previously present as non-breeders) unexpectedly nested on a constructed island in Oshawa Second Marsh, it was taken as an indicator that the marsh could assume a new function for one species of waterbird, not at the expense of, but in substitution for the loss of other historical breeders. It was also recognized that conditions on that island would be unlikely to remain suitable for the terns. Hence, a more suitable island was created nearby. In this and other cases where functional substitutions are made, the difficult issue of evaluating the success of techniques arises. How successful is a technique that targets one ecological function for one species but fulfills the same function or a different one for another species?

Measuring Success

Successful wetland restoration is often determined by a set of measures that describe how closely the restored site resembles the structure or appearance of the original or a similar undisturbed reference site. Because many restoration efforts are tied to regulatory actions, such measures of success may be dictated by the regulatory process rather than ecological principles. Most wetland restorations require considerable time to allow biological components to equilibrate with the altered or reconstructed environmental components. Thus, short-term regulatory measures of success are likely not indicative of long-term success. We agree with Ewel (1987) that measures that capture the essence of both the structure and function of a wetland are more meaningful targets for restoration efforts. We pose the five measures suggested by Ewel (1987) as a series of questions. 1) Sustainability. Is the wetland capable of perpetuating itself, or did the environment change, or was the restored community a temporary seral stage? 2) Productivity. Is net productivity of the restored site equal to the original, or have all environmental needs not been met? 3) Nutrient retention. Does the restored site retain nutrients in an equivalent manner to the original, or will it lose more nutrients and be invaded by new species that are better adapted to the new nutrient regime? 4) Invasibility. Does the restored site resist invasion by new species, or are all environmental needs not met, thus leaving niches for invaders? 5) Biotic interactions. Are the key species that link food webs and other functional processes present, or will missing links result in long-term failure to met expectations?

The traditional ecological approach to evaluating natural communities is reductionist, taking them apart and studying the pieces in order to understand the whole. Restoration, however, is synthetic, seeking to start with the pieces and building the whole (Diamond 1987, Ewel 1987). Determining success in that effort requires an understanding of not only what the whole looks like but how it functions. That challenge is the reason that restoration has been described as the acid test in determining if how much we understand about ecology (Bradshaw 1987).

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