Journal of Great Lakes Research 35 (2009) 206–214

Contents lists available at ScienceDirect

Journal of Great Lakes Research

journal homepage: www.elsevier.com/locate/jglr

Stimulating a Great Lakes coastal wetland seed bank using portable cofferdams: implications for habitat rehabilitation

Kurt P. Kowalski [a,*], Douglas A. Wilcox [a,1], Michael J. Wiley [b]

[a] U.S. Geological Survey, Great Lakes Science Center, 1451 Green Road, Ann Arbor, MI 48105, USA
[b] University of Michigan, School of Natural Resources and Environment, G504 Dana Building, 440 Church Street, Ann Arbor, MI 48109, USA

A R T I C L E I N F O

Article history:
Received 22 May 2008
Accepted 3 December 2008

Communicated by Joseph Makarewicz

Index words:
Lake Erie
Wetland
Habitat
Rehabilitation
Cofferdam
Herbivory

A B S T R A C T

Coastal wetland seed banks exposed by low lake levels or through management actions fuel the reestablishment of emergent plant assemblages (i.e., wetland habitat) critical to Great Lakes aquatic biota. This project explored the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank in a Lake Erie coastal wetland. A series of dams stretching approximately 450 m was installed temporarily to isolate hydrologically a 10-ha corner of the Crane Creek wetland complex from Lake Erie. The test area was dewatered in 2004 to mimic a low-water year, and vegetation sampling characterized the wetland seed bank response at low, middle, and high elevations in areas open to and protected from bird and mammal herbivory. The nearly two-month drawdown stimulated a rapid seed-bank-driven response by 45 plant taxa. Herbivory had little effect on plant species richness, regardless of the location along an elevation gradient. Inundation contributed to the replacement of immature emergent plant species with submersed aquatic species after the dams failed and were removed prematurely. This study revealed a number of important issues that must be considered for effective long-term implementation of portable cofferdam technology to stimulate wetland seed banks, including duration of dewatering, product size, source of clean water, replacement of damaged dams, and regular maintenance. This technology is a potentially important tool in the arsenal used by resource managers seeking to rehabilitate the functions and values of Great Lakes coastal wetland habitats.

Published by Elsevier Inc.

Introduction

There is a complex but well-established cyclical relationship between the seed bank and emergent wetland vegetation in systems with fluctuating water levels like those in the Laurentian Great Lakes. Although confounded by other factors (e.g., changing species pool, shoreline structures, wave action), the pattern of water-level fluctuation is critical to development and renewal of shoreline wetland plant communities (Keddy and Reznicek, 1985; Wilcox, 2004; Wilcox and Nichols, 2008). In fact, the extent and diversity of coastal wetlands is driven by changes in water levels (Keddy and Reznicek, 1986).

Water levels in the unregulated Great Lakes fluctuate on many scales (e.g., hourly, seasonally, annually, multiple-years). Although short-term hourly changes (i.e., seiches) and seasonal variations can affect plant community distribution (Batterson et al., 1991), it is the annual and multiple-year water-level changes that influence wetland plant communities most (Maynard and Wilcox, 1997). Shoreline and wetland plant assemblages have adapted to and thrive on cycling periods of low and high water levels. Each part of the cycle causes a moderate disturbance or stress to the ecosystem that plays a vital role in the long-term maintenance of diverse wetland plant communities.

As the water retreats during low-water periods and sediments are exposed, a number of physical and biological changes occur. Submersed aquatic and floating species are lost because there is no water to support them, but previously flooded mud flats, often containing very rich seed banks, oxygenate to some extent when exposed to air (Ponnamperuma, 1972). If water levels recede during the growing season, buried seeds in the seed bank germinate and normally reestablish a high diversity of mudflat and emergent vegetation (Harris and Marshall, 1963; van der Valk and Davis, 1978; Smith and Kadlec, 1983; Barry et al., 2004). Unless water levels rise again or the site is further disturbed by other forces (e.g., herbivory), some mudflat wetland plants are able to mature in one year and add their seeds to the seed bank. Many emergent species, however, need multiple growing seasons to mature enough to produce seeds (van der Valk and Davis, 1978). Given enough time to grow, emergent species replenish the seed bank and prepare the mud flat for the next time it is exposed after flooding. Woody plants and shrubs requiring drier conditions are able to colonize and grow at lower elevations during longer low-water periods. Over time, they often begin to dominate

* Corresponding author. Tel.: +1 734/214 9308.
E-mail addresses: kkowalski@usgs.gov (K.P. Kowalski), dwilcox@brockport.edu (D.A. Wilcox), mjwiley@umich.edu (M.J. Wiley).
[1] Present address: Department of Environmental Science and Biology, SUNY-Brockport, 350 New Campus Drive, Brockport, NY 14420, USA.

0380-1330/$ – see front matter. Published by Elsevier Inc.
doi:10.1016/j.jglr.2008.12.005

and out-compete the emergent vegetation (Maynard and Wilcox, 1997). If water levels remain low over decades, then succession occurs until disrupted by the next series of high water levels (Wilcox, 2004).

The alternate phase of the cycle begins when water levels rise. Flooding in lower elevations changes the sediments from oxic to anoxic (Ponnamperuma, 1972), inundates mudflat species (van der Valk, 1981), and stresses or kills trees and woody plants (Keddy and Reznicek, 1986). Similarly, soils in higher elevation areas that do not flood may become much wetter, thereby creating a lethal environment for trees and woody plants. As these woody plants die off, the upper limit for herbaceous wetland species is moved upslope and the total area of herbaceous wetland can increase (Keddy and Reznicek, 1982). With time, emergent plants respond to the new water levels and form new communities according to their preferred hydrologic conditions. However, this transformation may not occur if upper limits are determined by anthropogenic barriers (e.g., dikes) rather than woody species (Gottgens, 2000).

These cycles of water-level changes and plant response are repeated over and over again unless the cycle is broken by abnormal hydrology (e.g., stabilization through regulation, extended highs or lows; Wilcox, 2004), invasive species able to survive a wide-range of hydrologic conditions (Saltonstall, 2002), damage to the seed bank (e.g., burned, eroded), or extensive herbivory. Degradation or destruction of the wetland plant communities often occurs if the cycle is disrupted. For example, extended high water levels in Lake Erie and constructed earthen dikes on upslope edges have contributed to the degradation of coastal wetland plant assemblages (Sherman et al., 1996; Kowalski and Wilcox, 1999; Gottgens, 2000; Kowalski et al., 2006). These wetlands likely will remain in a degraded condition until water levels decrease or resource managers take action to promote plant reestablishment. Since the number of coastal wetlands providing critical ecological functions (e.g., fish and wildlife habitat, nutrient uptake, wave attenuation) has decreased in the Great Lakes (Herdendorf 1987; Mitsch and Wang, 2000), those remaining are a high priority for most management agencies.

Many methods to reestablish emergent plant assemblages are available, including direct planting and vegetation mats and logs, but most are expensive, labor-intensive, and difficult to implement over large areas (Kadlec and Wentz, 1974; Wilcox and Whillans, 1999). Furthermore, the hydrologic conditions that contributed to the initial degradation of plant assemblages often continue to make large-scale reestablishment difficult. Thus, there is a need for a means to induce localized low-water conditions temporarily where continuous submergence suppresses normal seed bank germination and plant reestablishment. Permanent solutions, such as installing earthen dikes to isolate the wetlands hydrologically and gain water-level control, have a proven track record but are expensive, require regulatory approval, and can have significant negative impacts on the ecology of coastal wetlands (Johnson et al., 1997; Mitsch et al., 2001; Herrick et al., 2007). Although not without challenges of their own, temporary solutions (e.g., portable, water-filled cofferdams) can have many advantages over permanent solutions, including lower cost, reusable material, less adverse environmental impact, and removal after management objectives are met. Portable cofferdams are available commercially in many shapes and sizes and are capable of making a tight but temporary seal with whatever substrate they rest on and preventing water movement into or out of target areas. The dams are removed after project completion. Portable, water-filled cofferdams are commonly used for construction, river diversion, or flood protection purposes but also have application for ecological rehabilitation projects.

This project explored the effectiveness of using portable, water-filled cofferdams as a management tool to promote the natural growth of emergent vegetation from the seed bank in a Lake Erie coastal wetland. These types of cofferdams have rarely been used to restore wetland habitat. The objectives of this project, therefore, were to evaluate how well portable, water-filled cofferdams temporarily isolate a portion of a wetland and to characterize the wetland seed-bank response at low, middle, and high elevations in areas open to and protected from bird and mammal herbivory.

Methods

Study area

This study focused on the approximately 345-ha Crane Creek drowned-river-mouth wetland located within the U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge (ONWR; 41.628611, −83.207778) along the southern shore of western Lake Erie approximately 30 kilometers east of Toledo, Ohio, USA (Fig. 1). Earthen dikes and rock revetment bound the wetland on all sides except where Crane Creek enters from the west and exits through a channel to Lake Erie on the eastern boundary. Water levels in the wetland are primarily determined by inter-annual and short-term fluctuations (seiches) in water levels of Lake Erie, but inputs from the approximately 146 km^2 Crane Creek watershed can magnify or reduce the effects of changes in Lake Erie water levels, especially after storm events (Kowalski et al., 2006). Open water less than 1 m deep covered much of the wetland in 2003, but short, periodic exposure of mudflats by extreme seiche events combined with high turbidity ensured submersed aquatic vegetation was sparse (Kowalski et al., 2006). Emergent wetland vegetation dominated by *Typha angustifolia* (Narrow Leaved Cattail) and *Phragmites australis* (Common Reed) was growing around the perimeter of the marsh, with floating-leaf assemblages of *Nelumbo lutea* (American Lotus) and *Potamogeton nodosus* (Longleaf Pondweed) extending further from shore. Surrounding earthen dikes and other upland areas supported woody plants, including *Salix* spp. (Willow) and *Populus deltoides* (Eastern Cottonwood). A rich seed bank existed in the approximately 30 cm of silty sediments that overlay hard pan clay (Barry et al., 2004). Very few logs, rocks, or other debris disrupted the nearly uniform sediment surface.

Historically part of the Great Black Swamp that extended from western Lake Erie southwestward to New Haven, Indiana (Kaatz, 1955), most of the coastal marshes along this section of U.S. shore, including parts of the Crane Creek wetland complex, were isolated by earthen dikes in the early 1900s to protect them from Lake Erie's wave energy (Herdendorf, 1987) and promote their management as migratory waterfowl habitat (Campbell and Gavin, 1995). High quality waterfowl habitat remains a priority focus for many managers, but managing coastal and diked wetland habitats for other waterbirds, fish, amphibians, reptiles, and other biota is especially important to the ONWR managers. Armored shoreline and other anthropogenic forces, coupled with frequent high Lake Erie water levels since the early 1970s, contributed to reduction in the area and diversity of coastal wetland vegetation (Kowalski and Wilcox, 1999). These degraded conditions remain because water levels have not dropped low enough during the growing season to expose the seed bank and allow emergent plants to reestablish (NOAA, 2006). Normally, the annual high water levels occur in June and the lowest levels occur in February (NOAA, 2006), but short-term wind tides or seiche fluctuations of up to 3 m above low-water datum are common throughout the year (Herdendorf, 1987).

Portable cofferdams

A series of AquaDams® (i.e., portable, water-filled cofferdams manufactured by Water Structures Unlimited in Carlotta, California, USA) approximately 450 m long was installed temporarily to isolate a 10-ha corner of the Crane Creek wetlands from Lake Erie (see Fig. 1). Conducting this study on a small section of the whole wetland prior to a large-scale implementation of cofferdam technology

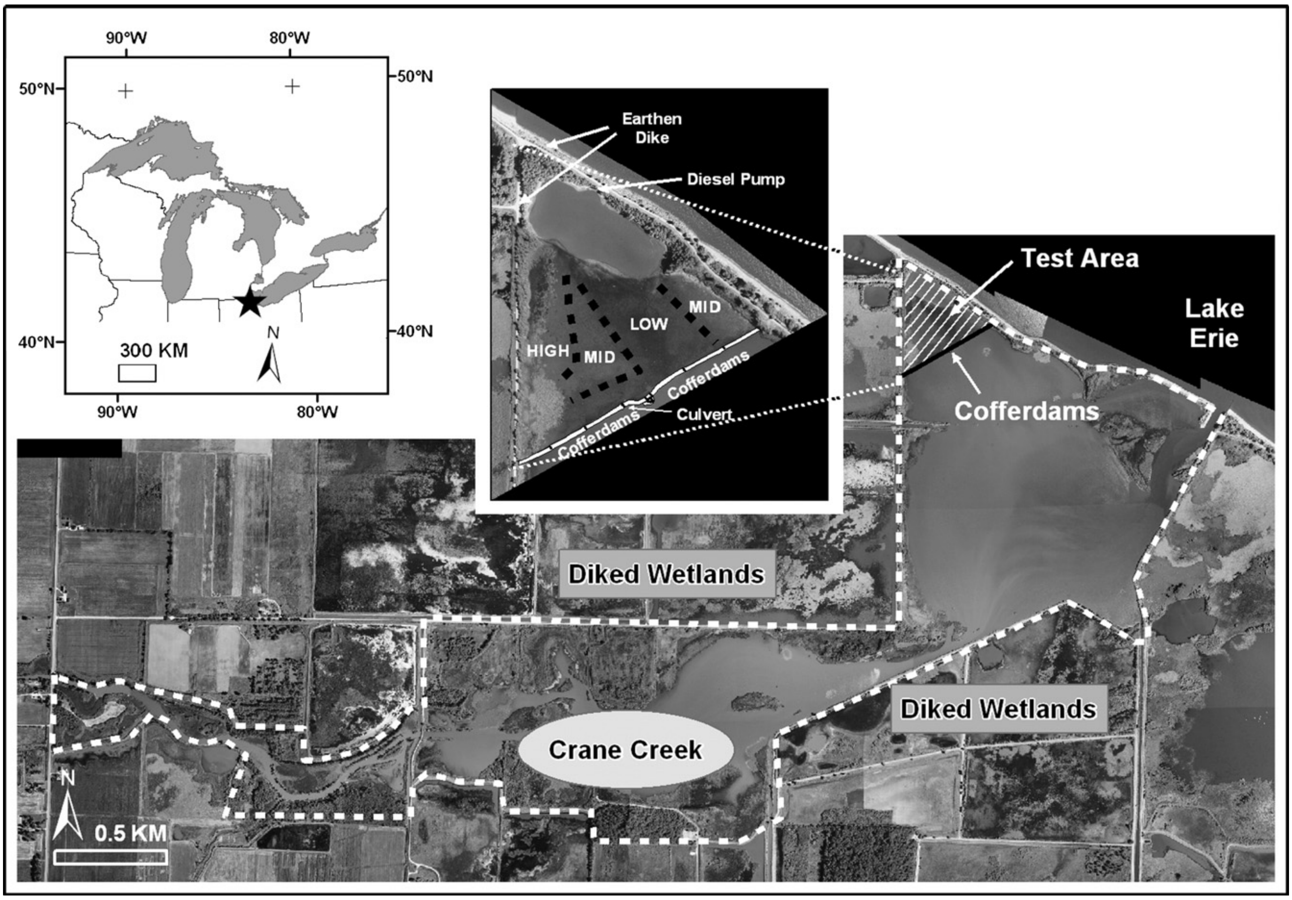

Fig. 1. Maps and 2004 digital orthorectified photograph of Crane Creek study site. Thick white dashed lines indicate boundaries of Crane Creek. Approximate boundaries of elevation zones are noted with black dashed lines.

maximized the likelihood of achieving research objectives and ensured efficient use of resources. Installation of the first set of 1.8-m-high cofferdams began on 19 April 2004 and was completed on 21 April 2004. During the installation, damage to one of the dams resulted in the need for additional dam material to fill a gap between dam sections. New dams were added to the site periodically 8 June 2004 – 25 June 2004 to achieve hydrologic isolation of the test site. Dewatering of the site was achieved by the second full week of July and maintained until the test site was flooded when sections of the cofferdam were washed into the dewatered area on 17 September 2004. Cofferdam material was removed the week of 14 October 2004.

The elevation of the substrate where the dams would be installed was surveyed using laser-plane surveying equipment, and historical water levels in Lake Erie were used to estimate the maximum normal water depth during the study. AquaDams® can range in height from less than a meter to over 4.8 m and are designed to operate in areas where the depth of the water being contained or diverted is less than approximately 70% of the dam height. Per the manufacturer's recommendation, six approximately 70-m-long sections of 1.8-m-high and 4-m-wide cofferdam were filled with water and linked together end-to-end to isolate the test area. In response to problems during the manufacturer's installation and first weeks of operation, additional 1.8-m-high and smaller 76-cm-high auxiliary support dams were added parallel to and on the dewatered side of the larger dams to create the final dam configuration shown in Fig. 1.

Using a diesel-powered pump with 30.5-cm-diameter hoses, the water behind the dams was drawn down to an elevation that fully exposed the majority of the marsh sediments, similar to a natural low-water year. Standard dam maintenance was performed and pumping occurred regularly to maintain moist-soil conditions in the test area from the initial drawdown in July 2004 through premature failure of the dams in September 2004 (Kowalski et al., 2006). As a result of the failure, all of the dams and maintenance equipment were removed from the site in October 2004 rather than in the Fall of 2005 as intended.

Sediment elevation measurements were made after the dewatering was complete to characterize the topography of the dewatered area. A total station, laser-plane surveying equipment, and standard land-surveying methods were used to collect and tie sediment surface elevation data to a first-order U.S. Geological Survey benchmark. Since there were small differences in sediment surface elevation in the dewatered area, the surveying equipment was used to identify the boundaries of three major elevation zones (i.e., low, mid, high). Measured elevations ranged from 173.70 m to 173.93 m. All vegetation sampling in the low zone occurred at elevations less than 173.78 m. Sampling in the mid zone occurred between 173.80 m and 173.86 m, and sampling in the high zone occurred at elevations greater than 173.88 m. All elevations are reported with reference to the International Great Lakes Datum 1985.

Since bird and mammal herbivory of young plants can significantly influence seed bank driven revegetation of a wetland (Lynch et al., 1947; Barry et al., 2004), thirty 2 m × 2 m herbivory exclosures were built and placed in the dewatered area behind the cofferdams after dam installation. The exclosures (i.e., poultry wire strung around and over four metal posts at least 1.5 m high) allowed analysis of the effects of herbivory in a recently dewatered area when compared to data collected inside the exclosures. Ten exclosures were placed randomly in each of the three elevation zones.

Sampling and analysis

The vegetation in the 30 exclosures was sampled quantitatively using a 1 m × 1 m quadrat centered in each exclosure prior to flooding in September 2004 and again in August 2005, approximately 11 months after the site was hydrologically reconnected to Lake Erie. During the same time periods, 10 open (i.e., unprotected from herbivory) quadrats were placed randomly outside the exclosures but within each of the three elevation zones in the dewatered area. Therefore, a total of 60 quadrats were sampled in the dewatered area each year. For this analysis, quadrats in each combination of elevation (e.g., low, mid, high) and herbivory protection (e.g., exclosure, open) were considered a sampling group. Additional quadrats were sampled in nearby areas of Crane Creek that were at elevations similar to the dewatered area yet remained under the hydrologic influence of Lake Erie. These reference quadrats were considered a separate sampling group for each year. Plant species found in all quadrats were identified and assigned a percent cover value using visual estimation. Investigators regularly estimated percent cover values in test plots to minimize differences among sampling teams. No sampling was done prior to Fall 2004 because photo interpretation and site visits revealed very little wetland vegetation in the study site, excluding fringe stands of *T. angustifolia*, *P. australis*, and *N. lutea* (Kowalski et al., 2006). Herbaceous plant nomenclature followed Flora of North America (www.eFloras.org) and tree nomenclature followed Barnes and Wagner (2004).

Plant species richness (i.e., number of taxa) and importance values (i.e., sum of relative frequency and relative cover of each taxon in a sampling group; Curtis and McIntosh, 1951) were calculated using data collected during quadrat sampling. The importance values were analyzed using non-metric multidimensional scaling (NMDS) to explore differences associated with herbivory (i.e., exclosure and open to herbivory) and low, mid, and high elevation zones (McCune and Grace, 2002). The analysis was performed using the PC-ORD version 5.1 with the Bray-Curtis distance measure (Bray and Curtis, 1957; McCune and Mefford, 2006). Dimensionality of the data set was determined by using a random starting number, 250 runs with real data, 250 runs with randomized data, and 500 maximum iterations. The analysis was repeated with only the recommended number of dimensions (i.e., three) and without the Monte Carlo test.

Results

The nearly two-month drawdown maintained by the cofferdams produced a rapid and diverse response from the seed bank that was not observed in the reference plots. Thirty-nine of the forty-two plant and alga taxa found during the 2004 sampling were identifiable to species (Table 1). Thirty of those taxa were emergent herbaceous or woody species. Even though they were found at elevations similar to the plots in the dewatered area, all taxa sampled in the 2004 reference group were submersed aquatic or floating-leaf species except *Eleocharis acicularis* (Needle Spike Rush) and *N. lutea*. Three of the six submersed aquatic taxa found in the reference group (i.e., *Ceratophyllum demersum* (Coontail), *Myriophyllum sibiricum* (American Watermilfoil), *Vallisneria americana* (Eel Grass)) were not found anywhere in the dewatered area. The alga taxa sampled in 2004 were not identifiable to species.

A different suite of eighteen taxa were sampled under the flooded conditions in 2005. All of the woody taxa found in 2004 were absent in 2005, and only three of the fifteen taxa identifiable to species (Table 1) were not submersed aquatic or floating-leaf species (i.e., *Butomus umbellatus* (Flowering Rush), *N. lutea*, *Pontederia cordata* (Pickerelweed)). Total species richness among the sampling years and groups ranged from the least (5 taxa) in the 2005 low elevation exclosure and high elevation open sites to the greatest (27 taxa) in the 2004 high open site (Table 2). The average species richness among the 2004 sampling groups (19.6 species) was more than double the 2005 sampling groups (7.1 species).

Differences among sampled groups and years were apparent when NMDS was used to analyze the importance value data. The data best fit a 3-dimensional model, but only axis 1 and axis 3 are shown because they accounted for most of the variation (Figs. 2a and 2b). There was a clear separation of groups based on the degree of flooding along axis one, which explained 57.4% of the variation. The mudflat assemblages found during the 2004 dewatered conditions were tightly grouped toward the left side of axis 1, while the submersed aquatic-dominated assemblages found in the 2004 reference plots and all of the 2005 plots were grouped toward the right side of axis 1 (Fig. 2a). For both years, there was a pattern of separation among the low, mid, and high zones along axis 3 that explained 24.4% of the variation (see Fig. 2a). The 2004 and 2005 reference data grouped with the 2005 high elevation data dominated by submersed aquatic species adapted to

Table 1
List of plant species collected in Crane Creek in 2004 and 2005.

Species	Code	Form	2004	2005
Abutilon theophrasti Medikus (Velvetleaf)	ABUTHE	E	X	
Ammannia robusta Heer & Regel (Grand Redstem)	AMMROB	E	X	
Butomus umbellatus L. (Flowering Rush)	BUTUMB	E		X
Ceratophyllum demersum L. (Coontail)	CERDEM	S	X**	X*
Cyperus bipartitus Torr. (Shining Flatsedge)	CYPBIP	E	X	
Cyperus diandrus Torr. (Umbrella Flatsedge)	CYPDIA	E	X	
Cyperus erythrorhizos Muhl. (Red Rooted Flatsedge)	CYPERY	E	X	
Cyperus odoratus L. (Rusty Flatsedge)	CYPODO	E	X	
Eleocharis acicularis (L.) R. & S. (Needle Spike Rush)	ELEACI	E	X*	
Eleocharis obtusa (Willd.) Schultes (Blunt Spike Rush)	ELEOBT	E	X	
Eragrostis hypnoides (Lam.) BSP (Creeping Lovegrass)	ERAHYP	E	X	
Heteranthera dubia (Jacq.) MacM. (Grassleaf Mudplantain)	HETDUB	S		X
Hibiscus trionum L. (Rosemallow)	HIBTRI	E	X	
Lactuca serriola L. (Prickly Lettuce)	LACSER	E	X	
Lemna minor L. (Common Duckweed)	LEMMIN	O	X*	X*
Myriophyllum sibiricum Komarov (American Watermilfoil)	MYRSIB	S	X**	
Myriophyllum spicatum L. (Eurasian Watermilfoil)	MYRSPI	S		X
Najas marina L. (Spiny Naiad)	NAJMAR	S	X	X
Najas minor Allioni. (Brittle Waternymph)	NAJMIN	S		X*
Nelumbo lutea Wildenow (American Lotus)	NELLUT	E	X*	X
Penthorum sedoides L. (Ditch Stonecrop)	PENSED	E	X	
Phalaris arundinacea L. (Reed Canarygrass)	PHAARU	E	X	
Phragmites australis (Cav.) Steudel (Common Reed)	PHRAUS	E	X	
Polygonum lapathifolium L. (Nodding Smartweed)	POLLAP	E	X	
Polygonum pensylvanicum L. (Pennsylvania Smartweed)	POLPEN	E	X	
Pontederia cordata L. (Pickerelweed)	PONCOR	E	X	X
Populus deltoides Marshall (Eastern Cottonwood)	POPDEL	O	X	
Potamogeton crispus L. (Curled Pondweed)	POTCRI	S		X*
Potamogeton foliosus Raf. (Leafy Pondweed)	POTFOL	S	X*	X*
Potamogeton nodosus Poiret. (Longleaf Pondweed)	POTNOD	S	X*	X*
Potamogeton pectinatus L. (Sago Pondweed)	POTPEC	S	X*	X*
Potamogeton richardsonii (Benn.) Rydb. (Redhead Pondweed)	POTRIC	S		X
Rhus hirta (L.) Sudworth (Staghorn Sumac)	RHUHIR	O	X	
Riccia fluitans L. (Crystalwort)	RICFLU	S	X	
Rorippa palustris (L.) Besser (Common Yellowcress)	RORPAL	E	X	
Rumex crispus L. (Curly Dock)	RUMCRI	E	X	
Sagittaria latifolia Willd. (Duck Potato)	SAGLAT	E	X	
Salix cordata Michx. (Heartleaf Willow)	SALCOR	O	X	
Salix eriocephala Michx. (Missouri Willow)	SALERI	O	X	
Salix exigua Nutt. (Sandbar Willow)	SALEXI	O	X	
Salix fragilis L. (Crack Willow)	SALFRA	O	X	
Schoenoplectus tabernaemontani (C.C. Gmelin) Palla (Softstem Bulrush)	SCHTAB	E	X	
Scirpus fluviatilis (Torr.) A. Gray (River Bulrush)	SCIFLU	E	X	
Typha angustifolia L. (Narrow Leaved Cattail)	TYPANG	E	X	
Vallisneria americana L. (Eel Grass)	VALAME	S	X**	X**

Code lists the abbreviations used in Fig. 2b. Form is designated as emergent (E), submersed aquatic (S), or other (O). "X" indicates present. "*" indicates found in reference plots and "**" indicates only found in reference plots. Table only includes taxa identifiable to species.

Table 2
List of the plant and alga taxa with the top five importance values collected in the drawdown area behind the cofferdam in Crane Creek in 2004 and 2005.

Taxa	Importance value													
	Low				Mid				High				Reference	
	Open		Excl		Open		Excl		Open		Excl			
	2004	*2005*	*2004*	*2005*	*2004*	*2005*	*2004*	*2005*	*2004*	*2005*	*2004*	*2005*	*2004*	*2005*
Butomus umbellatus		20.5												
Ceratophyllum demersum												7.9		
Cyperus erythrorhizos	44.0		29.6		82.1		42.4		28.6		21.4			
Eleocharis acicularis							14.5		31.2		60.3		8.9	
Eleocharis obtusa	17.1													
Heteranthera dubia		30.7		19.2										
Lemna minor													33.6	25.7
Myriophyllum spicatum		23.3												
Najas marina	9.7			25.8				9.6						
Najas minor						103.7		86.6		47.2		119.4		22.9
Nelumbo lutea										7.6				
Nitella sp.										15.7		15.7		
Polygonum lapathifolium							18.0							
Pontederia cordata			23.3			15.9								
Potamogeton crispus						16.7								
Potamogeton foliosus				90.4										
Potamogeton nodosus		14.6				19.0		36.5	39.4	116.0	25.2	33.8	105.6	92.2
Potamogeton pectinatus		90.5		64.6		24.7		24.5		13.6		7.7	21.1	18.2
Potamogeton richardsonii				19.2				22.6						
Rumex crispus					28.4				12.5		12.9			
Sagittaria latifolia	41.7		29.1		8.8		14.9							
Salix cordata					7.3									
Salix eriocephala					7.3				9.3					
Salix exigua											10.8			
Schoenoplectus tabernaemontani			42.2											
Typha angustifolia	16.0		23.9				18.3							
Vallisneria americana													14.0	15.1
Species richness	**19**	**7**	**17**	**5**	**21**	**7**	**23**	**8**	**27**	**5**	**22**	**8**	**9**	**8**

Missing values do not necessarily indicate the absence of taxa, because taxa might have importance values below the five highest values. Species richness of each sampling group is noted.

flooded conditions (Fig. 2b). There was no discrimination between the open or exclosure groups among the zones, but the presence of the emergent invasive species *B. umbellatus* in the 2005 low elevation open group contributed to its separation from the 2005 low elevation exclosure group. An additional 10.0% of the variation was explained by the second axis (not shown), although no ecological groups or patterns were apparent along that axis.

The importance values for the individual taxa sampled in 2004 revealed few differences in the dominant species (i.e., those with the five highest importance values) among all of the elevation zones except the presence of *Schoenoplectus tabernaemontani* (Softstem Bulrush) in the low exclosure and *P. nodosus* in the high elevation open and exclosure quadrats (Table 2). No *Schoenoplectus* was found in the low open quadrats. The *Potamogeton* spp. were rooted prior to the drawdown and survived on the wet mudflats. *Cyperus erythrorhizos* (Red Rooted Flatsedge) and other classic mudflat taxa were common among all 2004 sampling groups, which contributed to the high (i.e., 17–27 taxa) species richness in 2004. The species richness dropped significantly to a range of 5–8 taxa per sampling group by 2005 after cofferdam failure. A suite of *Potamogeton* species replaced most of the emergent species, and *Najas minor* (Brittle Waternymph) became much more dominant. Except for the presence of *P. nodosus* in the high open group, there were no clear differences in the composition of samples taken inside and outside of the exclosures.

Discussion

The loss of emergent vegetation in Great Lakes coastal wetlands during high water levels is part of the cycle of destruction and renewal caused by naturally fluctuating water levels (Keddy and Reznicek, 1985). Subsequent low-water levels during the growing season expose the seed-rich sediments and promote the natural regeneration of wetland plants. If anthropogenic disturbance (e.g., altered hydrology) or extended high water levels coupled with upslope backstopping (Gottgens, 2000) prevent exposure of the sediments, then the wetlands remain in a degraded state until water levels recede naturally or management actions are employed to restart the cycle. Water-filled, portable cofferdams are one of many technologies currently available to separate a section of river, lake, or wetland hydrologically from its parent waterbody. Unlike cofferdams with a rigid design made out of plastic or other materials, soft-bodied dams (i.e., geotextile material wrapped around a seamless liner) like the Aquadam® used in this study are flexible enough to mold around irregularities in sediments and make a water-tight seal with the bottom. This temporary seal allows managers to conduct a drawdown that mimics conditions found during a low-water year. If a viable seed bank exists in the marsh, then simply exposing the sediments elicits a positive response from the seed bank. However, this response is short-lived and habitat is not reestablished if dewatered conditions are not maintained long enough to allow the plants to mature. Unlike earthen dikes, the footprints of these portable cofferdams have minimal ecological impact (e.g., sediment disruption) and can be removed from the marsh after plants reestablish or management objectives have been met. Experiences during this study, however, revealed that a significant amount of effort (e.g., planning, installation, maintenance) is required to maximize the likelihood of maintaining dewatered conditions long enough to meet project objectives and technological improvements are needed to make these dams viable for extensive habitat restoration projects.

Maintaining dewatered conditions

Although the portable, water-filled cofferdams used in this project only maintained dewatered conditions for a short time, lessons were learned that can be used to improve future deployments in Great Lakes coastal wetlands (see Appendix A for additional details). We

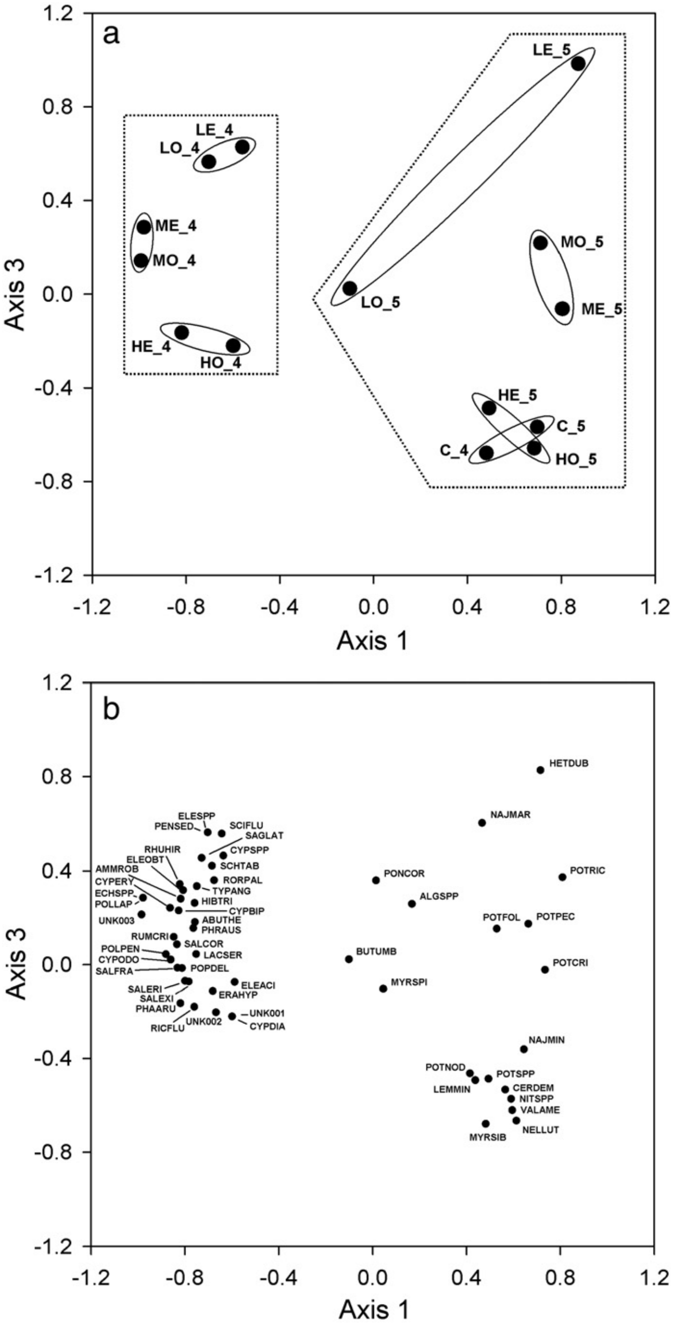

Fig. 2. The first and third axes of the non-metric multidimensional scaling ordination, based on Importance Values calculated on fifty-two wetland plant taxa and two algal groups collected in the sampling groups in 2004 and 2005. a) Ordination of sites identified by location in the elevation gradient (high (H), middle (M), low (L)); exclosure (E) or open (O); and year (2004, 2005). b) Ordination showing taxa with high importance values, labeled using the first three letters of the genus and first three letters of the species identified in Table 1. Taxa not found in Table 1 include alga spp. (ALGSPP), *Cyperus* spp. (CYPSPP), *Echinochloa* spp. (ECHSPP), *Eleocharis* spp. (ELESPP), *Nitella* spp. (NITSPP), *Potamogeton* spp. (POTSPP), and unknown emergents (UNK001-3). Final 3-dimensional solution stress was 4.98820 after 99 iterations.

found that selection and preparation of the study site is very important to establish dewatered conditions and maximize response from the seed bank. Optimal installation sites will have a reliable source of clean water to pump into the dams, easy access by people and heavy equipment, a limited amount of rocks, trees, or other debris in the sediments under the cofferdams, and a rich seed bank in the area to be dewatered. In addition to site selection, we found that using a product sized appropriately for the application is critical for maintaining dewatered conditions long enough to allow seedlings to reach maturity. Undersized cofferdams are vulnerable to being overtopped by high water levels or undermined by erosion, water seepage, or wildlife activities, while oversized dams are more expensive and may be more difficult to install and maintain. Although water depth is the most important factor to consider when selecting dam size (Water Structures Unlimited, 2004), there are many other factors that can influence cofferdam performance, including installation and maintenance.

It is essential to have the proper equipment (see Appendix B) on site during installation and maintenance of the dams to prevent delays and additional expenditures. In addition, problems encountered during installation must be fixed immediately, and any damaged dams must be replaced rather than repaired to minimize the chance of later problems. Once the cofferdams are installed and filled with water, regular and often labor-intensive maintenance activities are required to keep the dams full and to maintain dewatered conditions at the study site.

Wetland plant growth from the seed bank

Moist-soil conditions were maintained in our study site for about two months. During these two months, the cofferdams effectively created conditions for seed-bank derived growth of emergent wetland vegetation. Shortly after the seed bank was exposed in July 2004, seeds from over 40 different taxa began to germinate, as they likely would have during a low-water year (Keddy and Reznicek, 1985; see Table 1). Previous studies found an extensive seed bank in Crane Creek and neighboring coastal marshes (Wilcox and Kowalski, 1995; Davis and Welch, 2000; Barry et al., 2004), but areas that have not been vegetated for a long time or have been eroded by waves may have a severely diminished seed bank.

Most of the plants growing in the dewatered area of Crane Creek were mudflat wetland species with seeds that remain dormant but viable in the seed bank for a long time. However, there were some plants that likely came from seeds transported to the recently dewatered sediments via wind or other vectors. *Salix* spp. and *P. deltoides*, for example, are woody taxa that often become densely established in wetlands when sediments are exposed. If sufficient sources are available, wind-dispersed seeds land in fertile wetland sediment and quickly germinate. Unlike in a neighboring coastal marsh (Kowalski and Wilcox, 1999), these woody species were not a large component of the plant assemblages growing among the elevation zones within the dewatered area (see Table 2) because the marsh was not fully dewatered until July. Most *Salix* and *Populus* species flower and produce seeds in late spring or early summer (June for western Lake Erie). The drawdown occurred after most of these woody species should have reproduced (Chadde, 2002), so their seeds likely had already been distributed by the wind. The woody seedlings that did grow during the drawdown were not able to survive the flooding after the cofferdams failed and were removed, so the timing of the drawdown and subsequent flooding were important in preventing invasion by woody species. The absence of woody species growing at the reference sites both during and after the management drawdown suggests that the dewatering action allowed the temporary growth of woody species but flooded conditions were not conducive to their establishment or growth. Management-driven drawdowns often are conducted later in the growing season to minimize the establishment of woody species and promote a greater diversity of wetland species (Fredrickson and Taylor, 1982). Late season drawdowns also can be used to target the growth of certain emergent and submersed aquatic plant species for waterbirds (Keith, 1961; Payne, 1992), although certain plants established late in the season can become management problems in subsequent years (Meeks, 1969).

In addition to the timing of a management drawdown, small but ecologically important differences in elevation of the marsh sediments can influence species richness and composition (see Fig. 2b). Ordinations revealed similarities among taxa collected at each elevation zone as well as dissimilarity among the elevation zones. The NMDS-derived groupings (i.e., circles in Fig. 2a) show a pattern among the sampling groups along axis 3, with low elevation sampling groups having the largest axis 3 values and the high sampling groups with the smallest values. This pattern is apparent in both the 2004 data and the 2005 data, with the exception of the 2005 low open sampling group (LO_5). This group is an outlier because the invasive species *B. umbellatus* was present. *Butomus umbellatus* is an aggressive perennial herb that establishes quickly and can persist in flooded conditions (Hroudova et al., 1996). The LO_5 sampling group was the only one where *B. umbellatus* had a high importance value, so it plotted closer to the groups composed of emergent taxa. The reference sampling groups for both 2004 and 2005 grouped close to the 2005 high elevation data in the NMDS because they were located at similar elevations and there was a strong presence of submersed aquatic species. The reference groups did not receive the dewatering treatment, and their species composition did not differ much between the two years, so we are confident that the significant differences observed in the dewatered area were the result of the hydrologic changes associated with the 2004 drawdown treatment and subsequent reflooding in 2005. The observed differences in plant assemblages associated with each elevation in 2004 likely are tied to differences in soil moisture during germination suggesting that even small elevation differences in dewatered sediments can affect the seed bank germination success and ultimately the composition of plant assemblages. In contrast, the 2005 data suggest that, when flooded, only relatively large differences in water depth (and therefore light availability) associated with each elevation zone influence species presence.

Although grazing of wetland plant seedlings can be a management problem, this study did not detect a strong overall effect of herbivory on the species richness of wetland plants growing in the dewatered area. However, some plant species only occurred in the plots protected from herbivory, while others only grew in plots open to the full effects of herbivory. For example, *S. tabernaemontani* had the greatest importance value for the 2004 low exclosure data but unexpectedly did not appear at any of the low elevation areas not protected by exclosures. Five other species (*E. acicularis*, *Polygonum lapathifolium* (Nodding Smartweed), *P. cordata*, *Salix exigua* (Sandbar Willow), *T. angustifolia*) also had high importance values only in the protected sample sites. Conversely, only two species (*Eleocharis obtusa* (Blunt Spike Rush), *Najas marina* (Spiny Naiad)) had high importance values in the open sites. These results could be in response to many factors (e.g., synchronicity between waterbird migrations and seedling growth, herbivore disturbance by the presence of the cofferdams, a seed bank with high diversity and variation in density), but the absence of a strong pattern suggests that plant herbivory may be present at a site without impacting the composition of developing plant assemblages.

Regardless of protection from herbivory, the species richness was high during the 2004 drawdown in the low, mid, and high elevation zones. The low elevation zone had fewer taxa than the other zones, likely because the sediments in much of this zone remained saturated or in some places were covered by very shallow water. This zone was dry immediately after the drawdown began, but water channeling under a dam flowed over this zone throughout the project and likely prevented some emergent plants from germinating. Where present, the shallow surface water supported submersed aquatic taxa (e.g., *Potamogeton* spp.) common in the reference sampling group but generally absent from the higher elevation zones of the dewatered area. Similarly, the 2004 reference sampling group and all of the 2005 sampling groups remained inundated and, as a result, had many fewer species.

The plant assemblage changed dramatically after the cofferdams failed and the hydrologic connection to Lake Erie was restored to the site in late 2004, when much of the test area was covered by over 71 cm of water (see axis 1 values in Fig. 2). Although off to a good start, most emergent species had not grown tall enough during the brief drawdown to survive inundation by the late-summer high water levels. These emergent plants were replaced in 2005 by a suite of submersed aquatic species that tend to thrive in deeper water. A similar suite of species was found in other parts of Crane Creek that did not receive the dewatering treatment, so it appears that the post-cofferdam reflooding promoted the quick return of pre-drawdown submersed aquatic plant assemblages. If the sediments had been exposed during a time of low water levels in Lake Erie, emergent plants likely would have had one or more growing seasons to reach maturity. The height advantage achieved by many plants at maturity would allow them to survive higher water levels, as aerenchyma tissue could reach atmospheric oxygen, and the benefits of increased wetland habitat would last longer. Not surprisingly, the length of time that the marsh seed bank is exposed is critical to the longevity of seed-bank-driven plant growth in Great Lakes coastal marshes.

Implications for large-scale habitat rehabilitation

The intent of this study was to test a novel technology that created temporarily dewatered conditions in a section of coastal marsh to allow wetland plants to grow from the seed bank. The study revealed both the potential benefits of applying this management tool in coastal wetlands and a number of challenges that must be addressed prior to large-scale implementation. Understanding the operation and technical details of the cofferdam technology is critical in determining how to maximize the response from the seed bank and promote the long-term survival of emergent plants (i.e., habitat rehabilitation). Many significant problems were identified during tests of early cofferdam designs during the studies performed in Lake Ontario coastal wetlands (i.e., Cootes Paradise) in the early 1990s (Wilcox and Whillans 1999). Vandalism and product design issues proved to be the biggest challenges that prevented large-scale implementation in Cootes Paradise and in Crane Creek (see Appendix A). Although a different suite of challenges arose during the test at Crane Creek, our limited results show that this tool can be used to isolate portions of a coastal marsh temporarily and promote plant growth. However, the extent and longevity of that growth depends on the length of time that dewatered conditions are maintained and the hydrologic conditions present once the dams are removed. A tool like this is of particular interest to managers of highly-degraded coastal wetland habitats because it has the potential to provide the benefits of hydrologic isolation without causing long-term damage to wetland sediments or permanently altering the hydrology.

Advancements in the technology and the implementation process will continue to improve the odds of successfully achieving research and management objectives in similar wetland habitat rehabilitation projects throughout the Great Lakes. Although whole wetland complexes may not be able to be rehabilitated at once, these relatively small-scale habitat rehabilitation projects can provide localized benefit to the system and, in aggregate, improve the habitat available to Great Lakes biota. The temporary and highly customizable (e.g., height, length) design of portable cofferdams also supports their repeated use in one area over time or in multiple areas within a wetland. This technology, therefore, can be a potentially important tool in the arsenal used by Great Lakes resource managers.

Acknowledgments

Financial support from the U.S. Environmental Protection Agency Great Lakes National Program Office allowed the research team from the USGS Great Lakes Science Center and University of Michigan School of Natural Resources and Environment to complete this project successfully. We thank U.S. Fish and Wildlife Service Ottawa National Wildlife Refuge managers Doug Brewer and Dan Frisk for their permission to access refuge property during this project and ONWR staff for continued assistance with field data collection. We also thank Jim McKenna, Richard Grosshans, and the anonymous reviewers of this manuscript for their time and effort. We appreciate the help of the many students at the University of Michigan School of Natural Resources and Environment who assisted with field work and data analysis. A special thank-you goes to Gabe Eickhoff and Eli Davidian whose exceptional work ethic and tireless days in the field kept this project going. Any use of trade, product, or firm names is for descriptive purposes only and does not imply endorsement by the U. S. Government. This article is Contribution 1508 of the USGS Great Lakes Science Center.

Appendix A. List of recommendations for future Great Lakes coastal wetland habitat rehabilitation projects involving water-filled portable cofferdams

Site selection:

- Plan where the cofferdams will connect to upland areas and prepare the site by removing vegetation, rocks, and other debris. Easy access to the installation site by boat, truck, and heavy equipment saves time, money, and resources for the duration of the project.
- Ensure that clean water can be transported to each of the cofferdam sections the entire time they are installed.
- Consider the impacts of dam installation and drawdown occurring at different times of the year (e.g., spring fish spawning, summer seed dispersal).
- Prior to dam installation, search and remove debris, native clams, and any other objects from the intended cofferdam location.
- A seed bank study, historical observations, or other data are needed to verify presence of a seed bank prior to implementation of cofferdam technology.
- Ensure that soft sediments in the dam location are not too deep or otherwise unsuitable for the cofferdam to seal to the bottom.
- Install highly visible rope from endpoint to endpoint to serve as a guide during installation.

Dam characteristics:

- Calculate manufacturer-recommended cofferdam height based on evaluation of historical hourly high water levels recorded at the nearest Great Lakes water-level gage.
- Because of the high variability in water-level fluctuations in the Great Lakes, use larger dams than recommended by the manufacturer to promote a tight seal on the bottom, accommodate unexpectedly high water levels, and maintain dewatered conditions long enough to allow plants emerging from the seed bank to reach maturity.
- Minimize the number of dam sections used to isolate the project area because each connection is the most likely place where the integrity of the dam will be compromised.
- Prepare for over four weeks of dam manufacture time for large projects and the time associated with shipping replacement dams from distant locations.
- Request installation of air release valves in the dam bladders. Pumps often fill dams with air that is very difficult to remove without a release valve.
- If dams are purchased with the intent to reuse, plan for a labor-intensive effort to remove sediment and debris inside dams before storage in a dry, pest-free location.
- Consider the effects of water currents and wind on dams being prepared for installation.
- Install sheets of plastic connected to the dams and anchored by chains on the bottom of the uncontrolled side of the dam to minimize leakage under the dams.
- Install an appropriately sized culvert and water-control structure to allow controlled movement of water from one side of the dam to the other.

Operation:

- After the dams are installed, attach a network of hoses to the dams to allow the dams to be filled with minimal movement of the supply pump (see Kowalski et al., 2006).
- Routinely monitor water levels inside the dams. During our project, the dams required maintenance pumping nearly twice per week to maintain their full size and minimize water movement under the dams.
- Secure a taut rope across the top of each dam segment using 2.54-cm PVC conduit driven into sediments on each side of dam. Vertical movement (i.e., inflation) of the dam can be monitored by measuring the distance between the rope and top of the dam.
- Carefully monitor the presence of small holes in the seamless cofferdam liner daily. The holes can develop at weak spots in the material or where the liner is punctured by a sharp object. Until patched, water will leak out of the holes and the volume of water inside the dams will decrease enough to change their shape significantly and compromise connections with other dams or the dams' tight seals on the marsh bottom.
- Install signs and fences to educate the public and deter vandalism.
- Install fencing in the water on the uncontrolled side of the dam to prevent damage from fish spines.

Appendix B. List of items that were or could have been useful during cofferdam installation (I) and maintenance (M)

- Aluminum trash pumps with 7.62-cm-diameter light duty discharge hoses (I,M)
- PVC intake hoses (7.62-cm dia) with screens and buckets to limit sediment intake (I,M)
- Onsite fuel and oil supply (I,M)
- Spools of twine and heavy nylon rope (I,M)
- Neoprene and leather gloves (I)
- Professional grade duct tape (I,M)
- Bird deterrents on dams to prevent damage to fill tubes (M)
- Excavator or other heavy equipment to move large dams on land (I)
- High quality radios and cell phones in waterproof sleeves (I,M)
- Automotive tires and axles to allow rolled cofferdams to be transported over inflated cofferdams (I)
- Small portable boat (I,M)
- Sheets of plywood to serve as rigid platforms on dams (I,M)
- 5.08 cm × 30.48 cm treated boards to allow access to top of dam from land (I,M)
- Wagon or cart with pneumatic tires to carry equipment (e.g., water pumps) (I,M)

References

Barnes, B.V., Wagner Jr., W.H., 2004. Michigan trees: a guide to the trees of the great lakes region. The University of Michigan Press, Ann Arbor, Michigan.

Barry, M.J., Bowers, R., De Szalay, F.A., 2004. Effects of hydrology, herbivory and sediment disturbance on plant recruitment in a Lake Erie coastal wetland. Am. Midl. Nat. 151, 217–232.

Batterson, T., McNabb, C., Payne, F., 1991. Influence of water level changes on distribution of primary producers in emergent wetlands in Saginaw Bay. Mich. Acad. 23, 149–160.

Bray, J.R., Curtis, J.T., 1957. An ordination of the upland forest communities of southern Wisconsin. Ecol. Monogr. 27, 325–349.

Campbell, L.W., Gavin, C., 1995. The marshes of southwestern Lake Erie. Ohio University Press, Athens, Ohio.

Chadde, S.W., 2002. A Great Lakes wetland flora: a complete guide to the aquatic and wetland plants of the Upper Midwest, Second edition. PocketFlora Press, Laurium, Michigan.

Curtis, J.T., McIntosh, R.P., 1951. An upland forest continuum in the prairie-forest border region of Wisconsin. Ecology 32 (3), 476–496.

Davis, C.B., Welch, B.A. 2000. *Disturbance, vegetation, and floristic diversity in a Lake Erie coastal wetland*. Final Report, Lake Erie Protection Fund SF119-99.

Fredrickson, L.H., Taylor, T.S. 1982. *Management of seasonally flooded impoundments for wildlife*. U.S. Fish and Wildlife Service, Resource Publication 148.

Gottgens, J.F., 2000. Wetland restoration along the southwestern Lake Erie coast line: case studies and recommendations. Toledo Journal Gt. Lakes Law, Sci. Policy 3, 49–65.

Harris, S.W., Marshall, W.H., 1963. Ecology of water-level manipulations on a northern marsh. Ecology 44, 331–343.

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

Herrick, B.M., Morgan, M.D., Wolf, A.T., 2007. Seed banks in diked and undiked Great Lakes coastal wetlands. Am. Midl. Nat. 158, 191–205.

Hroudova, Z., Krahulcova, A., Zakravsky, P., Jarolimova, V., 1996. The biology of *Butomus umbellatus* in shallow waters with fluctuating water levels. Hydrobiologia 340, 27–30.

Johnson, D.L., Lynch, W.E., Morrison, T.W., 1997. Fish communities in a diked Lake Erie wetland and an adjacent undiked area. Wetlands 17, 43–54.

Kaatz, M.R., 1955. The Black Swamp: a study in historical geography. Ann. Assoc. Am. Geogr. 45, 1–35.

Kadlec, J.A., Wentz, W.A., 1974. State-of-the-art survey and evaluation of marsh plant establishment techniques: induced and natural, Volume I. U.S. Army Coastal Engineering Research Center, Fort Belvoir, Virginia.

Keddy, P.A., Reznicek, A.A., 1982. The role of seed banks in the persistence of Ontario's coastal plain flora. Am. J. Bot. 69, 13–22.

Keddy, P.A., Reznicek, A.A., 1985. Vegetation dynamics, buried seeds, and water level fluctuations on the shorelines of the Great Lakes. In: Prince, H.H., D'Itri, F.M. (Eds.), Coastal Wetlands. Lewis Publishers, Inc., Chelsea, Michigan, pp. 33–51.

Keddy, P.A., Reznicek, A.A., 1986. Great Lakes vegetation dynamics: the role of fluctuation water levels and buried seeds. J. Great Lakes Res. 12, 25–36.

Keith, L.B., 1961. A study of waterfowl ecology on small impoundments in southeastern Alberta. Wildl. Monogr. 6, 3–88.

Kowalski, K.P., Wilcox, D.A., 1999. Use of historical and geospatial data to guide the restoration of a Lake Erie coastal marsh. Wetlands 19, 858–868.

Kowalski, K.P., Wilcox, D.A., Wiley, M.J. 2006. *Coastal wetland habitat restoration and exploration*. United States Environmental Protection Agency, Great Lakes National Program Office, Final Report. Project No.: GL2003-304.

Lynch, J.J., O'Neil, T., Lag, D.W., 1947. Management significance of damage by geese and muskrats to gulf coast marshes. J. Wildl. Manage. 11, 50–76.

Maynard, L., Wilcox, D. 1997. *Coastal wetlands. State of the lakes ecosystem conference 1996 background paper*. Environmental Protection Agency 905-R-97-015b.

McCune, B., Grace, J.B., 2002. Analysis of ecological communities. Oregon: MjM Software, Gleneden Beach.

McCune, B., Mefford, M.J., 2006. Multivariate analysis of ecological data. Version 5.10. MjM Software, Gleneden Beach, Oregon, USA.

Meeks, R.L., 1969. The effect of drawdown date on wetland plant succession. J. Wildl. Manage. 33, 817–821.

Mitsch, W.J., Wang, N., 2000. Large-scale coastal wetland restoration on the Laurentian Great Lakes: determining the potential for water quality improvement. Ecol. Eng. 15, 267–282.

Mitsch, W.J., Wang, N., Bouchard, V., 2001. Fringe wetlands of the Laurentian Great Lakes: effects of dikes, water level fluctuations, and climate change. Verhandlungen der Internationalen Vereingung fur Theoretische und Angewandte Limnologie 27, 3430–3437.

NOAA, 2006. *Great Lakes water levels*. National Oceanic and Atmospheric Administration, Great Lakes Environmental Research Laboratory. Web site accessed February, 2006; http://www.glerl.noaa.gov/data/now/wlevels/levels.html.

Payne, N.F., 1992. Techniques for wildlife habitat management of wetlands. McGraw-Hill, Inc, New York, New York.

Ponnamperuma, F.N., 1972. The chemistry of submerged soils. Adv. Agron. 24, 29–96.

Saltonstall, K., 2002. Cryptic invasion by a non-native genotype of the common reed, *Phragmites australis*, into North America. Popul. Biol. 99, 2445–2449.

Sherman, D.E., Kroll, R.W., Engle, T.L., 1996. Flora of a diked and an undiked southwestern Lake Erie wetland. Ohio J. Sci. 96, 4–8.

Smith, L.M., Kadlec, J.A., 1983. Seed banks and their role during drawdown of a North American Marsh. J. Appl. Ecol. 20, 673–684.

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

van der Valk, A.G., Davis, C.B., 1978. The role of seed banks in the vegetation dynamics of prairie glacial marshes. Ecology 59, 322–335.

Water Structures Unlimited, 2004. *AquaDam® User's Guide ~ 2004*. Carlotta, California: Water Structures Unlimited.

Wilcox, D.A., 2004. Implications of hydrologic variability on the succession of plants in Great Lakes wetlands. Aquat. Ecosyst. Health Manag. 7, 223–231.

Wilcox, D.A., and Kowalski, K.P. 1995. *Alternative management strategies for diked marshes, studies of Metzger Marsh plant communities*. Ohio Department of Natural Resources, Division of Wildlife, Crane Creek Wildlife Research Station, Metzger Marsh Steering Committee Final Report.

Wilcox, D.A., Nichols, S.J., 2008. The effects of water-level fluctuations on vegetation in a Lake Huron wetland. Wetlands 28, 487–501.

Wilcox, D.A., Whillans, T.H., 1999. Techniques for restoration of disturbed coastal wetlands of the Great Lakes. Wetlands 19, 835–857.