



Impacts of Riprap on Wetland Shorelines, Upper Winnebago Pool Lakes, Wisconsin

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Abstract Isolation of causative factors has proved challenging in characterizing the physical, chemical, and biological effects of shoreline hardening on the nearshore environment because of logistical challenges in identifying comparable sites. Extensive shoreline hardening and interspersions with unaltered shores in the large, shallow lakes in central Wisconsin provide an opportunity to surmount this. We compared the effects of shoreline protection on wave climate, bottom topography and substrate, water quality, and plant and animal assemblages at five adjacent pairs of natural and armored (riprapped) shorelines. Armored shorelines were characterized by coarser, more compacted substrates with lower organic content; cooler temperatures with higher dissolved oxygen; and greater water clarity. Differences in physical and chemical properties likely influenced plant growth forms and fish feeding guild differences between rippapped and natural sites. For example, floating-leaved plants were more abundant and fish were nearly twice as abundant and were represented by larger individuals at natural versus armored shorelines. Substrate characteristics may account for the differences in water quality and plant and animal associations observed in this study. As shoreline property owners continue to install riprap as protection against erosion, we expect the nearshore environment and associated biological communities to increasingly reflect this practice.

Keywords Fish · Macrophytes · Shore protection · Water quality · Wetlands

Introduction

Understanding the effects that shoreline protection activities may have on nearshore ecosystems has proved challenging. Impacts of shoreline protection result from complex and cascading interactions of exposure, slope, substrate, water depth, and aquatic vegetation (Duarte and Kalff 1986; Jennings et al. 1999; Hatzenbeler et al. 2000). For example substrate stability is influenced by stands of aquatic vegetation including their effect on wave energy, and substrate sediment, in turn, influences the light environment, distribution of nutrients, and suitability as a growth medium for aquatic macrophytes (Barko et al. 1991; Nichols 1992). Another difficulty is identifying appropriate unprotected sites in proximity for comparisons. This has been addressed by comparing sites among different lakes (Christensen et al. 1996; Jennings et al. 1999; Radomski and Goeman 2001), and by comparing developed and undeveloped sites in general terms (Bryan and Scarnecchia 1992; Leslie and Timmins 1993; Loughheed et al. 2001), which limit interpretation of results. Also, models may be inherent to a lake type, in that impacts to large, deepwater lake systems (Brazner and Magnuson 1994; Meadows et al. 2005; Goforth and Carman 2009) may differ from shallow lakes. Other studies have focused on specific habitat components, such as aquatic macrophytes (Radomski and Goeman 2001), nearshore cover (Christensen et al. 1996), substrate (Jennings et al. 1999), and fish (Hook et al. 2001), and variations in assessment methodologies and scale present uncertainty in integrating these to characterize ecosystem response. Although these studies generally indicate that shoreline protection projects can cause

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changes in the nearshore environment, including the lake bottom, the water column, and the biota (Brazner and Magnuson 1994; Goforth and Carman 2005; Trebitz et al. 2009), the challenge is to understand the potential magnitude and the complexity of interactions among these factors.

Shoreline protection has become prominent issue for the large, shallow lakes of central Wisconsin, especially those associated with Lake Winnebago. A Winnebago County shoreline erosion inventory found that 91% of the county's shoreline with adjacent wetlands needed shore protection (Winnebago County Land and Water Conservation Department [WCLWCD] 1998), and ongoing and widespread shoreline loss has instigated property owners and managers to protect existing shorelines. Riprap is the preferred and most prevalent shore protection within the Winnebago System, as well as throughout Wisconsin (Engel and Pederson 1998; Gabriel 2004). Protection has largely been accomplished by armoring eroding shorelines with riprap in the zone of water level fluctuation, and this has been implemented system-wide without regard for ecological effects. Therefore, the goal of this study was to ecologically characterize and compare the structural and functional aspects of armored and natural wetland shorelines in the Upper Pool Lakes of the Winnebago System. To accomplish this we compared habitats and plant and animal assemblages between pairs of adjacent natural shorelines and riprapped shorelines to assess the environmental effects that the addition of limestone riprap in the zone of water level fluctuation had while controlling for exposure to wind and waves, substrate type, and pre-existing vegetation. We hypothesized that greater amounts of wave energy would be reflected lakeward by armored shorelines causing sediment suspension and transport, which would reduce, in turn, water clarity and the light environment, photosynthesis, and distributions of organisms, as well as the substrate characteristics themselves.

Methods

Study Area

Located in east central Wisconsin, the Winnebago Pool Lakes, composed of Lake Winnebago (55,772 ha), Lake Butte des Morts (3,587 ha), Lake Winneconne (1,825 ha), and Lake Poygan (14,102 ha), drain 16,654 km² and comprise 17% of Wisconsin's surface water acreage, with a mean depth of 2.25 m and a maximum depth of 6 m (Fig. 1). Prior to damming of the system in the 1850s, the lakes were fertile riverine marshes with dense emergent vegetation (Linde 1975). Historically, the diversity and abundance of aquatic macrophytes was exceptionally high,

but by 1953 decline was noticeable as these marshes transformed into large, turbid, open-water lakes (Zimmerman 1953). Cattail-dominated shoreline wetlands (*Typha* spp.) decreased as higher lake levels eroded tens of thousands of hectares. Extensive stands of common reed (*Phragmites australis*) declined by as much as 94% between 1937 and 1997 (Gabriel and Bodensteiner 2002). Currently, the lakes are hypereutrophic and turbid in part due to agricultural and urban impacts (Lillie and Mason 1983). Erosion of exposed shorelines and re-suspension of fine sediments is caused by wave action and ice scour, further aggravated by the long fetches and shallow depths (Sloey and Spangler 1977).

Since 1982 lake levels have been maintained 1 m higher in the summer and 10 to 30 cm higher in the winter than previously, with a maximum seasonal range of 1.05 m. Under the current management strategy, the water level rapidly increases in the spring and summer, resulting in high lake levels during early plant growth, thereby shortening the growing season, adversely affecting propagation and reproduction, and increasing exposure to wave action (Kahl 1993). Water levels gradually decrease through the fall to achieve a winter drawdown to prevent ice damage to lakeshore properties and to moderate spring runoff (Krug 1981). Lower water levels in the fall expose more of the shoreline to storms and increase erosion (Kahl 1993). Drawdown also exposes aquatic vegetation to freezing temperatures and ice scour.

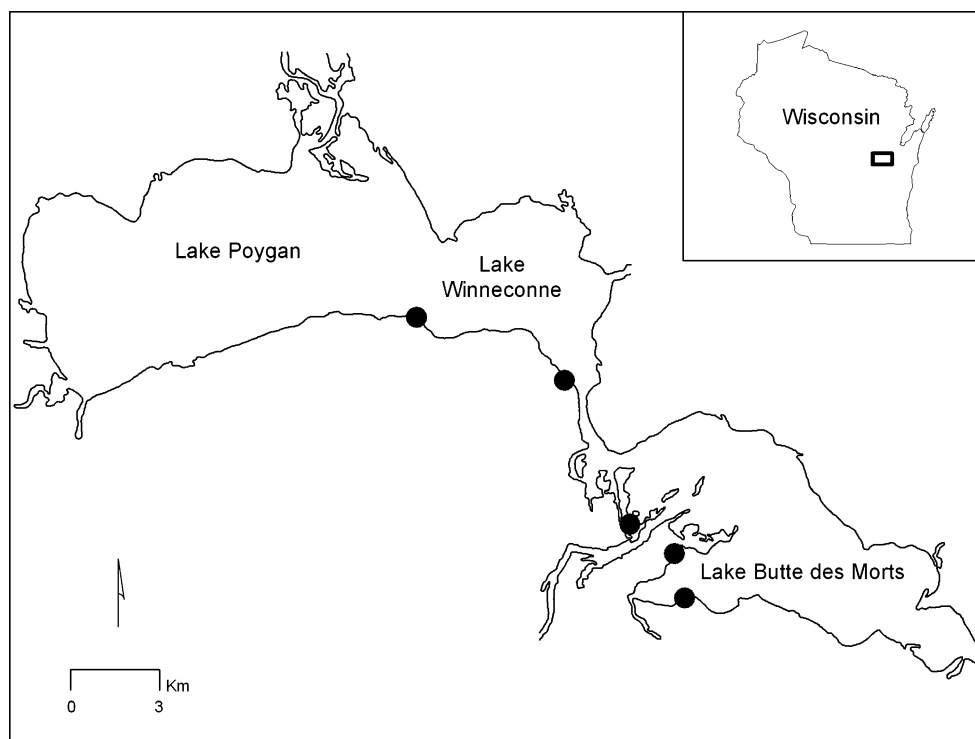
Our study compared five shoreline sites located on the Upper Winnebago Pool Lakes, including three sites on Lake Butte des Morts (Friedauer's Marsh, Fox River Hunt Club, and Reighmoor Marsh), one site on Lake Poygan (Lone Willow Marsh), and one site on Lake Winneconne (Wentzel Marsh; Fig. 1). Each site included a natural, unprotected broad-leaved cattail (*T. latifolia*)-dominated shoreline and a nearby shoreline armored with riprap. Although individual shoreline locations ranged from 250 to 600 m long, we used the entire length rather than a sub-section to avoid the potential influence of unidentified confounding variables, such as proximity of a sample location to the terminus of a shoreline type. These shorelines were identified by examining lake reports (e.g., Bohrer and Keil 1982; Wisconsin Department of Natural Resources [WDNR] 1989; WCLWCD 1998), aerial photographs and maps, conducting onsite visits, and conferring with WDNR staff.

Characteristics of Armored and Natural Shorelines

Bathymetry and Wave Climate

To assess changes in the slope of the bottom that may have been caused by armoring, we characterized bathymetry at each location at all sites. Water levels were measured at

Fig. 1 Location of paired armored and natural study sites, Winnebago Pool Lakes, Wisconsin



approximately 2.5 m intervals along three equally spaced 70-m long transects located perpendicular to and at the center of each location's shoreline; spacing of transects ranged from 75 to 250 m, depending on the lateral extent. Water depths were adjusted to a common level of 1.31 m relative to the corresponding elevation at the Lake Poygan gauge operated by the U.S. Army Corps of Engineers.

Wind speeds and wave heights were measured throughout the summer at a point 5 m from the shore along the study transects at each location to identify the impact the two shoreline types have on wave attenuation. Wind speed was determined 1 m above the water's surface with an anemometer. Waves were counted over a 2-min period, and the elevation (cm) of troughs and crests of 15 randomly selected waves were recorded using a 2 m high portable staff.

Substrate and Sedimentation

To characterize substrate composition, samples were collected at points 5 and 10 m from shore along the left and right transects at each site using a 10-L Peterson grab. Due to the fineness and softness of the substrate, the Peterson grab was effective at collecting samples of ample size, but occasionally sampling was repeated if the grab did not close. From each sample 80–100 ml were frozen for analysis of organic content. Another 250 ml of each sample were air-dried, crumbled, and then analyzed for particle size distribution using a sieve shaker with sieves of 0.063, 0.125, 0.250, 0.500, and 1.000 mm. Particles retained in each screen were weighed, from which the relative weight proportion was determined.

Organic content of substrate samples was determined by ashing. After drying to a constant weight at 60°C in an oven (DW), homogenized portions were weighed and placed into a muffle furnace at 550°C for two hours. Samples were reweighed to determine ash-free dry mass (AFDM). The proportion of organic matter to inorganic matter was reported as the ratio of AFDM to DW for each sample.

A sediment trap was installed 5 m from shore at the center transect and either the left or right transect at each location to assess net changes in sedimentation. Traps were constructed from cross-sections of 11.5-cm-diameter polyvinyl chloride pipe embedded 8–10 cm in a 24-cm-diameter piece of concrete; 8 cm of pipe was exposed facing the water's surface. Traps were deployed for 4 weeks until late August. Wet sediment was uniformly distributed in each trap, and the depth (mm) was measured.

Water Quality

To determine whether thermal conditions differed between the two types of shorelines, thermographs were used to record temperatures at 30-min intervals from early July through late August. Thermographs were placed 0.5 m below the surface 2 m from the shore on the center transect at each location. Water chemistry was measured 5 m from shore along transects used for bathymetry and vegetation assessments. Each pair of locations at each site was sampled 15 times throughout the summer at the same time and day, alternating between morning and afternoon. Two sets of measurements were taken at sunrise to evaluate the

effects of time of day on water chemistry. The vertical distributions of temperature and dissolved oxygen were measured at the surface, mid-depth, and bottom using a polarigraphic dissolved oxygen meter. Suspended sediment was assessed by measuring turbidity of water samples collected 0.5 m below the surface using a portable turbidimeter. Secchi depth (m) was determined to characterize water clarity. The pH was also measured at 0.5 m.

Aquatic Vegetation

Aquatic vegetation was assessed at each site in July, when both early- and late-growing plants are typically present and also when biomass is near maximum (Kahl 1993). Aquatic macrophytes were collected from a boat at 10-m intervals along the three 70-m-long transects using a 3-m long garden rake. Three rake samples and any additional species observed within 1.3 m of the boat were also collected and identified using Borman et al. (1997) and Eggers and Reed (1987). Water depths were measured at each interval with a calibrated PVC pole and standardized relative to the Army Corps of Engineers gauge.

Fish

Adult and juvenile fish were collected in mid-July using Indiana-style fyke nets and seines. Two fyke nets were set for two consecutive days at representative habitat at each natural and armored location, so that sampling effort was four trap-net days at each location and eight fyke-net days at each site. Fyke-netting gear and application conformed to WDNR research methods. Fish were identified according to Becker (1983), measured (total length in mm), and released.

To represent smaller fishes a 9-mm square mesh beach seine was deployed near the center of each location. The 1.2-X 15-m seine was stretched 10 m out from the shore and pulled parallel to the shore for 6 m in two consecutive efforts. Fish were identified and measured with the assistance of a dissecting microscope according to Becker (1983).

Statistical Methods

The principal goal of the design was to compare environmental conditions that may differentiate armored shorelines from natural shorelines. To address inherent differences in shorelines due to location around a lake, we examined pairs of armored and unarmored shorelines that were in proximity to each other at each site. Each pair of shoreline types composed a site. As a result, the availability of sites to select from was very constrained, which also meant that variables known to affect shoreline conditions, especially fetch and aspect, could not be controlled for among sites. Although multivariate analysis could be performed as a means of integrating the various factors

that we measured, we believe the limitations imposed by the small number of sites and uncontrolled influence of these other variables preclude a meaningful outcome. Therefore, our analysis was focused on individual comparisons of factors between pairs of armored and natural shorelines within a site and an assessment of the consistency of results among sites.

Data sets for each of the factors were tested for normality and homogeneity of variance. If these assumptions were met, then we performed pairwise comparisons using Student's *t*-test; if not, we used a non-parametric alternative, either a Wilcoxon's signed-rank test or a Mann–Whitney *U*-test.

To assess the relation of armoring to habitat conditions in terms of species, coefficients of similarity were determined for plants and for fish between the shoreline types within a site. We selected two: the Jaccard index and the Coefficient of Community (Mueller-Dumbois and Ellenberg 1974). The Jaccard index relies on presence-absence determination and is based on the proportion of species held in common among two groups to the total number of species present in those two groups. It was selected because it does not require information about abundance and provides more emphasis on species unique to each group versus alternatives such as the Sorensen index, which is weighted toward species in common. We also calculated the Coefficient of Community using frequency of occurrence to indicate the proportion of species held in common between two groups. Frequency of each aquatic macrophyte species was determined by dividing the number of intervals in which the species occurred by the total number of transect intervals at a location. For both coefficients values range from 0, indicating no similarity, to 100, indicating identical groups. To further assess effects of armoring based on proximity to the shore, comparisons were made of nearshore vegetation (0–30 m from shore) only and offshore vegetation (40–70 m from shore) only. A comparison of the similarity of nearshore to offshore vegetation across all locations was performed to provide context to comparisons based on shoreline types.

Results

Bathymetry and Wave Climate

Comparing locations within a site, water depths were deeper at two natural shorelines and one armored shoreline (two sample *t*-test, $p \leq 0.05$; Table 1). At the site with the deeper armored shoreline, the locations differed in that the natural shoreline occurred in a receding shallow embayment. Analysis of slope within 10 m of shore showed no significant differences between natural and armored locations. With a mean wind speed of 3.9 km/h that ranged from 2.3 to 13.5 km/h, mean wave height did not differ between armored and natural locations within a site (paired *t*-test, $p \leq 0.05$).

Table 1 Mean% slope and mean nearshore (5 m) and offshore (10 m) water depths (m) at armored and natural shorelines at sites in the Upper Winnebago Pool Lakes, Wisconsin.

Study site	Mean% slope ^a	Mean offshore depth (1 SD; cm) ^a	
		5 m	10 m
Wentzel Marsh			
Armored	7.8	103 (56)	140 (16)
Natural	4.5	57 (31)	79 (9)
Fox River Hunt Club			
Armored	7.4	74 (37)	99 (9)
Natural	3.9	64 (29)	88 (8)
Reighmoor Marsh			
Armored	3.2	54 (15)	67 (6)
Natural	5.2	64 (20)	71 (4)
Lone Willow Marsh			
Armored	5.1	49 (25)	73 (17)
Natural	1.4	72 (20)	89 (20)
Friedauer Marsh			
Armored	2.7	49 (15)	75 (9)
Natural	3.7	57 (20)	80 (13)

^a *P*-values ≤ 0.05 are highlighted in bold (depth: Student's two-sample *t*-test; slope: Wilcoxon *T* test)

Substrate and Sedimentation

Higher proportions of medium sand, very fine sand, and silt were found at natural locations within sites (Mann–Whitney *U* test, $p \leq 0.05$; Table 2). This difference was more evident 5 m from shore than at 10 m; proportions of very fine sand and silt differed at 5 m while only proportions of silt differed at 10 m (Mann–Whitney *U* test, $p \leq 0.05$). Natural locations also had higher proportions of organic content at both 5 m and 10 m from shore (Mann–Whitney *U*, $p \leq 0.05$). However, sedimentation rates did not differ between locations within sites (Mann–Whitney *U* test, $p > 0.05$; Table 2).

Water Quality

Surface water temperature was cooler at armored shoreline locations at most of the sites (paired *t*-test, $p \leq 0.05$). A Lone Willow Marsh thermograph malfunctioned, preventing comparison. These differences were primarily driven by significantly higher daily maximum temperatures and diel fluctuations in temperature (paired *t*-test, $p \leq 0.05$). Vertical profiles of temperature were similar between armored and natural locations (paired *t*-test, $p > 0.05$).

Dissolved oxygen concentration characteristically increased from a mean of 64% at dawn to 105% during mid-afternoon under clear skies, while extremes ranged

Table 2 Substrate characteristics of armored and natural shorelines in the Upper Winnebago Pool Lakes, Wisconsin

Substrate Characteristic	<i>N</i> _A , <i>N</i> _N	Median (IQR) ^{a,b}	
		Armored (A)	Natural (N)
Size distribution (% dry mass)			
5 m from shore			
Medium sand	10, 10	12.9 (13.2)	3.9 (9.0)
Fine Sand	10, 10	67.6 (39.8)	30.6 (31.1)
Very fine sand	10, 10	18.2 (15.3)	32.2 (21.8)
Silt	10, 10	2.7 (4.5)	21.9 (24.3)
10 m from shore			
Medium sand	10, 10	15.6 (11.4)	4.8 (14.5)
Fine Sand	10, 10	57.3 (46.6)	42.9 (31.9)
Very fine sand	10, 10	18.5 (13.1)	30.5 (13.3)
Silt	10, 10	2.2 (7.5)	19.4 (26.5)
Combined locations			
Medium sand	20, 20	13.3 (15.6)	4.1 (12.8)
Fine Sand	20, 20	64.8 (44.8)	36.3 (29.2)
Very fine sand	20, 20	18.3 (13.3)	31.6 (21.9)
Silt	20, 20	2.4 (3.5)	19.4 (26.4)
Organic content (% dry mass)			
5 m from shore			
	10, 11	1.3 (4.9)	11.2 (36.0)
10 m from shore			
	10, 9	1.0 (10.7)	19.6 (25.2)
Combined locations			
	20, 20	1.0 (6.5)	18.0 (30.9)
Accumulation depth (mm)			
	10, 10	20.5 (21.3)	39.0 (33.3)

^ainterquartile range

^b*P*-values ≤ 0.05 are highlighted in bold (Mann–Whitney *U*-test)

from 30% to 145%. The timing of dissolved oxygen peaks corresponded to peaks in surface temperature and to maximum differences between surface and bottom temperatures. Dissolved oxygen relative to saturation was 6% higher at armored than natural locations overall (two-sample *t*-test, $p \leq 0.05$), and 5.7–15.6% higher at the armored locations of all the sites except the Fox River Hunt Club (paired *t*-test, $p \leq 0.05$; Table 3). Differences in vertical profiles of dissolved oxygen relative to saturation were only evident at Reighmoor Marsh where the difference at the natural location was greater than at the armored location (paired *t*-test, $p \leq 0.05$).

Overall, pH measurements ranged between 7.7 and 9.7 throughout the summer at the five sites. At three sites pH differed between locations (Wilcoxon signed ranks test, $p \leq 0.05$) (Table 3). The peaks in pH corresponded with peaks in dissolved oxygen.

Secchi depths indicated that armored sites were less turbid overall and at two of the sites (two-sample *t*-test, $p \leq 0.05$; Table 3). However, turbidity at 0.5 m depth was only less at the armored location at the Fox River Hunt Club site (paired *t*-test, $p < 0.05$; Table 3).

Aquatic Macrophytes

A total of 22 aquatic macrophyte species were found in the nearshore zone of the armored and natural sites, including six emergent, five floating-leaved, and 11 submersed species with associated water depths ranging between 0.15–1.50 m (Table 4). Hardstem bulrush (*Scirpus acutus*), spatterdock (*Nuphar lutea*), coontail (*Ceratophyllum demersum*), sago pondweed (*Potamogeton pectinatus*), and wild celery (*Vallisneria americana*) had the greatest proportion of areal coverage among all sites. At armored locations overall species richness was 20 species with a mean of 7.8 species per location (Table 5). Natural locations had a combined species total of 17 with the mean number per location being 9.2. With respect to the most abundant species, armored sites tended to have more narrow-leaved cattail (*Typha angustifolia*), spatterdock, coontail, and sago pondweed, while natural sites had more common waterweed (*Elodea canadensis*) (Table 4). Jaccard similarity (JS) coefficients and coefficients of community (CC) reflect the variations in species composition between armored and natural locations among sites (Table 5). Species composition was most similar at the Fox River Hunt Club (JS 66.6%; CC 76%) and least similar at the Friedauer (JS 28.6%; CC 12.4%) and Wentzel Marsh (JS 30.8%; CC 23.7%).

Although the areal coverage, species richness, and dominance of vegetation differed between armored and natural locations at all sites combined, no pattern was evident among individual sites. For example, areal coverage by wild celery

Table 3 Mean water quality measures at armored and natural shorelines during July–August in the Upper Winnebago Pool Lakes, Wisconsin

Site	Water quality parameters: mean (1 SD) ^a									
	Dissolved oxygen (DO)% saturation		% DO saturation: surface-bottom		pH		Secchi depth (cm)		Turbidity (NTU)	
	Armored	Natural	Armored	Natural	Armored	Natural	Armored	Natural	Armored	Natural
All sites	87.5 (25.7)	79.5 (23.7)					45 (14)	39 (14)	13.8 (8.8)	23.8 (16.0)
Fox River Hunt Club							52 (15)	35 (10)	13.8 (8.8)	23.8 (16.0)
Friedauer Marsh	96.1 (24.9)	90.4 (22.7)								
Lone Willow Marsh	85.2 (30.0)	69.6 (21.1)			8.5 (0.5)	8.2 (0.6)				
Reighmoor Marsh	94.6 (24.9)	87.5 (22.0)	0.9 (5.8)	6.3 (6.3)	8.8 (0.4)	8.6 (0.6)				
Wentzel Marsh	88.3 (19.7)	73.4 (20.1)			8.6 (0.6)	8.3 (0.6)	52 (14)	44 (12)		

^a *P*-values ≤ 0.05 are highlighted in bold (paired *t*-test)

Table 4 Water depths and proportional coverage by aquatic macrophytes at armored and natural sites in the Upper Winnebago Pool Lakes, Wisconsin

Species by vegetation type	N	Water depth (cm): mean (range)	% Coverage: mean (range)	
			Armored	Natural
No vegetation	104	81 (15–110)	52.5 (8.3–83.3)	35.0 (4.2–79.2)
Emergent				
Hardstem bulrush (<i>Scirpus acutus</i>)	19	63 (20–90)	10.8 (0–54.2)	5.0 (0–16.7)
Narrow-leaved cattail (<i>Typha angustifolia</i>)	13	48 (15–80)	3.3 (0–12.5)	10.0 (8.3–12.5)
Softstem bulrush (<i>Scirpus validus</i>)	6	58 (50–70)	4.2 (0–4.2)	0.8 (0–16.7)
Pickereel weed (<i>Pontederia cordata</i>)	1	60 (–)	1.7 (0–8.3)	0 (0–0)
Wild rice (<i>Zizania palustris</i>)	2	72 (65–80)	0 (0–0)	1.7 (0–8.3)
Broad-leaved cattail (<i>Typha latifolia</i>)	1	80 (–)	0 (0–0)	0.8 (0–4.2)
Floating-leaved				
Spatterdock (<i>Nuphar lutea</i>)	29	67 (20–90)	9.2 (0–33.3)	14.2 (0–37.5)
Long-leaf pondweed (<i>Potamogeton nodosus</i>)	7	64 (30–85)	3.3 (0–12.5)	2.5 (0–8.3)
Duckweed (<i>Lemna</i> spp.)	2	32 (30–35)	1.7 (0–8.3)	0 (0–0)
White water lily (<i>Nymphaea odorata</i>)	1	80 (–)	0 (0–0)	0.8 (0–4.2)
Submersed				
Coontail (<i>Ceratophyllum demersum</i>)	71	71 (20–125)	18.3 (4.2–41.7)	30.9 (4.2–79.2)
Wild celery (<i>Vallisneria americana</i>)	65	74 (20–100)	20.8 (0–58.3)	24.3 (0–70.8)
Sago pondweed (<i>Potamogeton pectinatus</i>)	54	63 (20–110)	11.8 (4.2–25.0)	24.2 (0–62.5)
Common waterweed (<i>Elodea canadensis</i>)	58	73 (30–150)	18.3 (8.3–45.8)	12 (0–33.3)
Flatstem pondweed (<i>Potamogeton zosteriformis</i>)	12	88 (70–125)	3.3 (0–12.5)	6.7 (0–16.7)
Bushy pondweed (<i>Najas flexilis</i>)	4	74 (50–90)	2.5 (0–12.5)	0.8 (0–4.2)
Curly-leaf pondweed (<i>Potamogeton crispus</i>)	9	64 (30–80)	1.7 (0–4.2)	5.8 (0–29.2)
Northern water milfoil (<i>Myriophyllum</i> spp.)	8	72 (30–100)	5.0 (0–20.8)	1.7 (0–8.3)
Eurasian water milfoil (<i>Myriophyllum spicatum</i>)	7	63 (40–85)	3.3 (0–16.7)	2.5 (0–12.5)
Water stargrass (<i>Heteranthera dubia</i>)	4	74 (55–95)	1.7 (0–8.3)	1.7 (0–8.3)
Leafy pondweed (<i>Potamogeton foliosus</i>)	4	82 (75–85)	0 (0–0)	3.3 (0–12.5)

dominated armored sites at the Fox River Hunt Club and Lone Willow Marsh, but it was more dominant at natural sites at

Reighmoor and Wentzel Marshes. Similarly, coontail dominated the armored location at the Fox River Hunt Club, while

Table 5 Vegetation distribution similarities among shoreline types and locations, including overall similarity between nearshore habitat and offshore habitat, in the Upper Winnebago Pool Lakes, Wisconsin

Site	N of aquatic macrophyte species		Habitat similarity index values							
			Armored vs. natural shorelines						All shorelines	
			Nearshore and Offshore		Only Nearshore		Only Offshore		Nearshore vs. offshore	
			Armored	Natural	JS ^a	CC ^b	JS	CC	JS	CC
Fox River Hunt Club	11	9	66.6	76	54.5	51	75	66.6	53.8	67.4
Friedauer Marsh	6	4	28.6	23.7	28.6	23.7	NA	NA	56.3	66.1
Lone Willow Marsh	11	13	41.2	54.3	58.3	61.6	28.6	34.6	40	53.1
Reighmoor Marsh	7	7	40	61.3	40	60.6	75	49.9	61.5	69.1
Wentzel Marsh	4	13	30.8	12.4	9.1	12.1	30	12.4	0	0

^a Jaccard similarity coefficient

^b Coefficient of community

it was at much higher proportions at natural sites at both Lone Willow Marsh and Wentzel Marsh.

Much of the nearshore also tended to be bare, with proportions of bare samples ranging between 35–52% at armored and natural locations (Table 4). Within sites higher proportions of bare substrate were present at three natural locations and two armored locations. The highest proportions of bare substrate were found at both the armored and natural locations of Friedauer Marsh (75–79.2%), while the location with the lowest proportion of bare substrate was the natural shore at Wentzel Marsh with 4.2%; the site with the lowest proportions of unvegetated substrate was Lone Willow Marsh with 8.3% at the armored location and 16.7% at the natural location.

As the impacts of shore protection on vegetation would be expected to be more evident closer to the shoreline, this study also compared the species composition between the nearshore (0–30 m) and offshore (40–70 m) habitats of armored and natural locations at each site (Table 5). Jaccard similarity coefficients and coefficients of community differed more between nearshore habitat than offshore habitat between locations at four sites but not at Lone Willow Marsh. The least similarity in nearshore vegetation between armored and natural locations was at the Wentzel Marsh site (JS - 9.1%; CC -12.1%), while the greatest similarity in nearshore vegetation was at the Fox River Hunt Club and Lone Willow Marsh sites (JS 54.4–58.3%; CC 51.0–61.6%). Offshore vegetation differed the most between armored and natural locations at the Lone Willow Marsh and Wentzel Marsh sites (JS 28.6–30.0%; CC 2.4–34.6%). The species composition of offshore vegetation was more similar at the Fox River Hunt Club and Reighmoor Marsh sites, with Jaccard similarity coefficients for each of 75% and coefficients of community of 49.9 and 66.6%. When comparing similarity coefficients between neighboring offshore and combined nearshore-offshore locations, three sites had greater similarity between offshore distributions; comparing similarity coefficients between combined nearshore-offshore and armored-natural distributions, two sites had greater similarity between combined nearshore-offshore distributions.

Fish Communities

Of the 28 fish species captured by fyke netting at the five sites, 23 were collected at the armored locations and 24 at the natural locations (Table 6). Among the nine species only found at one of the two shoreline types, white suckers (*Catostomus commersoni*), fathead minnows (*Pimephales promela*), yellow bass (*Morone mississippiensis*), and smallmouth bass (*Micropterus dolomieu*) were unique to natural sites, and smallmouth buffalo (*Ictiobus bubalus*), quillback (*Carpionodes cyprinus*), largemouth bass (*Micropterus salmoides*), and rock bass (*Ambloplites rupestris*) were only found at armored

sites. More than twice as many fish were captured along the natural shorelines than along the armored shorelines using equal efforts of fyke netting. Overall, at the natural sites the species composing more than 10% of the catch were longnose gar (*Lepisosteus osseus*), gizzard shad (*Dorosoma cepedianum*), white bass (*Morone chrysops*), and freshwater drum (*Aplodinotus grunniens*). At the armored sites the most abundant species were shortnose gar (*Lepisosteus platostomus*), white bass, bluegill (*Lepomis macrochirus*), and freshwater drum.

To compare the most abundant fishes between locations that were captured by fyke netting, the species at each site were selected through rank ordering until the total cumulative proportion was at least 50%. Based on this assessment method, species composition differed between natural and armored sites (Table 7). Species that occurred among the most abundant fishes multiple times at natural sites include freshwater drum, white bass, channel catfish (*Ictalurus punctatus*), and gizzard shad. At armored sites the most abundant fishes were shortnose gar, longnose gar, and bluegill.

Seining captured 17 fish species at the five pairs of natural and armored sites with 11 species common to both sites (Table 6). Overall at natural locations, 95% of the total catch of 840 was composed of shiners (*Notropis* spp.), gizzard shad, and bluegill. At the armored locations these same three groups composed more than 98% of the total catch of 1,243 fishes. Any potential differences between shoreline types were likely masked by the dominant numbers of emerald shiners that composed more than 50% of the catch in both habitat types.

Combining the two capture techniques, 34 species were collected with 23 species common to both shoreline types. However, within those species that were unique to either type of shoreline, numbers of individuals were six or less, composing 0.3% or less of a sample, so the absence of these species from a particular site may be an artifact of low overall abundance or low susceptibility to capture. Generally, the fyke netting captured larger individuals and larger species, and seining captured the small individuals and smaller species. The total catch was similar between natural and armored sites with the biggest difference being the greater number of large fishes captured from the natural sites versus more small fish taken from the armored sites, largely composed of shiners.

Jaccard similarity (JS) coefficients and coefficients of community (CC) reflect the variations in species composition between armored and natural locations among sites (Table 8). Species composition of larger fish caught by fyke netting was fairly similar between the various sites (JS 47.4–69.6%; CC 24.1–59.7%). There was a greater variation in the distribution of smaller fish caught by seining, with most similar distributions found at Friedauer Marsh, Lone Willow Marsh, and Reighmoor Marsh (JS 54.5–87.5%; CC 11.7–58.1%) and least similar at the Fox River Hunt Club and Wentzel Marsh (JS 12.5–16.7%; CC 7.4–47.3%).

Table 6 Numbers of fish captured from armored (A) and natural (N) shorelines by equal effort among sites with minnow seines and fyke nets in the Upper Winnebago Pool Lakes, Wisconsin

Species by trophic guild	Seine		Fyke net		Total	
	A	N	A	N	A	N
Piscivore						
White bass (<i>Morone chrysops</i>)	3	23	58	115	61	138
Longnose gar (<i>Lepisosteus osseus</i>)	0	0	37	110	37	110
Shortnose gar (<i>Lepisosteus platostomus</i>)	0	0	70	54	70	54
Channel catfish (<i>Ictalurus punctatus</i>)	1	3	24	63	25	66
Black crappie (<i>Pomoxis nigromaculatus</i>)	9	4	31	25	40	29
Bowfin (<i>Amia calva</i>)	0	0	3	19	3	19
Walleye (<i>Stizostedion vitreum</i>)	0	0	2	15	2	15
Northern pike (<i>Esox lucius</i>)	0	0	7	5	7	5
Flathead catfish (<i>Pylodictis olivaris</i>)	0	0	2	4	2	4
Largemouth bass (<i>Micropterus salmoides</i>)	0	1	0	5	0	6
Yellow bass (<i>Morone mississippiensis</i>)	2	0	2	0	4	0
Smallmouth bass (<i>Micropterus dolomieu</i>)	0	0	1	0	1	0
Rock bass (<i>Ambloplites rupestris</i>)	0	0	0	1	0	1
Insectivore						
Shiners (<i>Notropis</i> spp.)	874	443	17	12	891	455
Bluegill (<i>Lepomis macrochirus</i>)	142	204	57	42	199	246
Northern redhorse (<i>Moxostoma macrolepidotum</i>)	1	0	23	59	24	59
Pumpkinseed (<i>Lepomis gibbosus</i>)	0	0	10	12	10	12
Yellow perch (<i>Perca flavescens</i>)	3	3	6	6	9	9
Logperch (<i>Percina caprodes</i>)	3	4	0	0	3	4
Smallmouth buffalo (<i>Ictiobus bubalus</i>)	0	0	0	2	0	2
Yellow bullhead (<i>Ameiurus natalis</i>)	0	0	1	2	1	2
Omnivore						
Gizzard shad (<i>Dorosoma cepedianum</i>)	205	152	22	213	227	365
Common carp (<i>Cyprinus carpio</i>)	0	0	15	62	15	62
Quillback (<i>Carpoides cyprinus</i>)	0	0	0	2	0	2
White sucker (<i>Catostomus commersoni</i>)	0	0	1	0	1	0
Fathead minnow (<i>Pimephales promelas</i>)	0	0	1	0	1	0
Invertivore						
Freshwater drum (<i>Aplodinotus grunniens</i>)	0	3	61	91	61	94
Totals	1243	840	451	919	1694	1759

Discussion

Substrate and Sedimentation

Nearshore habitats appear to be affected by shoreline protection on the Upper Winnebago Pool Lakes, especially in terms of substrate-related changes. As also found by Jennings et al. (1999), substrate characteristics differed between riprap and natural shorelines, with higher proportions of fine and organic material found along natural shorelines. Although wave heights did not differ during the relatively calm summer, the difference in sediment sizes and organic content may be indicative of differences in wave climates during windier seasons, when lower wave energies permit greater deposition of smaller, lighter sediment along natural

shorelines, while coarser sediments result from higher wave energies. Over time, one might also expect to find nearshore water depths to be greater at armored rather than natural sites, as erosional scouring of the nearshore results from increased wave activity and turbulence due to increased wave reflection from a hardened shoreline (Kraus 1988; Griggs 2005). However, unlike Jennings et al. (1999), we found deeper water depths more often at natural rather than armored shorelines, but shoreline slope was the same.

The difference in sediment sizes may also reflect differences in sediment supply between a stabilized, armored shoreline versus an unstable, eroding shoreline. Armored shorelines are cut off from a continuous, on-site supply of fine, organic sediment due to shoreline hardening. Over time, the finer sediment may be removed by wave action, while the

Table 7 Abundance of fishes at the five armored and five natural shoreline sites in the Upper Winnebago Pool Lakes, Wisconsin

Species	Number of sites demonstrating abundance ^a	
	Armored	Natural
Freshwater drum	1	3
White bass	1	3
Channel catfish	1	2
Gizzard shad	0	2
Common carp	0	1
Northern redhorse	0	1
Black crappie	1	0
Bluegill	2	0
Longnose gar	3	1
Shortnose gar	4	0

^a Fishes captured by fyke nets were ordered by abundance at each site and added to the pool until the proportion equalled or exceeded 50%, resulting in one to four species at each location

organic matter is broken down through biochemical activity. By comparison, natural, eroding shorelines have a continuous supply of fine, organically-rich sediment to replace the lost sediment and organics. The greater accumulation rate of sediment at natural shorelines during summer was likely due to higher proportions of fine and organic material, which is more easily suspended and transported.

Water Quality

Differences in water quality parameters between adjacent shoreline types may also be linked to differences in substrates and related circulation patterns. Based on water temperature differences, several armored shorelines exhibited greater circulation, as reflected by slightly cooler mean temperatures and a higher percent of saturation of dissolved oxygen. These conditions could be linked to changes in nearshore circulation patterns caused by differences in wave reflection and absorption, which may directly affect water circulation,

creating greater mixing and dispersion of warmer surface water at armored sites. Water clarity, based on Secchi depths, was also greater at armored sites, which lack the more mobile, finer, higher organic substrate found at natural sites; the higher proportion of organic matter may also cause higher uptake of dissolved oxygen due to higher biochemical oxygen demands. Higher water clarity may also have contributed to the higher pH levels at armored sites, by promoting primary productivity by phytoplankton, filamentous algae, and aquatic macrophytes, resulting in greater removal of dissolved carbon dioxide, especially during warm, sunny weather.

Aquatic Macrophytes

Although the amount and type of aquatic macrophytes differed between armored and natural locations, the overall patterns were inconsistent, and not directly linked to the presence of shoreline protection. Cover tended to be less at armored locations, likely due to the coarser substrate containing less organic matter, very fine sand, and silt. Species composition varied greatly between natural and armored locations at all the sites, with greater similarities evident farther from shore at the majority of sites, based on similarity coefficients. Of the more spatially dominant species, coontail, sago pondweed, and spatterdock were more prevalent along natural shorelines, while hardstem bulrush was more evident along armored shorelines, though this was largely driven by a disproportionate amount contributed by a single site. The difference in dominant species may also be associated with differences in plant morphologies. For example, hardstem bulrush has more flexible culms, allowing it to persist in higher wave energy environments (Bonham 1983). Jennings et al. (1999) found sites without shoreline structures tended to have more areal coverage by floating-leaved macrophytes, while armored shorelines had less emergent and more submersed species cover. Similarly, this study found greater abundance and species richness in submersed than emergent species along armored locations, while spatterdock was more common

Table 8 Fish distribution similarities among shoreline types and locations in the Upper Winnebago Pool Lakes, Wisconsin

Site	N of fish species		Habitat similarity index values			
			Armored vs. natural shorelines			
			Fyke Net		Seine	
	Armored	Natural	JS ^a	CC ^b	JS	CC
Fox River Hunt Club	10	17	62.5	37.3	16.7	47.3
Friedauer Marsh	6	16	50	24.1	87.5	26.7
Lone Willow Marsh	13	16	47.4	59.7	66.7	11.7
Reighmoor Marsh	16	20	50	25.6	54.5	58.1
Wentzel Marsh	18	23	69.6	40.6	12.5	7.4

^a Jaccard similarity coefficient

^b Coefficient of community

along natural shorelines, likely due to finer substrates and possibly indicative of calmer wave conditions.

The unstable unconsolidated substrate prevalent at natural shorelines may be less conducive for the establishment and survival of other vegetation species (Barko et al. 1991), as this instability may inhibit rooting, bury plants, and contribute to the lower water clarity found at natural shorelines, which further reduces the already limited photic zone characteristic of the hypereutrophic Winnebago Pool Lakes. The impact of this should be greatest along deeper, natural shorelines, evidenced by different community types and increased bare areas. An example of this is found at the Fox River Hunt Club site, which had significantly lower turbidity at the deeper natural shoreline coinciding with less submersed vegetation cover and a greater proportion of bare areas.

Fish

Natural shorelines had a different fish assemblage than armored shorelines, likely due to the overall differences in habitat conditions as a result of the controlling variables such as substrate, water quality, and aquatic macrophytes (Tonn and Magnuson 1982; Chick and McIvor 1994). Of the overall total species richness of 34 species, at least four-fifths were captured at a given shoreline type, indicating that most fishes are not avoiding a particular set of habitat conditions. However, the much higher numbers of fish within species at natural sites in comparison to armored sites suggests that they are demonstrating a preference. Bryan and Scarnecchia (1992) and Brazner and Magnuson (1994) have both recorded greater abundance at natural than developed shorelines, though they also found greater species diversity as well. In contrast, in comparing shorelines with vertical bulkheads, riprap, and no structures, Jennings et al. (1999) found greater fish diversity at riprapped sites, though their results are confounded by compromised habitat conditions and incomplete sampling at no structure sites.

Armored and natural shorelines also differ in fish assemblages based on ecological roles. Generally, natural shorelines had more large piscivores, omnivores and bottom-feeding insectivores, while armored shorelines exhibited an abundance of small forage fishes, primarily shiners, perhaps responding to the complex micro-habitat created by the interstices of the riprap. Similarly, Trial et al. (2001) found a higher abundance of smaller prey fishes along shorelines with riprap, in contrast to higher numbers of insectivores and ambush predators such as largemouth bass associated with more vegetated shorelines. Conversely, Jennings et al. (1999) counted more prey fishes (cyprinids) along shorelines without structures, and they caught more benthics, centrarchids, and other panfish along riprapped shorelines. The young-of-the-year fishes showed no difference between shoreline types. Fish abundance in shallow water did not

differ between developed and undeveloped shoreline types, but fewer kinds of nearshore fishes were found associated with unstable substrates (Goforth and Carman 2005). Larval fishes were not directly affected by human activities along the shoreline but were indirectly affected by water temperature and slope of the bottom (Hook et al. 2001).

Conclusion

The ability to detect habitat responses to shoreline development through comparison with undeveloped shoreline habitat is obfuscated by inherent differences in chemistry and morphometry, ecological response times, and baseline conditions because of the challenge in separating the effects caused by humans and those caused by concomitant changes to other habitat variables such as water quality (Jennings et al. 1999). This challenge appears to be present at scales of comparison ranging from adjacent locations within a lake to those among lakes. Assessing stress-response relationships is further complicated by accommodating seasonal and annual changes in nearshore habitat components such as fish assemblages and aquatic macrophytes (Leslie and Timmins 1993; Trial et al. 2001). Other direct site-scale effects on nearshore substrate may stem from unrelated updrift conditions, which may arise in larger lake systems (Meadows et al. 2005; Goforth and Carman 2009). Despite the challenges, through the comparison of neighboring armored and natural shorelines, we found that shoreline protection affected a variety of nearshore habitat components in the Upper Winnebago Pool Lakes. These included changes to substrate that can be linked to changes in bathymetry, water quality, aquatic macrophyte, and fish assemblages.

Evidence of other effects of shoreline protection is equivocal, likely due to the continued impacts of lake level manipulation. Drawdown exposure zones in controlled lakes are typified by low diversity and abundance of macrophytes (Quennerstedt 1958; Wagner and Falter 2002) as well as loss of fine sediments and organic matter (Furey et al. 2004). Impacts tend to be greatest on submerged vegetation, leading to reduced cover and diversity with artificially higher fluctuating water levels (Quennerstedt 1958; Richardson et al. 2002; Wagner and Falter 2002). The increased turbidity due to loss of vegetative cover may also induce further changes to species composition (Chambers and Kalff 1985; Nichols 1992). However, as with shoreline protection, impacts of water level manipulation on aquatic macrophytes reported in the literature has been highly varied, likely reflecting the large variability in the magnitude and season of drawdowns, size, morphology, trophic states of the lakes examined, and the wide variety of aquatic community types with differing levels of vulnerability to the effects of freezing, ice scour, or drying out resulting from nearshore

exposure (Turner et al. 2005). Several studies have also considered variations in within-lake factors, including differing impacts by water depth (e.g. Wilcox and Meeker 1991) as well as a combination of different exposures, slopes, substrates, and water depths (e.g., Hellsten and Riihimaki 1996).

The combined effects of shoreline protection and water level manipulation are further compounded by numerous other stresses found in the Upper Winnebago Pool Lakes, including impacts by winter ice scour, summer boat wakes, bioturbation by common carp and gizzard shad, as well as extremely high turbidity caused by sediment influx and resuspension and algal blooms resulting from external nutrient loading (Sloey and Spangler 1977; WDNR 1989; Kahl 1993). The interactions of these additional stresses reduce diversity and areal coverage by aquatic macrophytes, contributing to the high proportion of bare nearshore fronting both armored and natural shorelines. These conditions, in turn, influence water quality and the abundance and diversity of animals that will inhabit these locations.

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