Ecological consequences of shoreline armoring on littoral fish and benthic macroinvertebrate communities in an Eastern Ontario lake

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Abstract

Shoreline erosion in lakes is a pressing issue for many landowners, yet common erosion mitigation practices that involve armoring can alter littoral habitat and potentially diminish near-shore biodiversity. We studied the effects of two armoring methods (i.e., riprap, retaining walls) on habitat, taxonomic richness, relative abundance, and total abundance of fishes and benthic macroinvertebrates at shorelines on Big Rideau Lake in eastern Ontario, Canada. Snorkel surveys were conducted to assess aquatic habitat characteristics and fish diversity, and benthic infauna were sampled using kick-nets. Submergent and emergent macrophytes were more abundant at natural rocky shorelines compared to shorelines modified with riprap or retaining walls. Coarse woody debris was also more abundant at natural shorelines compared to riprap and retaining wall shorelines. Relative abundances of some fish species varied between shoreline types, but overall species richness and total abundance did not. Largemouth Bass (*Micropterus salmoides*) were more abundant at natural sites than armored sites. Conversely, Rock Bass (*Ambloplites rupestris*), Pumpkinseed (*Lepomis gibbosus*), Bluegill (*Lepomis macrochirus*), Yellow Perch (*Perca flavescens*), and baitfish from the family Cyprinidae were more abundant at both types of armored sites compared to natural sites. Taxonomic richness of benthic macroinvertebrates did not vary among shoreline types, however abundance of Amphipoda, Isopoda, Ephemeroptera, and Cladocera was greater at armored shorelines. These results suggest that human modification of shorelines is altering littoral ecosystems and potentially leading to shifts in the community structure of littoral nekton. More study is needed to fully understand the community level effects of shoreline erosion mitigation involving armoring in freshwater lakes and determine the effectiveness of alternative mitigation strategies that preserve natural habitat features.

Keywords Aquatic habitat · Erosion mitigation · Riprap · Retaining walls · Shoreline

Introduction

Anthropogenic alteration of natural ecosystems is arguably the greatest challenge facing biodiversity today (Vitousek et al. 1997). Historically, shorelines have been popular sites for human settlement with roughly one-third of the human population currently living within close proximity of a shoreline and the proportion expected to increase to one-half by 2030 (Small and Nicholls 2003; Liddle and Scorgie 1980; MEA 2005). This has resulted in the alteration of many shorelines for recreation (beaches, docks, marinas), industrial development (docking facilities), or for erosion mitigation (Arkema et al. 2013; Gittman et al. 2015). The latter is often accomplished through shoreline armoring, defined as the implementation of engineered shore structures to reduce the erosional effects of wave action, storms, or flooding (Gittman et al. 2015). However, armoring

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shorelines has been shown to cause numerous ecological detriments including reducing accumulation of washed-up organic matter (Wensink and Tiegs 2016), increased local turbidity (Bozek and Burdick 2005), and reduction in the ability to moderate pollutant runoff (NRC 2007). Armored shorelines can also cause habitat fragmentation or loss of habitat by creating barriers between the aquatic, riparian, and terrestrial realms (Peterson and Lowe 2009; Dugan et al. 2011). Currently, habitat degradation is the leading driver of extinction (Pimm and Raven 2000; Brooks et al. 2002; Giam et al. 2010; Pimm and Askins 1995) and as erosion mitigation strategies continue to be implemented worldwide, research is needed to examine their effects on aquatic biodiversity and guide developers towards adopting alternative strategies that maintain or enhance biodiversity.

Despite only occupying less than 1% of the earth's surface, freshwater ecosystems support approximately one-third of all vertebrate biodiversity and 10% of total biodiversity (Strayer and Dudgeon, 2010). In lentic ecosystems, the near-shore habitat accounts for a small fraction of the total lake area yet the overwhelming majority of lentic organisms frequent or permanently inhabit it (Vadeboncoeur et al. 2011). The near-shore area, commonly defined as the littoral zone in lakes, serves as critically important habitat for a variety of aquatic species by providing refuge from predators, overwintering sites, and spawning grounds (Hall and Werner 1977). Additionally, the vast majority of freshwater aquatic macrophyte diversity is found in the littoral zone (James et al. 1997). Freshwater ecosystems are among the most threatened, with biodiversity loss in freshwater ecosystems occurring at triple the rate of that in terrestrial or marine ecosystems (Tickner et al. 2020). As a result, destruction of shoreline habitat is one of the most pressing issues facing ecosystem management in lakes worldwide (Bryan and Scarnecchia 1992; Schmieder 2004; Vadeboncoeur et al. 2011).

In both freshwater and coastal systems, two similar methods of erosion mitigation are commonly used. Riprap revetments consist of shore-parallel, sloped barriers of unconsolidated rock or rubble (Quigley and Harper 2004; Gittman et al. 2015), are relatively cost-effective and require minimal excavation of shoreline substrate to construct. Conversely, retaining walls are fully consolidated barriers of concrete or stone perpendicular to the water line (Gittman et al. 2015) and implementation typically requires more excavation. Use of both of these shoreline armoring methods has the potential to disrupt riparian characteristics by either adding new substrate, removing existing substrate, or altering shoreline slope and composition. The ecology of the littoral zone is greatly influenced by these riparian characteristics (shoreline slope, substrate permeability), and alterations that affect runoff dynamics can lead to eutrophication of the littoral zone (Mallin et al. 2000; Dillon et al. 1994; Garrison et al. 2010). The implementation of armored erosion mitigation

techniques can cause a variety of physical changes to existing habitat such as reducing the input of coarse woody debris (Christensen et al. 1996), reducing aquatic macrophyte stands (Radomski and Goeman 2001; Jennings et al. 2003), and decreasing littoral habitat complexity (Schmude et al. 1998). Removal of riparian vegetation and coarse woody debris (CWD) during construction can reduce aquatic nekton communities as it provides important habitat for a variety of species (Harmon et al. 1986; Helmus and Sass 2008) and habitat complexity tends to be positively related to biodiversity along shorelines (Kostylev et al. 2005; Schmude et al. 1998; Newbrey et al. 2005).

Shoreline biodiversity is heavily dependent on physical structure, nutrient inputs, and climate (Strayer and Findlay 2010), and the aforementioned effects that riprap and retaining wall usage can have on shoreline habitat can alter the community structure of fish (Jennings et al. 1999; Maceina et al. 1991; Toft et al. 2007; Brazner 1997; Bryan and Scarnecchia 1992; Kornis et al. 2018), aquatic plants (Strayer et al. 2012; Patrick et al. 2016), and benthic macroinvertebrates (Brauns et al. 2007; Bilkovic and Mitchell 2013). These effects also vary strongly based on species-specific attributes like diet, body size, and habitat preference (Kornis et al. 2018). For example, shoreline armoring, especially riprap revetments, can create favourable habitat for rock-oriented taxa while reducing habitat for species preferring macrophyte-dominated substrate such as Largemouth Bass (Scott and Crossman 1973; Warren 2009). Kornis et al. (2018) also found that nekton community structure at hardened and natural shores was influenced mostly by their feeding strategy and biomass was highest at hardened shores relative to natural shores.

Habitat alteration can also reduce the diversity and abundance of benthic macroinvertebrates. Brauns et al. (2007) found a positive relationship between the proportion of developed shorelines in a lake and the abundance of chironomids, however the abundance of other macroinvertebrate taxa exhibited a negative trend. The same study also found variance in the abundance of different feeding strategies at developed shores (riprap, beaches, retaining walls) and natural shores, mirroring trends found in nekton communities. Additionally, Lovall et al. (2017) noted a decrease in benthic infauna biomass on armored shores, as well as a shift from predatory taxa to filter-feeding taxa. Morley et al. (2012) also observed a tenfold decrease in epibenthic invertebrate density at armored shorelines compared to unaltered shores. Recently, developers have been moving towards adopting alternative "natural" mitigation strategies that involve live plants (i.e., living shorelines; Bilkovic et al. 2016) instead of hard substrate as shoreline protection. However, hybrid shorelines still may not be as diverse as fully natural shores (Bilkovic and Mitchell 2013). As benthic macroinvertebrates serve as qualitative indicators of ecosystem health (Purcell

et al. 2009; Adakole and Anunne 2004) understanding the impact of shoreline armoring on benthos communities has important applications for a variety freshwater conservation issues.

Anthropogenic shoreline development is expected to increase considerably in the future in both intensity and spread (Schmieder 2004). Understanding the impacts that common erosion mitigation techniques can have on littoral habitat and communities allows developers to make ecologically responsible decisions that contribute to the preservation of aquatic ecosystems. On an almost daily basis, resource managers and regulators review plans related to erosion control but have little empirical data from freshwater systems to guide them (Abdel-Fattah and Hasnain 2017). By studying the community composition of fish and benthic macroinvertebrates alongside characterization of habitat metrics at riprap revetments, retaining walls, and natural shorelines, we aim to provide a comprehensive case study of the ecological impacts of armored erosion mitigation on freshwater shoreline ecosystems.

Methods

Study site

Big Rideau Lake is a deep (maximum depth 110 m), oligo-mesotrophic (total phosphorous = 0.013 mg/L), dimictic lake located in eastern Ontario, Canada (RVCA 2014). The lake is 32 km long and 6 km wide at its widest point, making it the largest lake in the Rideau Canal system—a manmade waterway connecting Kingston and Ottawa. The lake has dense cottage development on most stretches of shoreline and heavy boat traffic. Additionally, the lake is a popular recreational fishery for black bass, hosting multiple competitive fishing tournaments each summer. These attributes allow Big Rideau Lake to serve as a model for other heavily developed recreational lakes in eastern Ontario and more broadly in midwestern and northeastern North America. Due to the extent of development, Parks Canada, the responsible authority for shoreline protection, routinely provides guidance for better implementation of erosion mitigation on recreational shorelines.

Site selection

Fieldwork began on July 5th, 2018 and continued periodically until completion on August 20th, 2018. Satellite imagery (Google Earth V 7.3) was used to locate 20 riprap sites, 20 retaining wall sites, and 20 natural sites throughout Big Rideau Lake (Fig. 1). Although we attempted to equally distribute treatments among all shorelines of the lake, many riprap and retaining wall sites were situated on the northernmost and southernmost shore where cottage density was greater. Sites were visually inspected in order to meet the following definitions: Riprap sites were defined as shoreline altered with large (boulder, cobble—often broken rock), unconsolidated rubble with an upwards sloping shore. Retaining walls were defined as shoreline altered with a wall of concrete or consolidated rock approximately

Fig. 1 Transect locations on Big Rideau Lake. Natural shorelines are denoted by crosses, riprap shorelines are denoted by circles, and retaining wall shorelines are denoted by triangles
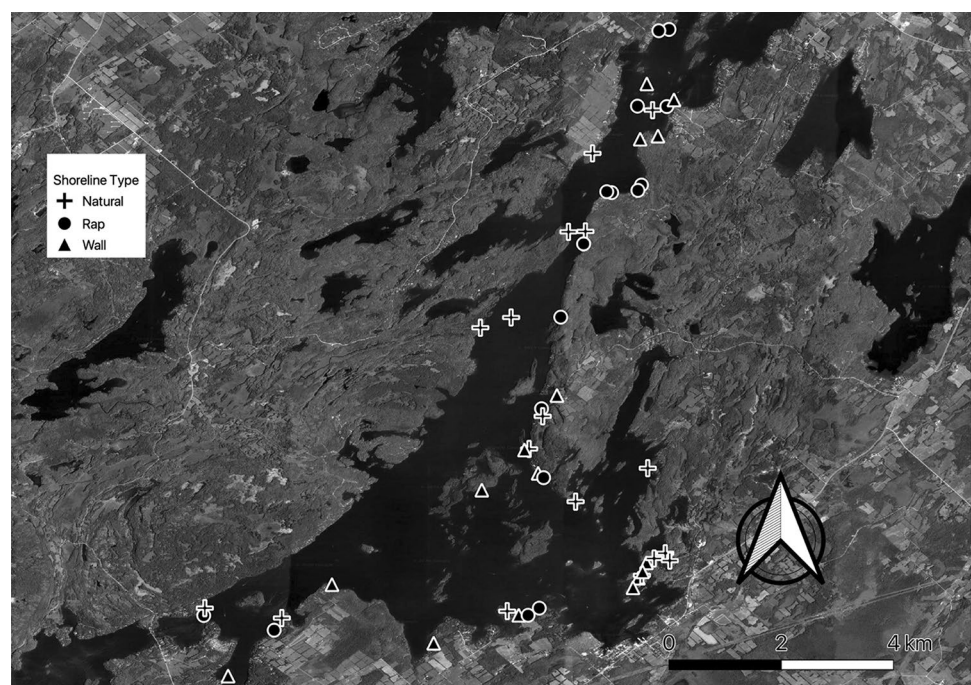
vertical. We selected natural rocky shorelines with similar slope angle and substrate size as altered rip rap shores. We felt this provided a strong comparison between anthropogenically altered and unaltered shore types. The sites we studied were not in backwater wetland areas nor artificial beach habitats where different forms of erosion mitigation were typically used. To the best of our ability we selected armored sites that would have been natural rocky shorelines prior to alteration (presumed based on habitats on either side of armored areas and general landscape configuration). We surveyed a different geographic region of the lake (north shore, south shore, east shore, west shore) each day and site order (riprap, retaining wall, natural) was systematically randomized on each excursion in order to prevent biasing results due to diurnal fish migrations (Romare et al. 2003). Additionally, surveying was conducted between the hours of 9:00 am and 4:00 pm under clear weather, thus avoiding crepuscular periods where fish community shifts might be expected. Upon arrival at a study site, we positioned the boat perpendicular to the shoreline with the bow approximately 5 m away from shore. Using a digital rangefinder (Halo XL450-7) we determined a 30 m transect parallel to the shore with a starting point at least 5 m away from the bow of the boat. This was done proactively to prevent disturbance of the study site prior to the swimmer entering the water. Sites were only selected if the shoreline treatment (riprap, retaining wall, natural rocky) spanned the entire 30 m of the transect in a continuous manner.

Habitat metrics

Habitat assessment was conducted following underwater visual census methods detailed in Zolderdo et al. (2019). Abundances of submergent macrophytes, emergent macrophytes, and coarse woody debris (CWD) were measured for each transect by assigning a score between 0 (no abundance) and 100 (total coverage of field of view). Assessment of habitat metrics was conducted at standardized intervals (0 m, 15 m, 30 m) along the transect, and only habitat characteristics within the immediate vicinity of the snorkeler (~5 m radius) were scored. Submergent macrophytes were defined as aquatic plants with photosynthetic parts exclusively underwater (*Ceratophylum, Elodea, Myriophyllum),* and emergent macrophytes were defined as aquatic plants with any photosynthetic parts exposed to air (*Lemna, Nymphaea*). Coarse woody debris was defined as any partially submerged or fully sunken wood below the water line and was assessed by the snorkeler underwater. Depth was visually estimated to the nearest 0.5 m and defined as the water depth immediately below the snorkeler. Habitat scores were assessed by a single snorkeler to prevent error/bias due to the subjective nature of the scoring system.

Fish identification and counting

All transects were surveyed by a single snorkeler using a standard kit (black mask, black snorkel, black fins, and black wetsuit) to limit the potential influence of equipment on fish behaviour. Snorkel surveys were conducted following standard snorkel survey protocols described in the Salmonid Field Protocols Handbook (O'Neal 2007). The field of observation was limited to the area between the swimmer and the shoreline with the swimmer looking slightly towards shore swimming at a steady pace (O'Neal 2007). Fish were counted and identified when passed to prevent double counting (O'Neal 2007). All observations were recorded underwater using pencil on a PVC dive slate which was also used to visually estimate the total length in centimeters of each individual fish encountered using a pre-drawn scale with 5 cm breaks. We believe this allowed us to estimate fish length with reasonable accuracy (nearest 5 cm). Due to difficulties with rapid identification, small baitfish were classified by two morphologically distinct families: Cyprinidae (minnows, dace, shiners) and Percidae (darters, small benthic fishes). These two families were easily distinguished by attributes such as the number of dorsal fins (1 in Cyprinidae, 2 in Percidae) and body shape (laterally compressed in Cyprinidae, dorsally compressed in Percidae). Due to the often schooling nature of these taxa, the snorkeler attempted to estimate the length of to the "average" sized individual although we recognize that this method was imperfect. Given that most fish in the Cyprinidae and Percidae family that we encountered were small, they were almost always scored as being 5 cm in length. The total number of individuals in a school was rapidly estimated to the nearest 5, although we noticed most schools were relatively manageable in size (1–10 individuals) and individuals were commonly of similar length. All surveying was conducted under sunny conditions with minimal wave action to maximize visibility and to limit the effect of weather on fish abundance and behaviour.

Benthic infauna sampling

A sample of bottom substrate was collected immediately after the snorkel survey by the snorkeler at the 15 m mark along the transect. Sediment sampling was conducted following the EPA protocol for kick-net sampling (EPA 2003). Samples were collected by agitating bottom substrate using rapid kicks to dislodge sediment into a 500 µm kick net while conducting a 2 m linear sweep. Samples were transported to the boat by the snorkeler and rinsed extensively to remove fine particles (~3 min) using a battery powered pump through a 500 µm sieve. Large debris such as rocks, twigs and broken shells were rinsed and manually discarded. Remaining material was transferred to a 500 mL resealable bag and stored on ice in a cooler until transport back to the

laboratory to prevent predation of small specimens by predatory taxa such as Anisoptera and Megaloptera. Samples were transported to the Queen's University Biological Station within hours of their collection where they were subjected to further rinsing in a 500 μm sieve and further removal of large debris. Due to the relatively close proximity of the study site to the laboratory, samples remained on ice for no longer than 1 h. Remaining material was then transferred into 500 mL Whirl–Pak bags and preserved in 90% ethanol.

Benthic infauna identification

Benthos identification and picking was conducted following the Ontario Benthos Biomonitoring Network (OBBN) standard protocol for benthos picking (Jones et al. 2007). Preserved samples were transported to the Paleoecological Environmental Assessment and Research Lab (PEARL) at Queen's University in Kingston, Ontario. For each sample, material was passed through a stacked sieve system comprised of a 2000 μm sieve on top of a 500 μm sieve. Material was rinsed extensively, and the coarse sieve was visually inspected for large specimens that remained. Large specimens were manually picked from the sieve using fine tweezers and transferred to a large 1 L beaker. These were usually composed of specimens from the orders Anisoptera, Amphipoda, Trichoptera, and Ephemeroptera. Material retained on the 500 μm sieve was transferred to a large 1 L beaker using a squeeze bottle of distilled water. Subsampling was then conducted following the OBBN "bucket" method (Jones et al. 2007). The large 1L beaker was filled with 500 mL of tap water and agitated. A small, 100 mL beaker was then submerged to gather a subsample of specimens. Floating specimens in the small beaker were counted, identified, and stored in a small vial of 90% ethanol. Excess water in the small beaker was poured back into the large beaker until ~ 80 mL remained. The sample was then transferred into shallow dissecting trays (80 mL) to be analyzed under a dissecting microscope (Nikon SMZ645). Specimens were picked using the OBBN benthos tally sheet under microscope and transferred into the ethanol filled vial for archiving. Only specimens with an intact head and thorax were fully identifiable and therefore counted towards the final tally. Empty protective cases of Trichoptera were not counted. Subsampling was conducted until at least 100 specimens were picked in total or the entire original sample was picked (Jones et al. 2007).

Benthic infauna pollution tolerance classification

Benthic infauna were categorized by their pollution tolerance. Group 1 taxa (mayflies, stoneflies, caddisflies) were classified as sensitive to pollution, Group 2 taxa (amphipods, isopods, damselflies, dragonflies) were classified as

intermediate tolerance, and Group 3 taxa (leeches, chironomids, aquatic worms) were classified as pollution tolerant (Kerans and Karr 1994).

Data analysis

R Version 1.1463 (R Core Team 2019) was used to conduct statistical analyses and create figures. Species richness and relative abundance of fish and benthic infauna were compared between shoreline types using generalized linear models (GLM's). GLM's for count data (richness, relative abundance) were run under a Poisson distribution with shoreline type as the single dependent variable. We used Tukey's honest significant difference test (TukeyHSD) to draw pairwise comparisons for the abundance of each fish species and benthic infauna taxa between shore types. Data for total abundance and depth were continuous, so GLM's were run under a normal distribution and pairwise comparisons made using TukeyHSD. Habitat metrics were scored on a scale of 0–100, and scores were rounded to the nearest 5. This allowed us to use Poisson regression analysis to compare habitat scores of submergent macrophytes, emergent macrophytes, and coarse woody debris between shore types and make pairwise comparisons using TukeyHSD. For each transect, biomass (W) of each fish species was calculated using total length (L), known regression coefficients for each species [intercept (a) and slope (b)] and the standard fish biomass equation $W = aL^b$ (Schneider et al. 2000), an approach previously used to survey fish communities in this region (Zolderdo et al. 2019). Because baitfish were only identified to the family level, regression coefficients of representative species (Spottail Shiner (*Notropis hudsonius*) for Cyprinidae and Blackside Darter (*Percina maculata*) for Percidae) were used for our calculations (Schneider et al. 2000). Kruskal–Wallis tests were used to compare differences in biomass of fish species between shore types as the data did not follow a normal distribution. Post-hoc comparisons were conducted using Dunn's test.

Results

Habitat characteristics

Transect depth ranged between 0.5 m and 1.5 m and mean depth of all reaches was 1 m. Mean transect depth varied significantly between shoreline types (df = 2, f = 13.88, p < 0.001) and was significantly greater in natural sites compared to armored sites (Riprap—Natural: p = 0.002, Wall—Natural: p < 0.001). Armored shorelines scored significantly lower for all habitat metrics. Submergent macrophyte scores varied significantly between shoreline types (f = 3.87, df = 2, p = 0.03) and were 4× higher in natural shorelines compared

to retaining walls (p = 0.03) (Fig. 2). Emergent macrophyte scores also varied significantly between shoreline types (f = 7.53, df = 2, p = 0.001) and were significantly higher in natural shores compared to retaining walls (p = 0.002) and riprap revetments (p = 0.01) (Fig. 2). Coarse woody debris scores varied significantly between sites (f = 4.85, df = 2, p = 0.01) and were significantly higher in natural shores compared to armored shores (Riprap—Natural: p = 0.04, Wall—Natural, p = 0.02) (Fig. 2).

Fish diversity and relative abundance

Centrarchids dominated assemblages at all shoreline types in Big Rideau Lake. Of the centrarchids, Bluegill (*Lepomis macrochirus*) were the most abundant while Largemouth Bass were the least abundant. Northern Pike and Percid minnows were rarely observed (< 3 individuals) during the study, thus data were not analyzed. Total abundance was typically highest in riprap sites and lowest in natural sites, however the difference was not statistically significant (Fig. 3). Species richness did not vary significantly among shoreline types (chisq = 1.08, df = 2, p = 0.5) (Fig. 3). Pumpkinseed abundance varied significantly among shoreline types (chisq = 28.13, df = 2,
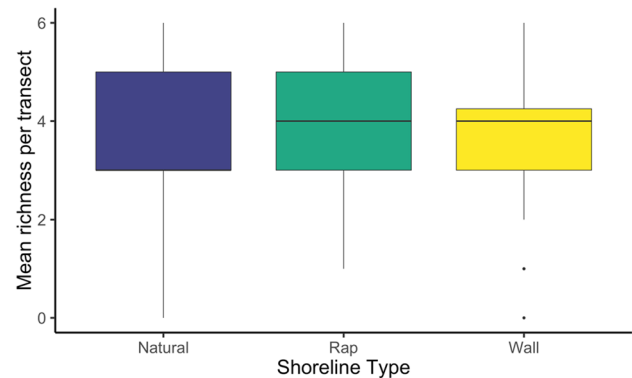
Fig. 3 Species richness and total abundance of fish at natural, riprap, and retaining wall shorelines. Error bars denote min/max values
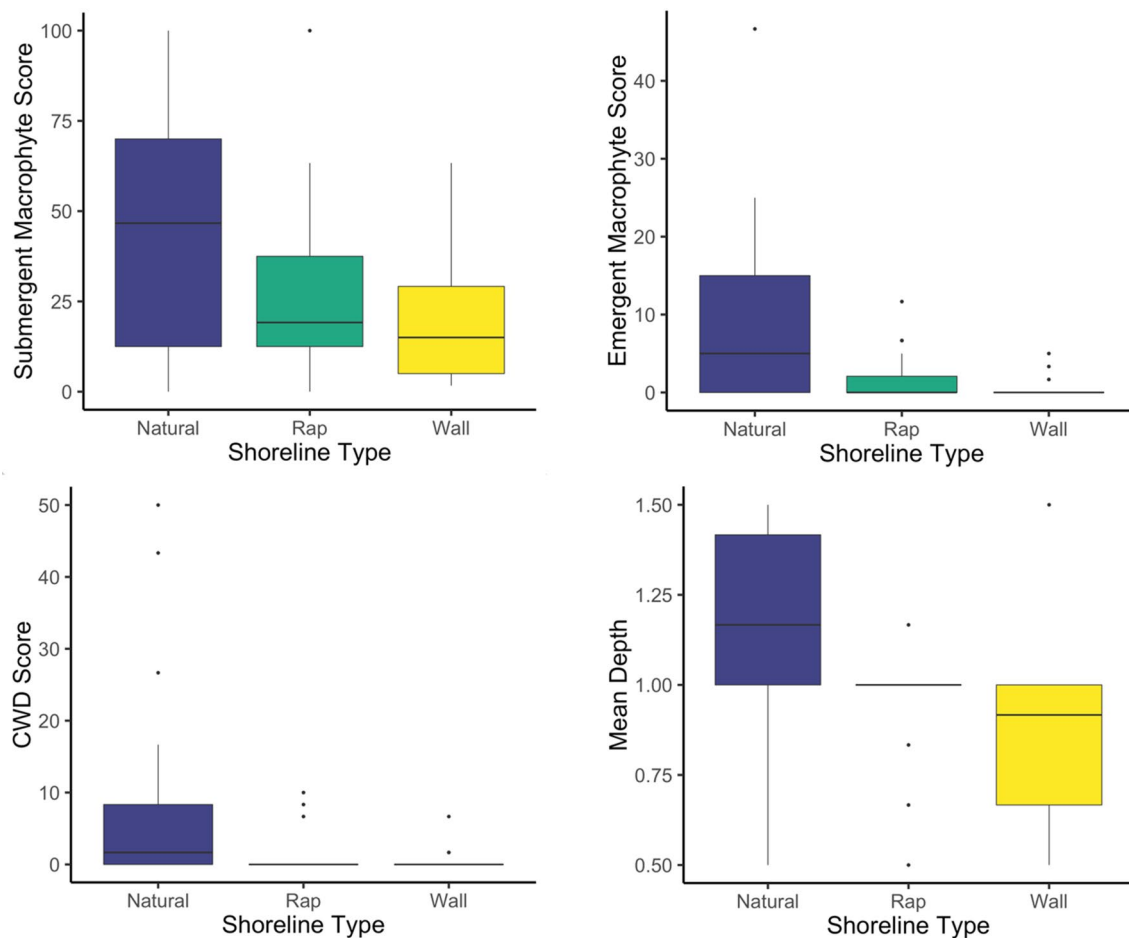
Fig. 2 (From top left): mean habitat score per transect for submergent macrophytes, emergent macrophytes, and coarse woody debris (CWD) along with mean transect depth (m). Error bars denote min/max values

p < 0.001) and was significantly greater at retaining walls and riprap shores compared to natural shorelines (natural-wall: p < 0.001, natural-rap: p = 0.02). Pumpkinseed abundance was also significantly higher at retaining walls compared to riprap shores (p = 0.04). Bluegill abundance varied significantly between shoreline types (chisq = 11.6, df = 2, p = 0.002) and was significantly higher at retaining wall sites compared to natural sites (p = 0.002) and trending towards being higher at riprap shores compared to natural shores (p = 0.07). Yellow Perch abundance varied significantly among shoreline types (chisq = 24, df = 2, p < 0.001) and was significantly higher in both armored sites compared to natural sites (wall-natural: p < 0.001, rap-natural: p = 0.003) but did not differ between armored sites (p = 0.3). Rock Bass abundance varied significantly between shoreline types (chisq = 72.25, df = 2, p < 0.001) and was significantly higher in riprap sites compared to both natural sites (p < 0.001) and retaining walls (p < 0.001). Cyprinid minnow abundance varied significantly among shoreline types (chisq = 42.1, df = 2, p < 0.001) and was significantly higher in natural sites compared to retaining walls (p = 0.001) and higher in riprap sites compared to retaining walls (p < 0.001). Largemouth Bass abundance varied significantly between shoreline types (chisq = 21.2, df = 2, p < 0.001) and was significantly higher in natural sites compared to both riprap (p = 0.008) and retaining walls (p < 0.01). Smallmouth Bass abundance did not vary among shoreline types (chisq = 0.02, df = 2, p = 0.99) (Fig. 4). Total biomass was greatest at riprap shorelines and lowest at natural shorelines but did not vary for individual fish species among shoreline types.
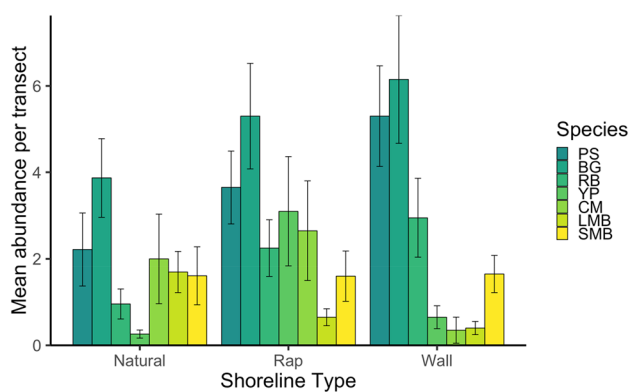

Fig. 4 Relative abundance of Pumpkinseed (PS), Bluegill (BG), Rock Bass (RB), Yellow Perch (YP), Cyprinid minnows (CM), Largemouth Bass (LMB), and Smallmouth Bass (SMB) at natural, riprap, and retaining wall shorelines. Error bars denote standard error. Data for Percid minnows and Northern Pike were not analyzed due to sample size constraints

Benthic macroinvertebrate diversity and abundance

Species richness of benthic infauna did not vary between shoreline types (Fig. 5). The majority of samples contained at least 100 specimens, and the four samples that did not were from retaining walls and natural shorelines. Amphipod abundance varied significantly between shoreline types (chisq = 21.7, df = 2, p < 0.001) and was significantly higher at riprap shorelines compared to retaining walls (p = 0.002) and natural shorelines (p < 0.001). Isopod abundance varied significantly between shoreline types (chisq = 16.4, df = 2, p < 0.001) and was significantly higher at riprap shorelines compared to retaining walls (p = 0.002) and natural sites (p < 0.01). Cladocera abundance varied significantly between shoreline types (chisq = 26.8, df = 2, p < 0.001) and was significantly higher at riprap shorelines compared to natural sites (p = 0.03) and retaining walls (p < 0.001). Cladocera abundance was also significantly higher at natural sites compared to retaining walls (p = 0.003). Mayfly abundance varied significantly among sites (chisq = 14.6, df = 2, p < 0.01) and was significantly higher at retaining walls compared to natural shorelines (p = 0.02) and riprap shorelines (p < 0.01) (Fig. 6).

Discussion

The aquatic macrophyte community was severely reduced at both riprap sites and retaining walls compared to natural shorelines. Loss of biotic habitat was the most prevalent at retaining wall sites where submergent and emergent macrophyte coverage was the lowest out of all shoreline types. Retaining walls may increase reflectance of wave action by the shoreline into the littoral zone, causing increased turbidity and inhibiting root attachment (Patrick et al. 2016). Decreased water clarity also reduces light penetration into
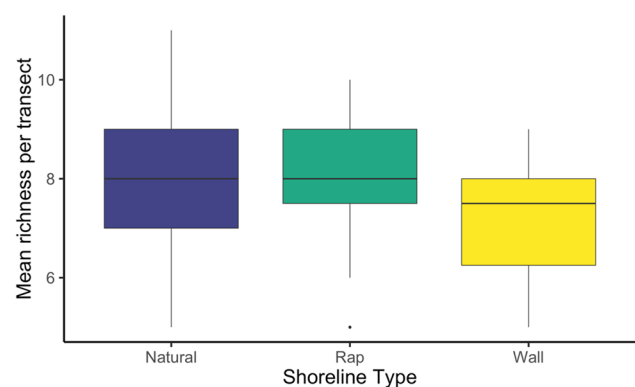

Fig. 5 Taxonomic richness of benthic macroinvertebrates at natural, riprap, and retaining wall shorelines. Error bars denote min/max values
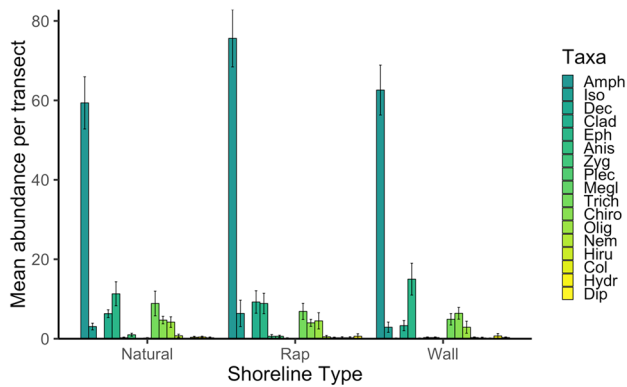
Fig. 6 Relative abundance of benthic macroinvertebrates at natural, riprap, and retaining wall shorelines. *Amph* amphipoda, *Iso* isopoda, *Dec* decapoda, *Clad* cladocera, *Eph* ephemeroptera, *Anis* anisoptera, *Zyg* zygoptera, *Plec* plecoptera, *Megl* megaloptera, *Trich* trichoptera, *Chiron* chironomidae, *Oligo* oligochaeta, *Nem* nematoda, *Hiru* hirudinea, *Col* coleoptera, *Hydr* hydrachnidia, *Dip* diptera. Error bars denote standard error

the littoral zone and could also negatively impact some macrophyte species (Engel and Nichols 1994). We also hypothesize that excavation resulting from the implementation of riprap and retaining walls could have resulted in the removal of existing macrophyte stands and led to long-term reductions in macrophyte abundance. It is also possible that because these sites are adjacent to cottages, there may be efforts by the cottagers to actively remove macrophytes although we saw no specific evidence of weed harvesting. Unfortunately, we were unable to determine the exact time erosion mitigation was implemented at any reach, and thus lacked information on whether construction of mitigation techniques has lasting impacts on adjacent macrophyte abundance. Future studies could investigate this link by experimentally altering natural shorelines and monitoring macrophyte abundance over time. Macrophytes provide the primary energy inputs in the littoral zone and also serve as vital habitat for both prey fish and ambush predators. Many studies have found correlations between submerged aquatic macrophyte stands and fish abundance in both coastal and freshwater environments (Durocher et al. 1984; Bryan and Scarnecchia 1992). The effects of macrophyte reduction can vary between environments and species. Predator–prey interactions in fish communities can be mediated by the abundance and density of submerged macrophytes, which provide cover for both refuging and ambushing (Savino and Stein 1982). In one case, macrophyte removal increased abundance of forage fish and subsequently increased growth and recruitment of Black and White Crappie (*P. nigromaculatus* and *P. annularis*, respectively) (Maceina et al. 1991). We observed greater abundance of some piscivorous species (Yellow Perch, Rock Bass) at shorelines with lower macrophyte density (retaining wall, rip rap), however other species

were observed at equal or lower abundance (Largemouth Bass), possibly due to species specific foraging methods. One study showed that Largemouth Bass favour moderately dense but diverse macrophyte assemblages for optimal foraging (Valley and Bremigan 2002). Aquatic macrophytes also provide habitat for fish forage such as macroinvertebrates and zooplankton. In an African lake, gradual senescence of aquatic macrophytes resulted in a significant reduction of invertebrate biomass which correlated to reductions in the abundance of macrophyte-associated fish (Whitfield 1984).

Overall, reductions in aquatic macrophytes and loss of complex habitat have the potential to limit species richness as a whole and may increase abundance of species who prefer reduced macrophyte stands.

Significantly more coarse woody debris was observed in natural sites compared to both armored shoreline sites. Coarse woody debris is an important measure of high-quality littoral habitat (Harmon et al. 1986) as it increases habitat heterogeneity while also providing alternative refuging sites in areas where aquatic macrophyte density is reduced (Everett and Ruiz 1993). High abundance of woody debris reduces energetic costs in predators by allowing them to use ambush hunting as opposed to foraging in deep open water (Ahrenstorff et al. 2009). In riverine systems, coarse woody debris also provides velocity refuges from current (Fausch 1993). Littoral abundance of coarse woody debris relies primarily on terrestrial inputs from the riparian zone and anthropogenic alteration of the riparian zone that removes vegetation can significantly decrease coarse woody debris inputs. Human activities, especially residential development, generally result in reductions of coarse woody debris in lakes (Christensen et al. 1996), and whole lake reductions in coarse woody debris can significantly and rapidly decrease abundance of fish who rely on wood as spawning substrate such as Yellow Perch (Helmus and Sass 2008). In developed sites where coarse woody debris is reduced or non-existent, fish communities could shift towards pelagic, rocky substrate associated species and predators that rely on overhead cover for ambush may be forced to use more energetically costly hunting methods.

Natural sites were substantially deeper than sites with riprap and retaining walls. This could be viewed as confounding, yet littoral depth may contribute to both the need for erosion control in the first instance and the selection of appropriate erosion control methods. It is generally known that shallower water leads to increased wave height, which could lead cottagers to adopt erosion mitigation methods. Shallow littoral zones are also typically related to a gentler terrestrial slope and thus may also be more attractive for cottage developers for both ease of construction and recreation.

Anthropogenic alteration of shoreline habitat commonly modifies species richness and community structure of fish in a variety of settings (Teixeira et al. 2009; Jellyman and

Harding 2012; Jennings et al. 1999; Seitz et al. 2006; Bilkovic and Roggero, 2008). Nonetheless, we did not find any differences in fish species richness between the two shoreline types altered by erosion mitigation (riprap, retaining walls) and unaltered shorelines. This may be due to relatively low taxonomic richness of fish in the lake as well as our ability to resolve species (e.g., cyprinids were grouped into a single taxonomic group rather than expressed as individual species) using snorkel surveys. Only six species (Largemouth Bass, Smallmouth Bass, Rock Bass, Pumpkinseed, Bluegill, Yellow Perch*)* were regularly encountered, with five of those from the family Centrarchidae (Largemouth Bass, Smallmouth Bass, Rock Bass, Pumpkinseed, Bluegill). Centrarchids dominate most if not all assemblages in the Rideau Waterway which could be due to their ability to thrive in a wide range of habitat types (Rahel 1984; Tonn and Magnuson 1982). This trend could also be due to the similar feeding strategies of Centrarchids found in the lake (generalist, piscivore-benthivore), as Kornis et al. (2017) and Kornis et al. (2018) both observed that shifts in diversity between altered and natural shorelines was mediated primarily by functional group. Total abundance of all fish species was slightly higher in sites with riprap compared to both retaining wall and natural sites. While no metrics of habitat complexity were measured for our study, riprap sites provide generally more complex habitat compared to retaining wall sites as the nature of their loose construction creates more overhead cover and refuging areas among the unembedded rocks (Pister 2009; Garland et al. 2002; Erös et al. 2008). Much like in terrestrial systems, aquatic habitat complexity and increased habitat heterogeneity strongly correlates with species richness (Gratwicke and Speight 2005; Roberts and Ormond 1987; Eadie and Keast 1984). More specifically, Eadie and Keast (1984) found that fish diversity could be best predicted by substrate diversity and macrophyte complexity. Complex habitats provide the opportunity for a greater variety of niches and therefore can provide habitat for a greater diversity of species (August 1983). As anthropogenically altered shorelines are generally more homogenous (Jennings et al. 1999), we could observe decreased species richness at altered shorelines in environments where overall species richness is greater.

Recent research suggests that taxonomic richness may not be the best metric for measuring the effects of anthropogenic disturbance on communities. Grouping taxa by functional traits instead of by taxonomic rank may be more effective for comparing a variety of ecosystems with multiple disturbances (Mouillot et al. 2013). Grouping species by trait values to form a "functional matrix" allows researchers to quantify the consequences of a single disturbance on a single trait across multiple species (Mouillot et al. 2013). Because taxa often share similar niches and exhibit similar traits, functional grouping could be more accurate at quantifying the community level effects of anthropogenic disturbance. Using this approach in future studies combined with identifying all individuals to the species level could better reveal changes to fish community structure due to implementation of erosion mitigation methods.

Use of riprap and retaining walls had variable effects on the relative abundance of different fish species. Rock Bass, Bluegill, Pumpkinseed, Yellow Perch, and Cyprinid minnow abundances were greater at both types of armored sites compared to natural sites, contrary to observations for Largemouth Bass. These differences in abundance could be due to species-specific preference towards different levels of macrophyte abundance and the presence of coarse woody debris. Additionally, the increased density of aquatic macrophytes at natural shores could have reduced the snorkeler's ability to count and identify small baitfish, as opposed to developed shorelines with more open water. Significant positive correlations between percent submerged vegetation and Largemouth Bass size and recruitment have been observed (Durocher et al. 1984), suggesting Largemouth Bass have greater success in areas with high macrophyte coverage. The presence or absence of coarse wood can also alter the behaviour of Largemouth Bass. In lakes with low abundance of coarse woody debris, Largemouth Bass have larger home ranges, spend more time in deep water, and spend more time foraging in open water compared to bass in lakes with an abundance of coarse woody debris (Ahrenstorff et al. 2009). This could ultimately decrease size and growth due to greater energetic costs associated with foraging in open water compared to a sit-and-wait approach often used by predators with adequate cover (Sass et al. 2006).

Contrary to the trends observed for Largemouth Bass, Rock Bass, Bluegill, Pumpkinseed, Yellow Perch, and Cyprinid minnow abundance was greater at both types of armored sites compared to natural sites. This relationship could be explained by a preference for habitat with limited macrophyte coverage. Increased coverage of macrophytes sometimes results in resource partitioning between Rock Bass and other littoral Centrarchids. In juvenile Rock Bass, increased abundance of aquatic macrophytes shifts their diet from small, abundant benthic macroinvertebrates (amphipods, isopods) to larger, less abundant benthic macroinvertebrates such as Caenidae and Trichoptera as a method of resource partitioning with Pumpkinseeds who continue to consume small macroinvertebrates (French 1988). Dense vegetation can also alter migration patterns in Rock Bass. In a lake in the Rideau System, Keast (1984) found suppressed diurnal migration when densities of the invasive macrophyte *M. spicatum* were high. This suggests that habitat preference in Rock Bass is complex and influenced by multiple factors including macrophyte coverage and abundance of other Centrarchid species.

We found no evidence to suggest differences in taxonomic richness of benthic macroinvertebrates between the three shoreline types. Many prior studies have shown varying levels of change in littoral macroinvertebrate communities in response to different alteration methods, however most suggest that taxonomic richness is greatest in natural sites (Brauns et al. 2007; Sobocinski et al. 2010; Kasangaki et al. 2006). Kasangaki et al. (2006) also found the highest abundance of pollution intolerant taxa such as Ephemeroptera, Plecoptera and Trichoptera at natural sites. Natural sites also have higher habitat complexity compared to anthropogenically altered sites (Jennings et al. 1999). Complex habitats often support more ecological niches and subsequently can support a higher level of taxonomic richness (O'Connor 1991). The relative abundance of taxa from the three pollution tolerance groups can provide insight on water quality metrics such as dissolved oxygen, phosphorous concentration, and heavy metal pollution (Gaufin and Tarzwell 1956; Goodnight 1973; Fleituch et al. 2002; Mousavi et al. 2003; Arimoro and Muller 2010). Developed shores tended to support an increased abundance of amphipods, isopods and Cladocera (*Daphnia* spp.), while mayflies (*Ephemeroptera* spp.) were the most abundant at retaining wall sites, and lowest at riprap sites. The abundance of other Sensitive (Group 1) taxa did not vary between altered and natural shores, indicating a marginal difference in water quality. The differences in mayfly abundance could be better explained by habitat and substrate characteristics as some mayfly species prefer areas of soft sediment in which they can burrow (McCafferty 1975).

While benthic macroinvertebrates are strong indicators of anthropogenic impact on ecosystems (Cairns 1993; Adakole and Anunne 2004), changes to water quality at altered shores on Big Rideau Lake may not have exceeded certain thresholds for shifts in benthic communities. Water quality on Big Rideau is classified as "fair" by the Rideau Valley Conservation Authority which noted few nutrient exceedances, good oxygen conditions, clear water and occasionally elevated pH (RVCA 2014). Additionally, the relatively recent watershed developments of Big Rideau may not have allowed benthic communities to show signs of shifting. Helmus and Sass (2008) found no effects on littoral macroinvertebrate assemblages after rapidly reducing coarse woody debris throughout the lake and observing changes to benthic communities over two years. In some highly developed areas of the lake, the RVCA has classified water quality as "poor" based on increased nutrient concentrations and poor water clarity (RVCA 2014). We were unable to gather macroinvertebrate samples from these intensely developed areas but macroinvertebrate assemblages from these areas could serve as an example of what is to be expected in the future. As development of the Big Rideau Lake watershed continues, we may slowly start to see changes in benthos assemblages however it may be too soon to observe such changes.

Our study demonstrates that erosion mitigation through use of riprap and retaining walls is correlated with littoral habitat alterations and can subsequently affect the community structure of littoral fish. Due to the ecological importance of the littoral zone, long-term changes to littoral fish communities could ultimately lead to large changes in whole-lake diversity. By combining assessment of habitat, fish and benthic communities, we provide a comprehensive report on the impacts that erosion mitigation methods can have on littoral ecosystems at multiple levels. This study will allow landowners and management officials to promote ecologically responsible development and will support implementation of alternative methods of erosion mitigation that better mimic a naturalized shoreline in order to conserve littoral habitat. Recent research has suggested that naturalized shorelines with high riparian vegetation density are often better at coping with the effects of wave induced erosion compared to armored shores, which can induce erosion of the intertidal zone (DFO 2008; Gittman et al. 2015). We suggest that there is a need for research on alternative erosion control methods that achieve erosion control objectives while maintaining or enhancing aquatic biodiversity.

References

Abdel-Fattah S, Hasnain S (2017) What are the impacts on temperate fish productivity of shoreline works activities? A systematic review protocol. Environ Evid 6:31

Adakole J, Anunne P (2004) Benthic macroinvertebrates as indicators of environmental quality of an urban stream, Zaria, Northern Nigeria. J Aquat Sci 18(2):85–92

Ahrenstorff T, Sass G, Helmus M (2009) The influence of littoral zone coarse woody habitat on home range size, spatial distribution, and feeding ecology of largemouth bass (*Micropterus salmoides*). Hydrobiologia 623(1):223–233

Arimoro FO, Muller WJ (2010) Mayfly (Insecta: Ephemeroptera) community structure as an indicator of the ecological status of a stream in the Niger Delta area of Nigeria. Environ Monit Assess 166:581–594

Arkema KK, Guannel G, Verutes G, Wood SA, Guerry A, Ruckelshaus M, Kareiva P, Lacayo M, Silver JM (2013) Coastal habitats shield people and property from sea-level rise and storms. Nat Climate Change 3:913–918

August P (1983) The role of habitat complexity and heterogeneity in structuring tropical mammal communities. Ecology 64(6):1495–1507

Bilkovic DM, Mitchell MM (2013) Ecological tradeoffs of stabilized salt marshes as a shoreline protection strategy: effects of artificial structures on macrobenthic assemblages. Ecol Eng 61(A):469–481

Bilkovic D, Roggero M (2008) Effects of coastal development on nearshore estuarine nekton communities. Mar Ecol Prog Ser 358:27–39

Bilkovic DM, Mitchell M, Mason P, Duhring K (2016) The role of living shorelines as estuarine habitat conservation strategies. Coast Manag 44(3):161–174

Bozek CM, Burdick DM (2005) Impacts of seawalls on saltmarsh plant communities in the Great Bay Estuary, New Hampshire USA. Wetl Ecol Manag 13(5):553–568

Brauns M, Garcia X, Walz N, Pusch M (2007) Effects of human shoreline development on littoral macroinvertebrates in lowland lakes. J Appl Ecol 44(6):1138–1144

Brazner JC (1997) Regional, habitat, and human development influences on coastal wetland and beach fish assemblages in Green Bay, Lake Michigan. J Great Lakes Res 23(1):36–51

Brooks TM, Mittermeier RA, Mittermeier CG, Da Fonseca GAB, Rylands AB et al (2002) Habitat loss and extinction in the hotspots of biodiversity. Conserv Biol 16(4):909–923

Bryan M, Scarnecchia D (1992) Species richness, composition, and abundance of fish larvae and juveniles inhabiting natural and developed shorelines of a glacial Iowa lake. Environ Biol Fishes 35(4):329–341

Cairns Jr J (1993) A history of biological monitoring using benthic macroinvertebrates. In: Freshwater biomonitoring and benthic macroinvertebrates, pp 10–27

Christensen D, Herwig B, Schindler D (1996) Impacts of lakeshore residential development on coarse woody debris in north temperate lakes. Ecol Appl 6(4):1143–1149

Dillon P, Scheider W, Reid R, Jeffries D (1994) Lakeshore capacity study: part I—test of effects of shoreline development on the trophic status of lakes. Lake Reserv Manag 8(2):121–129

Dugan JE, Airoldi L, Chapman MG, Walker SG, Schlacher T (2011) Estuarine and coastal structures: environmental effects, a focus on shore and nearshore structures. In: Wolanski E, McLusky D (eds) Treatise on estuarine and coastal science. Elsevier Press, New York, pp 17–41

Durocher P, Provine W, Kraai J (1984) Relationship between abundance of largemouth bass and submerged vegetation in Texas reservoirs. N Am J Fish Manag 4(1):84–88

Eadie J, Keast A (1984) Resource heterogeneity and fish species diversity in lakes. Can J Zool 62(9):1689–1695

Engel S, Nichols SA (1994) Aquatic macrophyte growth in a turbid windswept lake. J Freshw Biol 9(2):97–109

Enironmental Protection Agency (EPA) (2003) Benthic macroinvertebrate protocols. https://archive.epa.gov/water/archive/web/html/ch07main.html. Accessed 4 Feb 2020

Erös T, Tóth B, Sevcsik A, Schmera D (2008) Comparison of fish assemblage diversity in natural and artificial rip-rap habitats in the littoral zone of a large river (River Danube, Hungary). Int Rev Hydrobiol 93(1):88–105

Everett R, Ruiz G (1993) Coarse woody debris as a refuge from predation in aquatic communities. Oecologia 93(4):475–486

Fausch K (1993) Experimental analysis of microhabitat selection in juvenile steelhead (*Oncorhynchus mykiss*) and coho salmon (*O. kisutch*) in a British Columbia stream. Can J Fish Aquat Sci 50(6):1198–1207

Fisheries and Oceans Canada (DFO) (2008) The shore primer. In: Fisheries and Oceans Canada. Fish Habitat Management Program, Praries Area (ISBN: 978-0-662-48025-9)

Fleituch T, Sozka H, Kudelska D, Kownacki A (2002) Macroinvertebrates as indicators of water quality in rivers: a scientific basis for Polish standard method. Large Rivers 13(3–4):225–239

French J (1988) Effect of submersed aquatic macrophytes on resource partitioning in yearling rock bass (*Ambloplites rupestris*) and pumpkinseeds (*Lepomis gibbosus*) in Lake St Clair. J Great Lakes Res 14(3):291–300

Garland R, Tiffan K, Rondorf D, Clark L (2002) Comparison of subyearling fall chinook salmon's use of riprap revetments and unlatered habitats in Lake Wallula of the Columbia River. N Am J Fish Manag 22(4):1283–1289

Garrison PJ, Laliberte GD, Ewart BP (2010) The importance of water level changes and shoreline development in the eutrophication of a shallow, seepage lake. Proc Acad Nat Sci Phila 160(1):113–126

Gaufin AR, Tarzwell CM (1956) Aquatic macro-invertebrate communities as indicators of pollution in Lytle Creek. Sew Ind Wastes 28(7):906–924

Giam X, Bradshaw CJA, Tan HTW, Sodhi NS (2010) Future habitat loss and the conservation of plant biodiversity. Biol Cons 143(7):1594–1602

Gittman R, Fodrie F, Popwich A, Keller D, Bruno J, Currin C, Peterson C, Piehler M (2015) Engineering away our natural defenses: an analysis of shoreline hardening in the US. Front Ecol Environ 13(6):301–307

Goodnight CJ (1973) The use of aquatic macroinvertebrates as indicators of stream pollution. Trans Am Microsc Soc 92(1):1–13

Gratwicke B, Speight M (2005) The relationship between fish species richness, abundance and habitat complexity in a range of shallow tropical marine habitats. J Fish Biol 66(3):650–667

Hall D, Werner E (1977) Seasonal distribution and abundance of fishes in the littoral zone of a Michigan lake. Trans Am Fish Soc 106(6):545–555

Harmon M, Franklin J, Swanson F, Sollins P, Gregory S, Lattin J, Anderson N, Cline S, Aumen N, Sedell J, Lienkaemper G, Cromack K, Cummins K (1986) Ecology of coarse woody debris in temperate ecosystems. Adv Ecol Res 15:133–302

Helmus M, Sass G (2008) The rapid effects of whole-lake reduction of coarse woody debris on fish and benthic macroinvertebrates. Freshw Biol 53(7):1423–1433

James MR, Weatherhead M, Stanger C, Graynoth E (1997) Macroinvertebrate distribution in the littoral zone of Lake Coleridge, South Island, New Zealand—effects of habitat stability, wind exposure, and macrophytes. N Z J Mar Freshw Res 32(2):287–305

Jellyman P, Harding J (2012) The role of dams in altering freshwater communities in New Zealand. N Z J Mar Freshw Res 46(4):475–489

Jennings M, Bozek M, Hatzenbeler G, Emmons E, Staggs M (1999) Cumulative effects of incremental shoreline habitat modification on fish assemblages in north temperate lakes. N Am J Fish Manag 19:18–27

Jennings M, Emmons E, Hatzenbeler G, Edwards C, Bozek M (2003) Is littoral habitat affected by residential development and land use in watersheds of Wisconsin lakes? Lake Reserv Manag 19(3):272–279

Jones C, Somers KM, Craig B, Reynoldson TB (2007) Ontario benthos biomonitoring network: protocol manual (**ISBN: 978-1-4249-2121-8**)

Kasangaki A, Babaasa D, Efitre J, McNeilage A, Bitariho R (2006) Links between anthropogenic perturbations and benthic macroinvertebrate assemblages in Afromontane forest streams in Uganda. Hydrobiologia 563:231–245

Keast A (1984) The introduced aquatic macrophyte, *Myriophyllum spicatum*, as habitat for fish and their invertebrate prey. Can J Zool 62:1289–1303

Kerans BL, Karr JB (1994) A benthic index of biotic integrity (B-IBI) for rivers of the Tennessee Valley. Ecol Appl 4(4):768–785

Kornis MS, Breitburg D, Balouskus R, Bilkovic DM, Davias LA et al (2017) Linking the abundance of estuarine fish and crustaceans

in nearshore waters to shoreline hardening and land cover. Estuar Coasts 40(5):1464–1486

Kornis MS, Bilkovic DM, Davias LA, Giordano S, Breitburg DL (2018) Shoreline hardening affects nekton biomass, size structure, and taxonomic diversity in nearshore waters, with responses mediated by functional species groups. Estuar Coasts 41:159–179

Kostylev VE, Erlandsson J, Ming MY, Williams GA (2005) The relative importance of habitat complexity and surface area in assessing biodiversity: fractal application on rocky shores. Ecol Complex 2(3):272–286

Liddle M, Scorgie H (1980) The effects of recreation on freshwater plants and animals: a review. Biol Cons 17(3):183–206

Lovall CD, Seitz RD, Knick KE (2017) Direct and indirect impacts of shoreline development on shallow-water benthic communities in a depauperate estuarine system. Bull Mar Sci 93(3):715–741

Maceina M, Bettoli P, Klussmann W, Betsill R, Noble R (1991) Effect of aquatic macrophyte removal on recruitment and growth of black crappies and white crappies in Lake Conroe, Texas. N Am J Fish Manag 11(4):556–563

Mallin M, Williams K, Esham E, Lowe R (2000) Effect of human development on bacteriological water quality in coastal watersheds. Ecol Appl 10(4):1047–1056

McCafferty WP (1975) The burrowing mayflies (Ephemeroptera: Ephemeroidea) of the United States. Trans Am Entomol Soc 101:447–504

Millenium Ecosystem Assessment (2005) Ecosystems and human well-being: synthesis. Island Press, Washington, DC

Morley SA, Toft JD, Hanson KM (2012) Ecological effects of shoreline armoring on intertidal habitats of a Puget sound urban estuary. Estuar Coasts 35(3):774–784

Mouillot D, Graham N, Bestien Villé Ger S, Mason N, Bellwood D (2013) A functional approach reveals community responses to disturbances. Trends Ecol Evol 28(3):167–177

Mousavi SK, Primicerio R, Amundsen PA (2003) Diversity and structure of Chironomidae (Diptera) communities along a gradient of heavy metal contamination in the subarctic watercourse. Sci Total Environ 307(1–3):93–110

National Research Council (NRC) (2007) Mitigating shore erosion along sheltered coasts. The National Academies Press, Washington, D.C.

Newbrey MG, Bozek MA, Jennings MJ, Cook JE (2005) Branching complexity and morphological characteristics of coarse woody structure as lacustrine fish habitat. Can J Fish Aquat Sci 62(9):2110–2123

O'Connor N (1991) The effects of habitat complexity on the macroinvertebrates colonising wood substrates in a lowland stream. Oecologia 85(4):504–512

O'Neal JS (2007) Snorkel surveys. Salmonid field protocols handbook: techniques for assessing status and trends in salmon and trout populations. American Fisheries Society, Bethesda, pp 324–336

Patrick CJ, Weller DE, Ryder M (2016) The relationship between shoreline armoring and adjacent submerged aquatic vegetation in Chesapeake Bay and nearby Atlantic Coastal Bays. Estuar Coasts 39(1):158–170

Peterson MS, Lowe MR (2009) Implications of cumulative impacts to estuarine and marine habitat quality for fish and invertebrate resources. Rev Fish Sci 17(4):505–523

Pimm SL, Askins RA (1995) Forest losses predict bird extinctions in eastern North America. Proc Natl Acad Sci USA 92(20):9343–9347

Pimm SL, Raven P (2000) Extinction by numbers. Nature 403:843–845

Pister B (2009) Urban marine ecology in southern California: the ability of riprap structures to serve as rocky intertidal habitat. Mar Biol 156(5):861–873

Purcell A, Bressler D, Paul M, Barbour M, Rankin E, Carter J, Resh V (2009) Assessment tools for urban catchments: developing biological indicators based on benthic macroinvertebrates. J Am Water Resour Assoc 45(2):306–319

Quigley J, Harper D (2004) Streambank protection with rip-rap: an evaluation of the effects on fish and fish habitat. Can Manuscr Rep Fish Aquat Sci 2701:76

Radomski P, Goeman T (2001) Consequences of human lakeshore development on emergent and floating-leaf vegetation abundance. N Am J Fish Manag 21(1):46–61

Rahel F (1984) Factors structuring fish assemblages along a bog lake successional gradient. Ecology 65(4):1276–1289

Roberts CM, Ormond RFG (1987) Habitat complexity and coral reef fish diversity and abundance on Red Sea fringing reefs. Mar Ecol Prog Ser 41(1):1–8

Romare P, Berg S, Lauridsen T, Jeppesen E (2003) Spatial and temporal distribution of fish and zooplankton in a shallow lake. Freshw Biol 48(8):1353–1362

Sass G, Gille C, Hinke J, Kitchell J (2006) Whole-lake influences of littoral structural complexity and prey body morphology on fish predator-prey interactions. Ecol Freshw Fish 15(3):301–308

Savino J, Stein R (1982) Predator-prey interaction between largemouth bass and bluegills as influence by simulated, submersed vegetation. Trans Am Fish Soc 111(3):255–266

Schmieder K (2004) European lake shores in danger—concepts for a sustainable development. Limnologica 34(1–2):3–14

Schmude KL, Jennings MJ, Otis KJ, Piette RR (1998) Effects of habitat complexity on macroinvertebrate colonization of artificial substrates in North Temperate lakes. Freshw Sci 17(1):73–80

Schneider JC, Alexander GR, Merna JW (2000) Modules for lake and stream surveys. In: Scheider JC (ed) Chap. 2: manual of fisheries survey methods II: with periodic updates. Michigan Department of Natural Resources, Fisheries Special Report 25, Ann Arbor

Scott WB, Crossman EJ (1973) Freshwater fishes of Canada. Bulletin 184 of the Fisheries Research Board of Canada

Seitz RD, Lipcius RN, Olmstead NH, Seebo MS, Lambert DM (2006) Influence of shallow-water habitats and shoreline development on abundance, biomass, and diversity of benthic prey and predators in Chesapeake Bay. Mar Ecol Prog Ser 326:11–27

Small C, Nicholls RJ (2003) A global analysis of human settlement in coastal zones. J Coast Res 19(3):584–599

Sobocinski K, Cordell J, Simenstad C (2010) Effects of shoreline modifications on supratidal macroinvertebrate fauna on Puget Sound, Washington Beaches. Estuar Coasts 33(3):699–711

Strayer DL, Dudgeon D (2010) Freshwater biodiversity conservation: recent progress and future challenges. Freshw Sci 29(1):344–358

Strayer DL, Findlay SG (2010) Ecology of freshwater shore zones. Aquat Sci 72:127–163

Strayer DL, Findlay SG, Miller D, Malcom HM, Fischer DT, Coote T (2012) Biodiversity in Hudson River shore zones: influence of shoreline type and physical structure. Aquat Sci 74:597–610

Teixeira T, Neves L, Araújo F (2009) Effects of a nuclear power plant thermal discharge on habitat complexity and fish community structure in Ilha Grande Bay, Brazil. Mar Environ Res 68(4):188–195

Tickner T, Opperman J, Abell R, Acreman M, Arthington AH et al (2020) Bending the curve of global freshwater biodiversity loss—an emergency recovery plan. BioScience 70(4):330–342

Toft JD, Cordell JR, Simenstad CA, Stamatiou LA (2007) Fish distribution, abundance, and behaviour along city shoreline types in Puget Sound. N Am J Fish Manag 27:465–480

Tonn W, Magnuson J (1982) Patterns in the species composition and richness of fish assemblages in Northern Wisconsin lakes. Ecology 63(4):1149–1166

Vadeboncoeur Y, McIntyre P, Vander Zanden M (2011) Borders of biodiversity: life at the edge of the world's largest lakes. Bioscience 61(7):526–537

Rideau Valley Conservation Association (RVCA) (2014) Rideau Lakes subwatershed report: Big Rideau Lake—portland catchment. Rideau Valley Conservation Authority, pp 1–19

Valley R, Bremigan M (2002) Effect of macrophyte bed architecture on largemouth bass foraging: implications of exotic macrophyte invasions. Trans Am Fish Soc 131(2):234–244

Vitousek PM, Mooney HA, Lubchenco J, Melillo JM (1997) Human domination of Earth's ecosystems. Science 277(5325):494–499

Warren Jr M.L (2009) Centrarchid identification and natural history. In: Centrarchid fishes: diversity, biology, and conservation. Blackwell Publishing, Oxford, pp 375–534

Wensink SM, Tiegs SD (2016) Shoreline hardening alters freshwater shoreline ecosystems. Freshw Sci 35(3):764–777

Whitfield AK (1984) The effects of prolonged aquatic macrophyte senescence on the biology of the dominant fish species in a southern African coastal lake. Estuar Coast Shelf Sci 18(3):315–329

Zolderdo AJ, Abrams AEI, Reid CH, Suski CD, Midwood JD, Cooke SJ (2019) Evidence of fish spillover from freshwater protected areas in lakes of eastern Ontario. Aquat Conserv 29(7):1106–1122