Science of the Total Environment 953 (2024) 176088

Contents lists available at ScienceDirect

Science of the Total Environment

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

Evaluating the efficacy of ecological restoration of fish habitat in coastal waters of Lake Ontario

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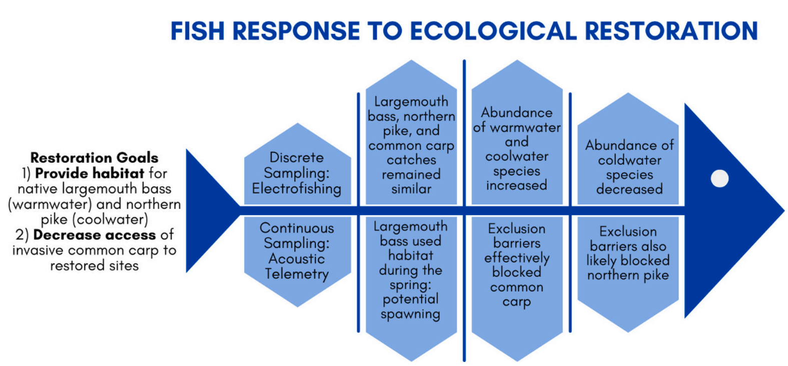
HIGHLIGHTS

- Ecological restoration of freshwater wetlands resulted in species-specific responses.
- Coldwater species decrease, while coolwater and warmwater species increased.
- Restored habitat provided habitat for largemouth bass, particularly in spring during potential spawning.
- Exclusion barriers effectively decreased access to non-native common carp, but also impeded movements of northern pike.

GRAPHICAL ABSTRACT



ARTICLE INFO

Editor: Rafael Mateo

Keywords:
Restoration ecology
Conservation
Spatial ecology
Fisheries
Environmental management

ABSTRACT

Ecological restoration is a common strategy applied to degraded wetlands and tributaries in large lakes. As resources are typically limited for restoration, it is essential to ensure that such efforts achieve associated goals. Using both discrete and continuous methods, we evaluated the efficacy of ecological restoration efforts on fish habitat within Canada's largest city, Toronto (Cell 2 and Embayment D of Tommy Thompson Park) relative to a control site (Toronto Islands). First, we used a long-term electrofishing dataset (i.e., discrete) to examine catch and community composition relative to restoration status. Catch for northern pike (*Esox lucius*) remained constant at both restoration sites, and catch of invasive common carp (*Cyprinus carpio*) decreased at Embayment D, indicating that exclusion barriers may be effective. Restoration was less effective for largemouth bass (*Micropterus nigricans*) as catches remained similar after restoration at Cell 2, but decreased within Embayment D. We also found that relative abundance for coldwater species at both restoration sites decreased post-restoration, with increases in warmwater species at Cell 2 and coolwater species at Embayment D. Next, we used a long-term acoustic telemetry dataset (i.e., continuous sampling) with three focal species: largemouth bass, northern pike, and invasive common carp. Based on telemetry, we found that restoration efficacy was species-specific, with largemouth bass present before and after ecological restoration (particularly in spring, which may be

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https://doi.org/10.1016/j.scitotenv.2024.176088
Received 12 March 2024; Received in revised form 3 September 2024; Accepted 4 September 2024
Available online 10 September 2024
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associated with spawning), but clear reductions in use of the restored areas for common carp and northern pike. Exclusion barriers, while effective at blocking common carp, appeared to also negatively influence access for northern pike. Using both discrete and continuous methods longitudinally and across both treatment and control sites provided complementary information on the efficacy of restoration works within Toronto Harbour, with electrofishing data highlighting changes in fish community composition while acoustic telemetry provided continuous information on timing and duration of habitat use.

1. Introduction

In response to anthropogenic habitat alterations, environmental managers across the world have turned to ecological restoration, which broadly aims to conserve biodiversity through deliberate management efforts to return the system to a more natural state (Hobbs and Norton, 1996; Jordan et al., 1988). As resources are typically limited for restoration, it is essential to ensure that such efforts achieve associated goals. Further, there remains work to be undertaken to produce robust, evidence-based data in support of its efficacy, particularly in freshwater (Cooke et al., 2019). Broadly, impacts of restoration projects are rarely evaluated (Palmer and Stewart, 2020) and population-level benefits remain unclear (McLean et al., 2015; Radinger et al., 2023). Without these evaluations, environmental managers are left with minimal evidence as to the success of such efforts and how to improve best practices to ultimately benefit biodiversity (Wortley et al., 2013). With billions of dollars spent on the restoration (BenDor et al., 2015), it is essential to ensure restoration activities achieve associated objectives and targets with ecological monitoring.

Unfortunately, as restoration projects are developed and implemented, ecological monitoring is rarely prioritized/resourced to assess efficacy or suffers from poor design (Block et al., 2001). Monitoring of ecological restoration is essential because it permits the assessment of restoration outcomes relative to prior conditions and is crucial for improving best practices (Gann et al., 2019; Wortley et al., 2013). To advance the evidence-base, rigorous monitoring should be designed as true experiments to allow for stronger inferences (Block et al., 2001) and conducted throughout all phases of restoration (i.e., before, during, and after). In addition to monitoring longitudinally, it is crucial to incorporate the use of controls, which allow for a comparison between restored and non-restored sites (e.g., Before-After-Control-Impact; see Smokorowski and Randall, 2017). This can isolate the effects of the ecological restoration and provide a more complete understanding of the impacts. Such comprehensive monitoring for assessments is particularly crucial for the ecological restoration of freshwater fish habitat (Lapointe et al., 2013), where globally, populations are facing steep declines mainly attributed to habitat alterations (WWF, 2021).

Within freshwaters, ecological restoration of fish habitat can be assessed using both discrete such as electrofishing and continuous sampling including acoustic telemetry. First, discrete sampling uses gear such as gillnets, trapnets, trawls, and electrofishing to provide detailed information about a single sampling event. For example, single metrics could include abundance, while multimetric indices could provide estimates of community composition (Lorenzen et al., 2016). Next, continuous sampling could include methods used to assess behavior and spatial ecology, such as acoustic telemetry (Brooks et al., 2019). Acoustic telemetry involves the placement of electronic tags on wild animals which subsequently transmit data to receiver stations (Cooke et al., 2016; Hussey et al., 2015). The benefit of this approach is that fish are monitored longitudinally (i.e. over multiple seasons) as opposed to a discrete snapshot in space and time (Heupel et al., 2006). Taken together, combining both discrete approaches such as community indices from electrofishing and habitat use derived from acoustic telemetry (i.e., continuous monitoring) has the potential to provide a more complete understanding of ecological restoration efficacy within freshwater ecosystems such as those within the Laurentian Great Lakes (see Brooks et al., 2017).

The Great Lakes and particularly the densely populated Lake Ontario have been subjected to extensive anthropogenic degradation, especially along the coasts and nearshore habitat. For example, 70 % of all ecologically important coastal wetlands have been lost along the northern shore of Lake Ontario (Whillans, 1982). Further, the majority of remaining aquatic habitat has been subject to anthropogenic degradation since human settlement (Brinson and Malvárez, 2002; Niemi et al., 2007). These important ecosystems play an integral role for freshwater fish providing habitat for spawning, foraging, nursery, and refugia (Jude and Pappas, 1992). Unfortunately, the health of fish populations and quality of associated habitat has been jeopardized due to anthropogenic activities. The most populous urban center within the Great Lakes is Toronto, Canada, located in the western portion of Lake Ontario, and it is associated with an area of extensive destruction and degradation wherein Toronto Harbour's ecosystem has lost an estimated 400 ha of aquatic habitat (Whillans, 1979). As suitable habitat is integral to supporting fish populations, the loss and alteration of these habitats has resulted in degradation of native fish populations (Midwood et al., 2022). Looking to the future, the well-established urban center of Toronto will continue to grow. As such, scientists and practitioners should work collaboratively to use ecological restoration as one key tool to enhance fish habitat that can support healthier fish populations (see Piczak et al., 2022).

There has been extensive ecological restoration throughout Toronto Harbour to recreate coastal wetland habitat that had been historically lost and to provide warmwater and coolwater (i.e., 28 and 24 °C thermal guilds, respectively) habitats to support overwintering, refugia, nurseries and foraging (Murphy et al., 2012). Another goal of the restoration was to decrease access of invasive common carp (*Cyprinus carpio*) to these restored sites to limit their potential negative ecological impacts (e.g., uprooting aquatic vegetation, the impairment of water quality, and altered fish community; Weber and Brown, 2009). Restoration efforts were also intended to provide habitat for specific native species including warmwater preferring largemouth bass (*Micropterus nigricans*) and coolwater, wetland spawning northern pike (*Esox lucius*). There has been some preliminary evidence suggesting that restoration actions in Cell 1 and Cell 2 were successful (i.e., increased piscivore and forage catch-per-unit-effort; Barnes et al., 2020); however, these works include data from only before and during restoration efforts, not after.

The purpose of our research was to assess the efficacy of ecological restoration on fish populations within Toronto Harbour (TH). Specifically, we used Cell 2 and Embayment D within TTP as treatment sites (i.e., where major ecological restoration was undertaken), with the Toronto Islands acting as a control. First, we used a long-term electrofishing dataset to examine trends in catch and community across restoration periods. Specifically, we compared the catch of largemouth bass, northern pike, and common carp within the treatment and control sites, before and after restoration. Next, we investigated differences in community traits based on relative abundance before and after restoration with specific focus on warmwater and coolwater fishes that were thought to benefit most from the restoration actions. Then, using a long-term acoustic telemetry dataset, we investigated the impacts of ecological restoration on the timing and duration of habitat and depth use for two native fishes, largemouth bass and northern pike, as well as one invasive species, common carp. All three of these species use coastal wetlands at some life stage, typically for spawning or foraging. The approach used here and associated findings have the potential to inform restoration efforts throughout the Great Lakes and beyond to other freshwater ecosystems that can benefit native fish populations.

2. Methods

2.1. Field methods

2.1.1. Study sites

Lake Ontario, the most easterly of the Laurentian Great Lakes, has been subjected to anthropogenic activity for over 200 years, particularly in the densely populated western portion, which is home to the City of Toronto (43.631–79.369; Fig. 1; Allan et al., 2013). Toronto, with a population of over five million people in the greater metropolitan area, has experienced widespread loss of littoral and wetland habitat along its waterfront, mainly owing to infilling in support of urbanization, and the expansion of industry (Barnes et al., 2020). Due to historic and ongoing anthropogenic disturbance, habitat impairment and loss, TH was identified as an Area of Concern (AOCs) in 1985 and remains the focus of considerable remediation efforts (Midwood et al., 2021). TH is a heavily urbanized area comprised of four zones: Inner Harbour, Toronto Islands, Outer Harbour, and Tommy Thompson Park (TTP). The Toronto Islands are relatively natural compared to the rest of the harbour (i.e., high percentage coverage of submergent aquatic vegetation) and therefore have not been the focus of major restoration efforts, although some smaller habitat enhancement works have been completed (i.e., 1.5 ha of created wetlands; Barnes et al., 2020). In contrast, there has been extensive and on-going restoration throughout three Cells (1 through 3) and the four Embayments (A through D) of TTP (Barnes et al., 2020). Specifically, within Cell 2 from 2015 to 2017 (Table 1; Fig. 1), 6.70 ha of area was restored with the addition of a wetland berm, common carp exclusion barrier (vertical bars spaced 5 cm apart), shoreline shoals, log tangles and anchored logs (for habitat structure), reptile habitats (e.g., basking structures), aquatic vegetation (bulrush, cattails, fragrant water lily, bur-reed), and lowland riparian woods. During 2015, only the eastern half of Cell 2 was hydrologically disconnected, therefore fish were able to use the western side and access Cell 1; however, both cells became inaccessible starting April 2016 and were reconnected March 2021. The average depth within Cell 2 prior to restoration was 2.37 m, compared to 1.37 m after. For Embayment D, from 2012 to 2014 (Table 1), 5.96 ha of habitat (aquatic and riparian) was restored (Fig. 1) and restoration techniques included the addition of wetland berms and a common carp exclusion barrier, as well as increases in shoreline complexity, aquatic vegetation, and riparian vegetation zones. Although after the restoration activities Embayment D was hydrologically connected in 2015, there were beavers (*Castor canadensis*) that continually constructed dams (in 2018 and 2019), thereby likely blocking fish movements into Embayment D. During both 2017 and 2019, highwater levels in Lake Ontario resulted in flooding throughout TTP including both restoration sites, as well as the Toronto Islands (Table 1). The average depth within Embayment D prior to restoration was 1.06 m compared to 1.25 m after. As the Toronto Islands have not received major habitat restoration and have remained relatively natural compared to the rest of TH, we used this site as a control.

2.1.2. Electrofishing

Population trends for largemouth bass, northern pike, and common

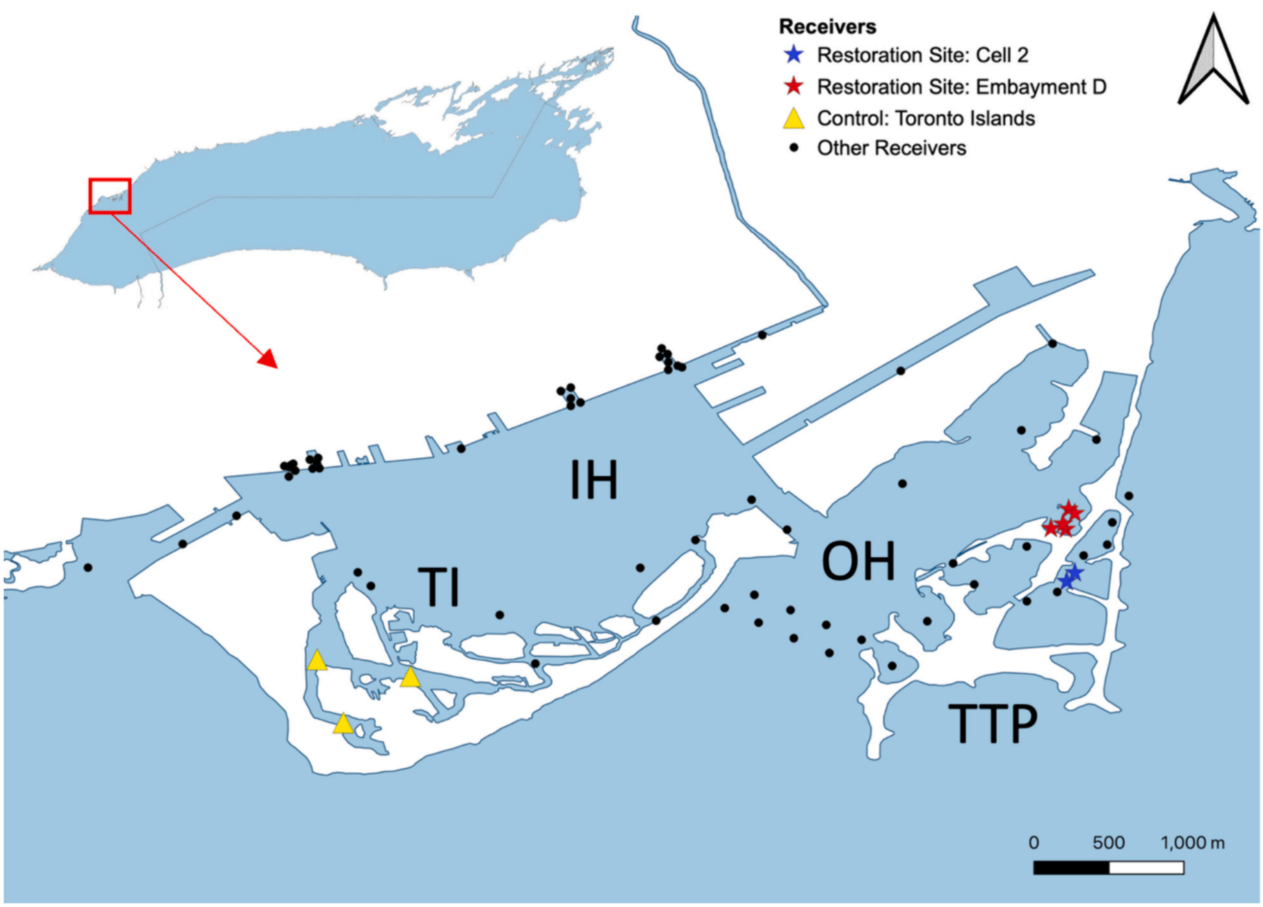
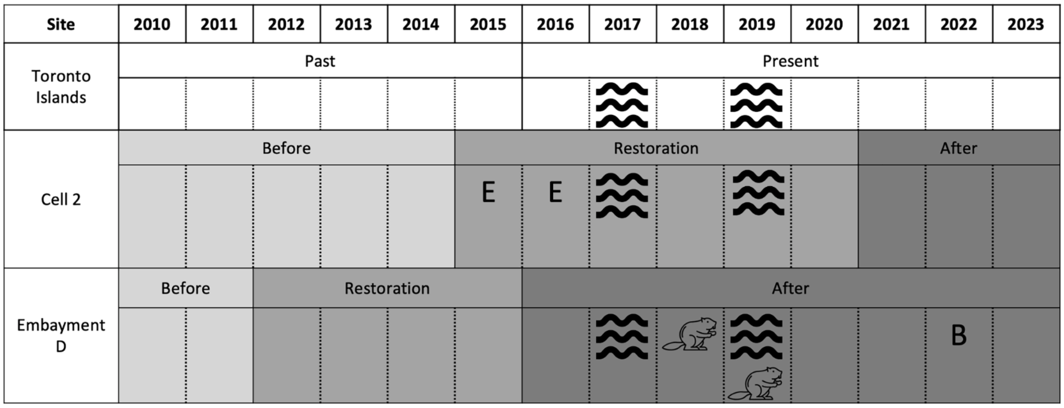


Fig. 1. Within Lake Ontario (red square inset), Toronto Harbour is located on the northern shore of the western portion of Lake Ontario. Toronto Harbour is comprised of four zones: Toronto Islands (TI), Inner Harbour (IH), Outer Harbour (OH), and Tommy Thompson Park (TTP). TI acted as a control, while Cell 2 and Embayment D within TTP received ecological restoration of fish habitat. Receivers were deployed throughout the harbour to cover multiple habitat types and movement corridors, including both the control and restoration sites. Electrofishing transects also occurred within these locations.

Table 1
Timeline and details of major restoration activities within Toronto Harbour by site. Restoration was considered complete once the site was hydrologically connected to the rest of the harbour (i.e., stoplogs were removed to permit waterflow and fish passage). The Toronto Islands (TI) served as a control site as minimal restoration activities were conducted, while both Cell 2 and Embayment D of Tommy Thompson Park received major restoration. During 2015 and 2016, only the eastern (E) portion of Cell 2 was hydrologically disconnected. Water (waves) indicates years of flooding and highwater, and beavers indicate years where sites were potentially blocked by beaver dams and hydrologically disconnected. In 2022, there was a beaver cone (B) installed in Embayment D to permit water connectivity.



carp, and changes in community traits for the Toronto Islands, Embayment D, and Cell 2 between the years of 2008 to 2022. The Toronto and Region Conservation Authority (TRCA) conducted 200 boat electrofishing runs conducted in the summer and fall by using either a 4.3, 5.5, or 6.4 m, 7.5 kW pulsed DC electrofishing boat (Smith-Root; Mount Vista, USA; Fig. 1B). There were 147 runs at the Toronto Islands, 28 at Embayment D, and 25 at Cell 2. Effort for each run was recorded in shock seconds, ranging from 686 to 1002 s. Fish data were available as catch in each run, individuals were identified to species, and their catch was standardized to 1000 shock seconds, which is the standard run length for the local monitoring program (Barnes et al., 2020).

2.1.3. *Telemetry array*

A biotelemetry array has been deployed in TH since spring 2011 (Fig. 1B), and a variety of fish species have been implanted with acoustic transmitters (Midwood et al., 2019). Key movement corridors, as well as various habitat types were strategically instrumented with VR2W 69 kHz acoustic receivers (Innovasea, Bedford, Nova Scotia, Canada). Since 2011, array coverage has varied due to the loss of receivers or expansion of coverage into new areas of interest (see Supplemental 1 for receiver details). Receivers were combined into 37 groups based on habitat consistency/proximity (Midwood et al., 2019), as well as range-testing results (conservative estimate of 350 m; see Veilleux 2014). Detections in TH were available from spring 2011 to spring 2023, and this entire period was included in the analysis.

2.1.4. *Fish capture and tagging*

Largemouth bass, northern pike, and common carp were captured within TH between 2010 and 2021 using the same model electrofishing boats as the fish community surveys (Table 2). After capture, fish were placed in live wells with ambient lake water and either transported to shore or surgery was conducted on the vessel while floating. Fish were immobilized for surgery using a Portable Electroanesthesia System (Smith-Root, Mount Vista, Washington, USA) and some were anesthetized using clove oil (Rous et al., 2015). Next, fish were placed supine in a trough with ambient lake water passed over the gills to aid respiration. All surgical tools and acoustic transmitters (V13, V13P, and V13TP, Innovasea, Bedford, Nova Scotia, Canada; see Supplemental 2 for additional transmitter details) were disinfected with an iodine solution and rinsed. An incision (~20 mm) was made with a scalpel and the transmitter was inserted into the coelom. Incisions were closed with two or three interrupted sutures. Fish size (total length) was measured,

Table 2
Number of individuals tagged per year and number of individuals detected with mean total length (TL) and range (mm), presented within Toronto Harbour per species per year. Fish were typically tagged during the spring or fall.

Year	Largemouth bass			Northern pike			Common carp		
	n tagged	*n* detected	TL	*n* tagged	*n* detected	TL	*n* tagged	*n* detected	TL
2010	17	15	440 (301–497)	17	15	812 (476–965)	17	16	703 (615–854)
2011	18	31	444 (301–500)	18	29	707 (476–965)	0	15	720 (615–854)
2012	20	44	448 (305–535)	20	43	747 (476–1003)	20	31	651 (470–854)
2013	53	59	397 (156–535)	51	56	718 (250–1003)	14	28	575 (470–741)
2014	5	35	332 (197–478)	7	40	551 (250–972)	0	29	558 (470–741)
2015	15	27	367 (244–508)	24	44	673 (254–972)	8	29	588 (470–750)
2016	14	23	387 (244–495)	15	36	716 (510–960)	7	22	660 (497–805)
2017	17	29	362 (253–490)	17	27	800 (491–960)	9	21	704 (520–805)
2018	4	27	356 (253–490)	8	26	785 (510–920)	6	13	676 (448–805)
2019	0	17	387 (253–490)	0	26	763 (562–920)	0	11	633 (448–785)
2020	0	11	373 (310–489)	0	7	687 (648–897)	0	12	626 (448–785)
2021	17	21	343 (218–451)	17	12	613 (260–728)	16	22	705 (630–930)
2022	0	15	302 (218–454)	0	12	582 (260–728)	0	20	731 (630–930)
2023	0	4	356 (240–405)	0	4	500 (274–563)	0	8	708 (630–795)
Total	180	144	391 (156–535)	194	144	711 (250–1003)	97	82	650 (448–930)

and fish were returned to a live well with circulating lake water. Fish were released at their point of capture after ensuring full recovery. Fish handling and surgical procedures were approved and followed a Canadian Council on Animal Care protocol administered by Carleton University (Certificate CU 110723).

2.2. Analyses

2.2.1. Water temperature: seasonal delineation & loggers

All analyses were completed in R Studio (version 1.1.456; R Core Team; 2021). Water temperature transitions were used to define seasons (after Larocque et al., 2020) using temperature-profile data collected from a chain of temperature loggers deployed nearby in Lake Ontario (Ajax, Ontario; 43.461–78.584). We delineated seasons by taking an average of the temperature logger profile: spring started when water temperatures first exceeded 5 °C, until they surpassed 15 °C, which was then classed as summer. Fall occurred when temperatures consistently decreased below 15 °C until falling below 5 °C, which was categorized as winter (Supplemental 3). Temperature loggers (Hobo Pro v2 model, Bourne, Massachusetts, USA) were also deployed throughout each of the control and restoration sites 1 m below the surface, anchored with a cinder block to monitor water temperatures over time (Fig. 1). The mean water temperature in the summer was estimated for each site to assess changes throughout restoration periods.

2.2.2. Restoration delineation

For both acoustic telemetry and electrofishing data, Cell 2 and Embayment D were divided into three periods of time according to their restoration status: before, during, and after (Table 1). Data from the Toronto Islands was split into two broad periods to provide points of comparison: past (2008 to 2015) and present (2016 to 2022) for electrofishing data, and past (2011 to 2015) and present (2016 to 2023) for telemetry data.

2.2.3. Telemetry data collection and processing

Data from the TH telemetry array were downloaded biannually once in spring and fall. Erroneous detections were removed if they met criteria for false-positive detections (single occurrences with >3600 s between successive detections; (Pincock et al., 2012)). The dataset was also filtered to remove fish that died or expelled their transmitters, which was presumed to have occurred when consistent depth profiles and/or locations were evident for an extended period (Klinard and Matley, 2020). Additionally, fish that were detected for <14 days total were removed from the dataset to eliminate those that died following surgery or had malfunctioning tags.

2.2.4. Restoration efficacy: electrofishing

Trends in catch of largemouth bass, northern pike, and common carp were analyzed for all three species. For consistency across both electrofishing and telemetry approaches, we examined catch of larger individuals that were the minimum size or larger of the acoustically tagged fishes (i.e., common carp >448 mm; northern pike >250 mm; largemouth bass >156 mm; see Table 2). However, we also explored trends in catch of individuals smaller than these sizes for all three species. Catch data were converted to catch-per-unit-effort (CPUE; catch/1000s) and mean CPUE was compared across restoration periods for each restoration site and across time (i.e., past and present) for the control site (Toronto Islands) through Kruskal-Wallis tests (alpha < 0.05) and pairwise comparisons for the restoration sites (Dunn's test if H0 rejected, Holm-Bonferroni adjusted p-values; (Dinno, 2015)). Non-parametric tests are useful when working with skewed data, small sample sizes, and potential outliers. However, they also tend to be less statistically powerful in their ability to detect smaller significant changes between groups or treatments (Harwell, 1988; Thomas and Juanes, 1996).

Community changes were evaluated based on relative abundance per species and species traits (Hellinger-transformation). To capture community changes, we calculated relative abundance, which measures the proportion of each species within a sample run (square root of sample total standardized data; Legendre and Gallagher, 2001). We then assessed the fish community in an average electrofishing run across time for the Toronto Islands and for each treatment site before and after restoration. Using an average electrofishing run can be helpful in cases with high catch differences among runs and with highly abundant species that can inflate overall catch (e.g., alewife, *Alosa pseudoharengus* caught in bottom trawls; Weidel et al., 2017). Species traits that were considered included preferred environment, maximum adult length, feeding guild, tolerance (ability to adapt to environmental perturbations or anthropogenic stressors; e.g., brown bullhead, alewife, common carp, and white perch; Boston et al., 2016), thermal guild, and preferred habitat. These were chosen since they reflected components of the fish community that were likely to respond to ecological restoration (Table 3). Trait contribution to total Hellinger-transformed abundance was plotted in a heatmap and compared for each period (before and after; adonis and pairwise comparison; accepted Alpha < 0.05; vegan 2.6–4; Oksanen et al., 2022). Adonis, as a function that calculates distance-based permutational approach, is commonly used in the analysis of ecological community data (e.g., species or trait matrices) and can robustly describe how variation is attributed to different experimental treatments, including restoration status in the context of this study (Legendre and Gallagher, 2001; Oksanen et al., 2022). We noted individual species and trait changes exceeding 5 %, regarding contribution to relative abundance, when comparing before and after restoration communities. Functional trait dominance and composition across spatial and temporal scales has become a well-established and more commonly applied approach to assess ecosystem health and restoration success (Mouillot et al., 2013; Stuart-Smith et al., 2013).

2.2.5. Restoration efficacy: acoustic telemetry

To examine habitat use of control and restored sites, we counted the total number of individual largemouth bass, northern pike, and common carp detected per week at each site. From there we calculated the weekly proportion of individuals detected, which was the total number of individuals detected per week at each site over the total number of individuals active within the array per year for each species (see Table 2).

Table 3
Species traits as indicators (proxy) for community changes across study sites and restoration status and key references.

Trait	Sub-trait	Proxy	Key reference
Environment	Pelagic; Benthopelagic; Benthic	Effect of restoration on habitat use and nearshore species vs offshore species	(Hoyle et al., 2018)
Maximum length	Small; Medium; Large; Very large	Effect of access restriction measures (e.g., carp gate)	(Piczak et al., 2023b; Rahel and McLaughlin, 2018; Piczak et al., 2023c)
Feeding guild	Piscivore; Non-piscivore	Trophic balance; restoration goal of increasing piscivore presence	(Drenner and Hambright, 2002; Potthoff et al., 2008)
Tolerance	Tolerant; Intermediate; Intolerant	Restoration effect on sensitive and potentially rare species	(Boston et al., 2016)
Thermal guild	Warmwater; Coolwater; Coldwater	Embayment D and Cell 2 serve as warmwater refugia	(Murphy et al., 2011, 2012)
Habitat	Vegetation and cover preference; No vegetation and cover preference	Effect of habitat provision and planting in Embayment D and Cell 2	(Trebitz and Hoffman, 2015)

Next, residency index (RI) is often calculated as the number of days an individual fish was detected at each receiver or receiver group divided by the total number of days the fish was detected anywhere within the acoustic array (Kessel et al., 2016). However, in the present study, we calculated a modified seasonal RI for fish detected within each site, which was calculated as the time spent at a given receiver group, divided by the total length of a given season (as per Piczak et al., 2023a), using the residency function in the GLATOS package (Holbrook et al., 2016). We then calculated the mean RI for each season-year combination, across time and each restoration status. Lastly, we characterized depth use across time and before and after restoration for each species by calculating mean depth used per season for each site and all three species across restoration status.

3. Results

3.1. Water temperatures

At the Toronto Islands, there was minimal interannual variation in water temperatures over time, with mean annual summer water temperatures fluctuating between 19.1 °C (2017) and 21.8 °C (2021; Fig. 2A). The mean summer water temperature at Cell 2 before restoration was 20.2 °C and after increased to 22.1 °C (Fig. 2B). Similarly, the mean summer water temperature at Embayment D before restoration was 18.8 °C and increased to 22.2 °C after (Fig. 2C).

3.2. Electrofishing

3.2.1. Catch

CPUE for larger largemouth bass (>156 mm), northern pike (>250 mm) and common carp (>448 mm) did not change significantly over time at the Toronto Islands (Fig. 3A); however, during the same period catch decreased for individuals of both northern pike (<250 mm; 1.66 ± 3.42/1000s to 0.18 ± 0.58/1000s) and largemouth bass (<156 mm; 10.4 ± 23.1/1000s to 4.68 ± 8.23/1000s; $p < 0.01$), and remained similar for smaller common carp (<448 mm; 0.06 ± 0.23/1000s to 0.19 ± 0.65/1000s; Supplemental 4). CPUE at Cell 2 for largemouth bass, northern pike and common carp in all size classes did not change significantly with restoration status (Fig. 3B; Supplemental 4). Catch per unit effort at Embayment D for largemouth bass exceeding 156 mm decreased significantly from 0.75 ± 0.50/1000s before to 0.08 ± 0.28/1000s after restoration ($p < 0.05$), but did not change for largemouth bass smaller than 156 mm (Fig. 3C; Supplemental 4). CPUE for northern pike in all size classes did not change significantly across restoration status at Embayment D. Catch per unit effort at Embayment D for common carp (>448 mm) decreased significantly from 2.50 ± 3.32/1000s before restoration to 0.12 ± 0.33/1000s after restoration ($p < 0.05$; Supplemental 4). No common carp smaller than 448 mm were caught in electrofishing runs at Embayment D after restoration (Fig. 3C).

3.2.2. Community composition

Trait-based community assessments for the Toronto Islands, Cell 2, and Embayment D, showed no significant community changes over time (past, present; Toronto Islands) or across restoration status (before, during, after; Cell 2 and Embayment D; Supplemental 5). Despite the absence of statistically significant overall community changes, several smaller trait or species-specific changes exceeding 5 % were noted between fish communities before and after restoration in Cell 2 and Embayment D but not for the Toronto Islands (Fig. 4A). Relative abundance (Hellinger-transformed) showed that coldwater species (e.g., Atlantic salmon, *Salmo salar*; freshwater drum, *Aplodinotus grunniens*; and lake trout, *Salvelinus namaycush*) decreased in their respective relative abundance after restoration for Cell 2 (−19.0 %) and Embayment D (−15.1 %; Fig. 4A). The relative abundance of species with intermediate tolerance to environmental disturbances and anthropogenic stressors in Embayment D and Cell 2 increased (+6.6 % and +8.7 %, respectively) compared to tolerant ones. Communities in both restored areas had a lower relative abundance (~−8–16 %) of small bodied species after restoration. Non-piscivore species increased in Embayment D by around 7.1 % in relative abundance after restoration (with a 7.1 % decrease in piscivore species), with no major changes in piscivore to non-piscivore relation in Cell 2. Overall, after completion of restoration efforts, communities in Cell 2 and Embayment D were predominantly composed of mixed warmwater and coolwater (~88–100 %), phytophillic (~61–66 %), benthopelagic (~55–61 %) species with intermediate tolerance levels (~62–68 %).

Species-specific relative abundance and community composition changed more at the two restored sites compared to the Toronto Islands with a larger number of species experiencing (>5 %) shifts in relative abundance contribution, as well as species with changing presence-absence based on electrofishing runs (Fig. 4B). Alewife, round goby (*Neogobius melanostomus*), and white sucker (*Catostomus commersonii*) decreased from their average pre-restoration relative abundance of ~5 % respectively to not currently being detected in electrofishing surveys. Bluegill (*Lepomis macrochirus*) and brook silverside (*Labidesthes sicculus*) increased the most in their relative abundance after restoration at Cell 2 (+9.8 & +8.7 %, respectively; Fig. 4B).

3.3. Acoustic telemetry

3.3.1. Weekly proportion

Relative to the restoration sites, there was more consistent use over

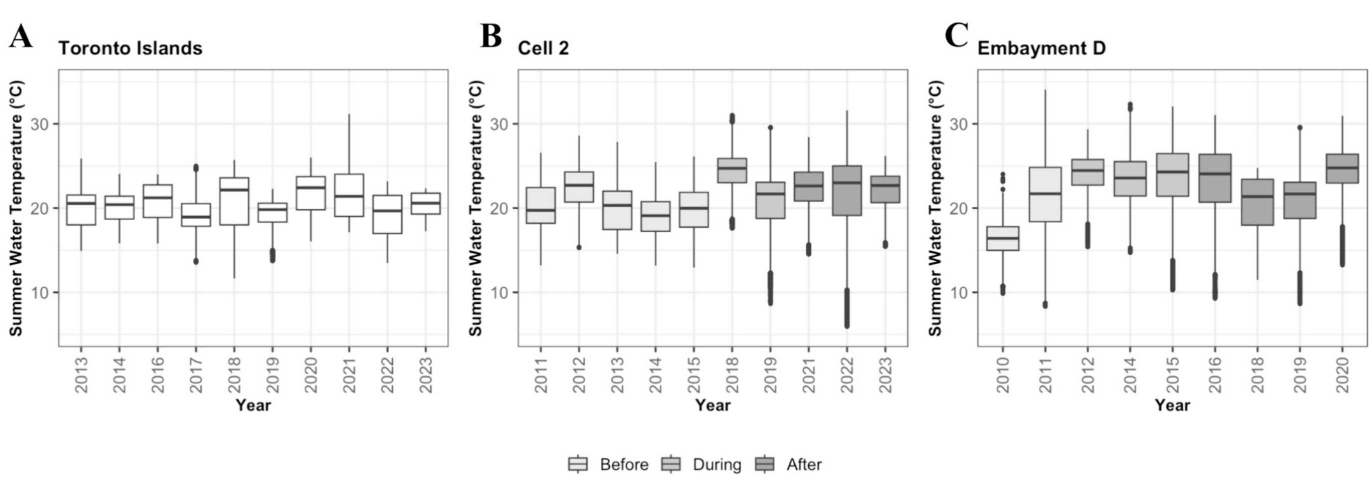


Fig. 2. Change in temperature across A) Toronto Islands (control site), B) Cell 2, and C) Embayment D (restoration sites).

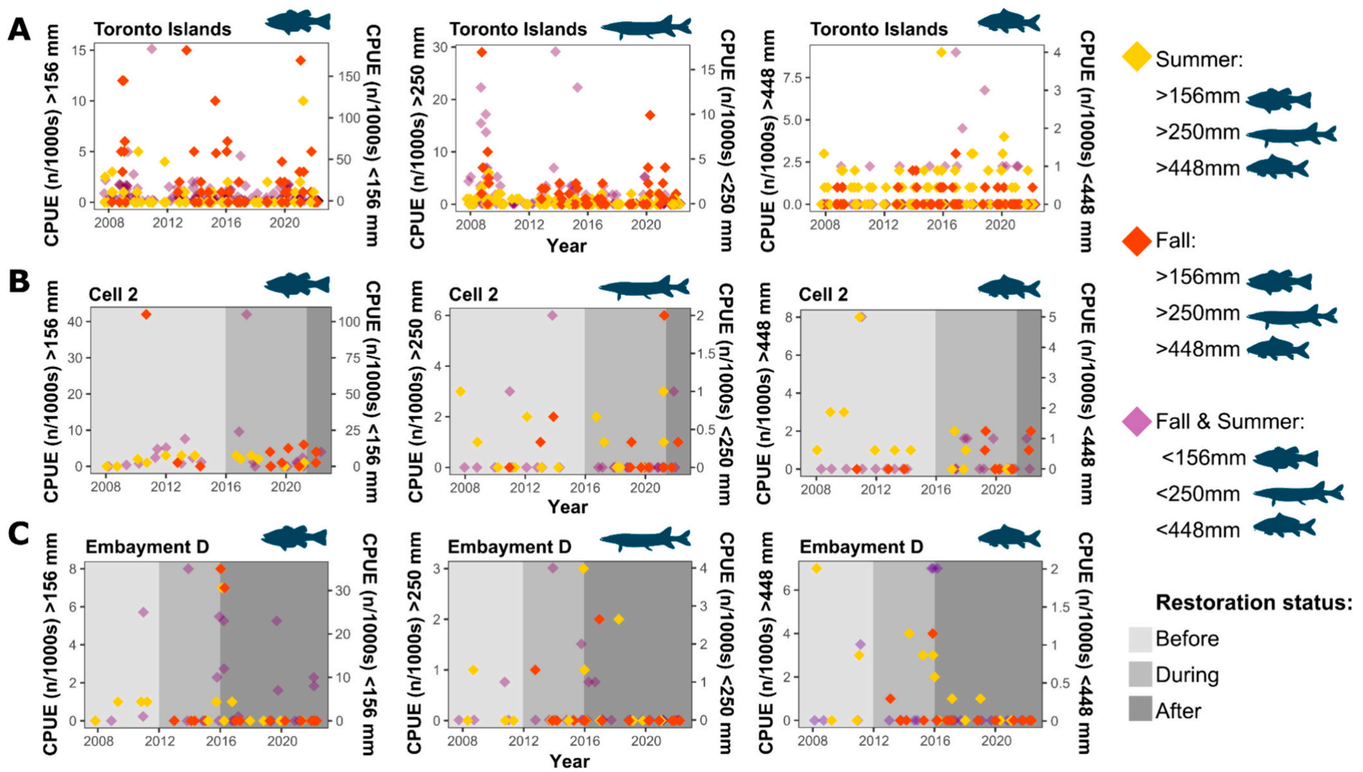


Fig. 3. Catch per unit effort (CPUE per 1000s electrofishing run) changes for largemouth bass >156 mm, northern pike >250 mm, common carp >448 mm in A) Toronto Islands (past, present) and B) Cell 2 and C) Embayment D across restoration status (before, during, after). Seasons (fall = red; summer = gold) and catches in smaller size classes (purple) are highlighted.

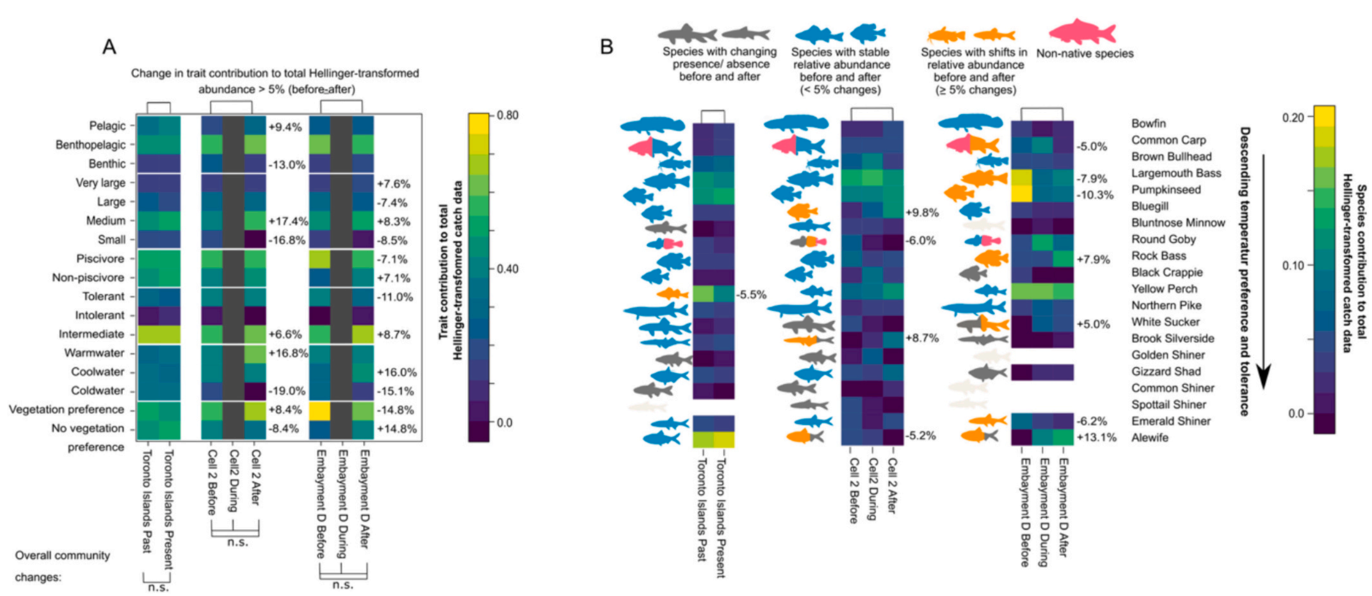


Fig. 4. A) Trait-based community changes for Toronto Islands (past, present) and Embayment D and Cell 2 across restoration status (before and after). Trait changes are based on trait contribution to total Hellinger-transformed abundance. Trait changes between before and after restoration exceeding 5 % are indicated. Overall community changes were compared through adonis analyses (permutational manova; accepted Alpha < 0.05). B) Species-based community changes for Toronto Islands (past, present) and Cell 2 and Embayment D across restoration status (before, during, after). Species changes and community dominance are based on species contribution to total Hellinger-transformed abundance. Species shifts (>5 % before and after restoration), presence/absence as well as invasive species are highlighted. Species contributing <2.5 % to total relative abundance were excluded.

time of the Toronto Islands by all three telemetry-tracked species (Fig. 5A). While the weekly proportion of individuals present decreased for largemouth bass after 2013, there was evidence of largemouth bass being present in the Toronto Islands in all seasons. Next, northern pike were also consistently present at the Toronto Islands across all years of study, with the weekly proportion of individuals present increasing over time (Fig. 5B). Northern pike were also present at the Toronto Islands across all four seasons, with spikes of increased presence during the

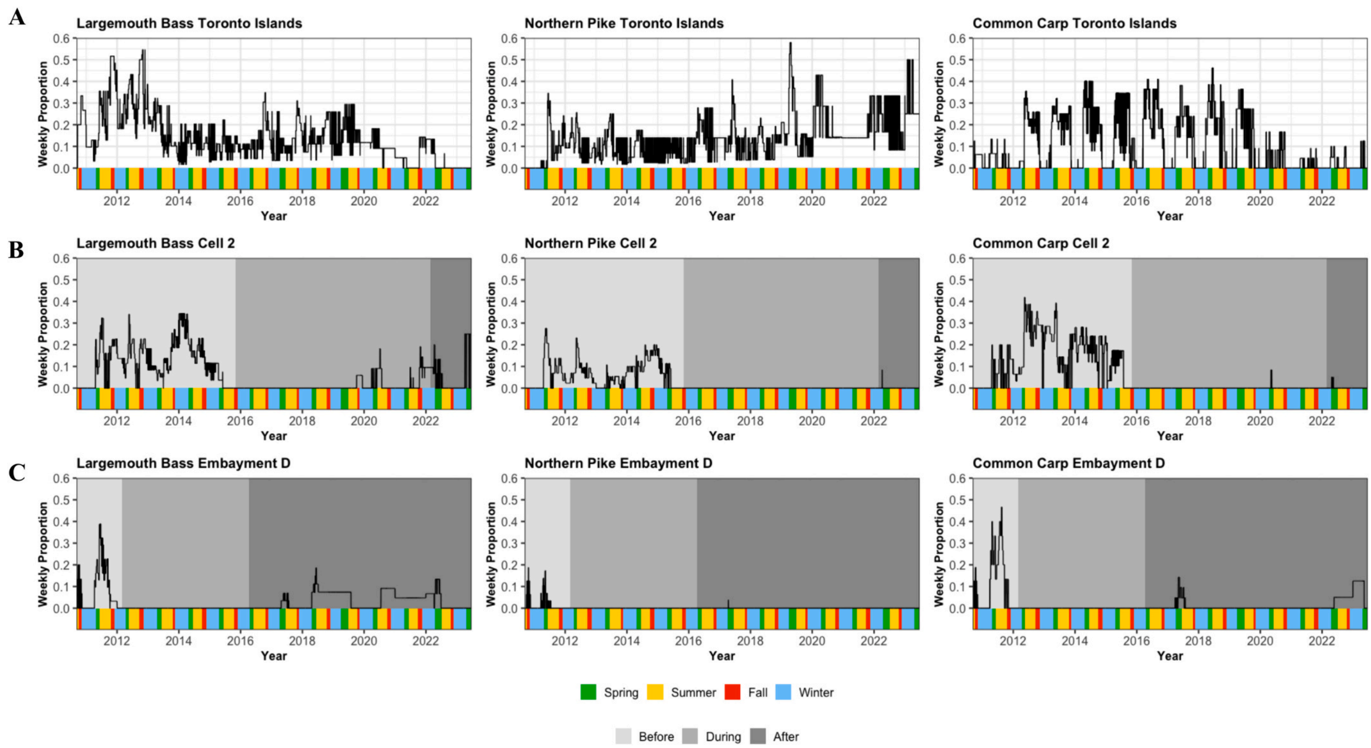


Fig. 5. The weekly proportion of tagged individuals present (out of total number of individuals in the system for that given year), across A) Toronto Islands, B) Cell 2, and C) Embayment D, across seasons, for each largemouth bass, northern pike, and common carp.

spring. Finally, common carp proportions increased in 2013 and remained consistent until approximately 2020 when they returned to pre-2013 levels (Fig. 5C). Seasonally, common carp were largely absent from the Toronto Islands during the winter, but were consistently present during spring, summer, and fall.

Before restoration, there was consistent presence of largemouth bass in Cell 2 across all four seasons accounting for up to 35 % of individuals present within the TH array (Fig. 5B). While the sample size did decrease for the after restoration period (see Table 2), the weekly proportion of largemouth bass detected at Cell 2 was less consistent and peaked sporadically at 25 % individuals present. Seasonally, largemouth bass were absent more frequently from Cell 2 after restoration with no individuals detected from early summer to late winter (although this a relatively short period of time). There was a minimal lag time associated with largemouth bass re-entering Cell 2 after restoration since tagged individuals were detected both before the site was fully opened (i.e., during restoration) and immediately after restoration was complete (Fig. 5B). Comparatively, the weekly proportion of both northern pike and common carp within Cell 2 decreased drastically after restoration as before restoration there were detections across all four seasons for both species (Fig. 5B). Specifically, only one individual for each northern pike and common carp were detected within Cell 2 after restoration and in both instances, this occurred during the spring and more than a year after restoration was complete (Fig. 5B). Taken together, the consistency of presence for both northern pike and common carp decreased after restoration in Cell 2.

Within Embayment D, the weekly proportion of largemouth bass present decreased after restoration relative to before and it took more than one year for the first individual to be detected (Fig. 5C). Specifically, largemouth bass had higher proportions of weekly individuals detected in Embayment D before restoration (particularly during the spring and summer); however, there was fairly consistent presence of largemouth bass after restoration, with spikes during the spring. Further, after restoration, largemouth bass were present in Embayment D across all four seasons in some years. Tagged largemouth bass, however, were completely absent from the system in some years (e.g., 2020). For northern pike, there was a higher percentage of individuals detected weekly before restoration (particularly during the spring) compared to after within Embayment D. Similar to Cell 2, only one northern pike was detected after restoration over one year after restoration was completed in Embayment D (during spring; Fig. 5C). For common carp, the weekly proportion of individuals present decreased after restoration compared to before; however, a couple of common carp were sporadically present across all four seasons (i.e., 2017 and 2023). Broadly, there were extended periods (i.e., multiple years at a time) when tagged common carp were absent from Embayment D after restoration.

3.3.2. *Residency index*

For all three species within the Toronto Islands, the mean RI was fairly consistent for each season between restoration periods (Fig. 6A). In both Cell 2 and Embayment D, the mean RI values for the after period were derived from smaller sample sizes than the before period, which means a single individual would have a greater influence on the final RI value during this period. For largemouth bass in Cell 2, the mean RI slightly increased after restoration during spring (albeit with a high standard deviation and small sample size), whereas the mean RI decreased after restoration for summer, fall, and winter, (with higher variation in winter; Fig. 6B). For both northern pike and common carp across all seasons, the mean RI decreased after restoration, as very few individuals accessed these sites and did not spend substantial time in Cell 2 (Fig. 6B). In Embayment D, for summer, the mean RI was similar for largemouth bass after restoration compared to before, the mean RI appeared to increase for both spring, fall, and winter after restoration (albeit with a high standard deviation and small sample size; Fig. 6C). For northern pike, both before and after restoration, the mean RI was very low in Embayment D, although it peaked in spring before restoration (Fig. 6C). For common carp, the mean RI decreased after restoration for spring and winter, while it increased slightly during the summer, albeit with very low sample sizes and high standard deviation (Fig. 6C).

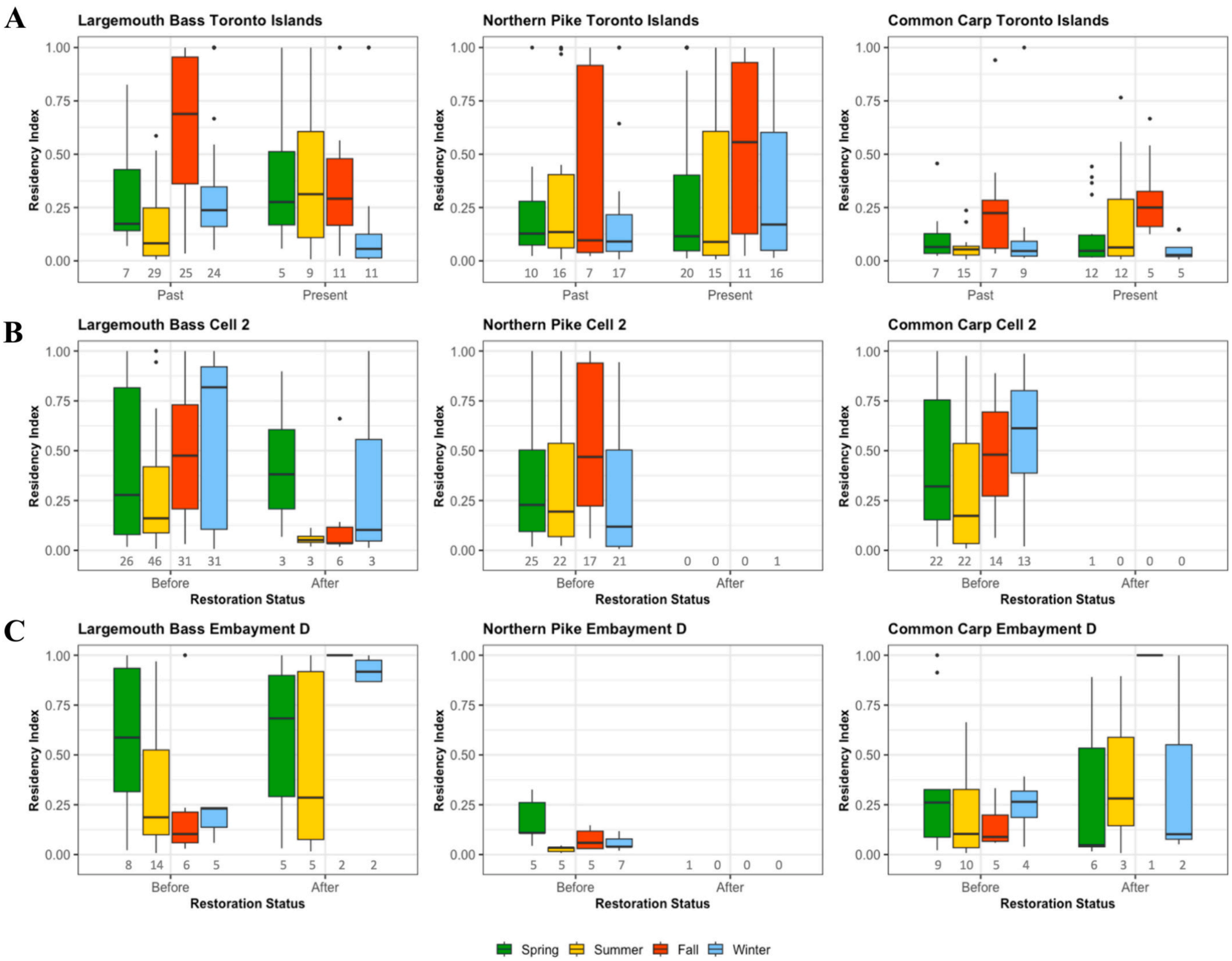


Fig. 6. Mean seasonal residency index of the fish that were detected within the sites for A) Toronto Islands, B) Cell 2, and C) Embayment D for each largemouth bass, northern pike, and common carp, with sample sizes per season indicated at the bottom of box.

There was one individual common carp that spent the entirety of one fall in Embayment D after restoration, leading to the increase in mean RI after restoration.

3.3.3. *Depth*

For all three species in the Toronto Islands, depth use was fairly consistent for each season between across time (Fig. 7A). Across all three species and both restoration sites, only largemouth bass within Cell 2 had a sufficient number of individuals detected with pressure transmitters to examine patterns of depth use after restoration. Specifically, before restoration, largemouth bass were detected up to 8 m deep within Cell 2, however after restoration, individuals were not detected deeper than 3 m (Fig. 7B). For largemouth bass mean depth use in both spring and summer remained similar across restoration periods, whereas for fall and winter, mean depth use was shallower after restoration (Fig. 7B). Post-restoration depth use patterns in the fall and winter for largemouth bass in Cell 2 were markedly different from both the before period and the before and after period at the Toronto Islands where largemouth bass showed progressively deeper depth use from the summer to fall to winter likely related to increases in water temperatures after restoration (see Fig. 2). Within Embayment D, there were insufficient sample sizes of fish with pressure sensors after restoration to compare depth use (Fig. 7C).

4. Discussion

We used both discrete and continuous sampling methods to assess if ecological restoration in TH achieved its goals of providing warmwater fish habitat, limiting access to invasive common carp, and providing suitable habitat for warmwater largemouth bass and coolwater northern pike. First, we found that while catch for largemouth bass, northern pike, and common carp did not change at Cell 2, catch for largemouth bass and common carp decreased and catch for northern pike remained similar after restoration at Embayment D. We also determined that both restoration sites saw declines in coldwater species with increases in warmwater species at Cell 2 and coolwater species at Embayment D. Using acoustic telemetry, we found that despite the lower weekly proportions after restoration (likely due to smaller sample sizes), there was still consistent presence of largemouth bass in both Cell 2 and Embayment D, particularly during the spring, which could be associated with spawning activities. Our results from acoustic telemetry also showed that common carp had decreased presence at restoration sites compared to the control site after restoration, which highlights the apparent efficacy of exclusion barriers. However, these exclusion barriers are not species-specific in that they can also block other large-bodied native species such as northern pike, which could explain their decrease in weekly proportion and mean RI after restoration for both Cell 2 and Embayment D. Here, we discuss the efficacy of fish habitat restoration in

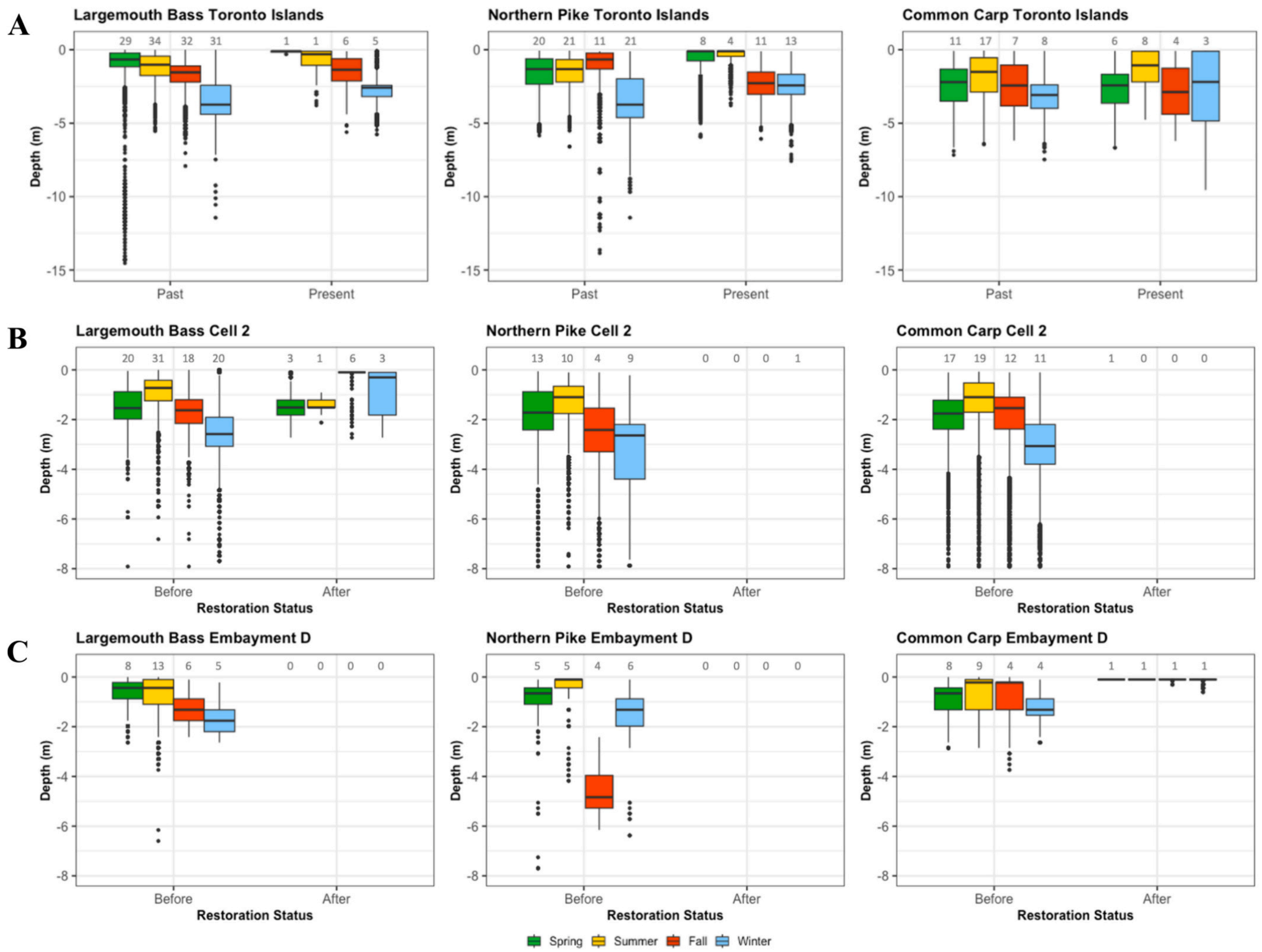


Fig. 7. Depth use of the fish tagged with pressure sensors that were detected for A) Toronto Islands, B) Cell 2, and C) Embayment D for each largemouth bass, northern pike, and common carp, with sample sizes per season indicated at the bottom of box.

TH, lag times associated with disturbance during restoration, potential limitations, avenues for future research, and management implications.

Exclusion barriers installed in both Cell 2 and Embayment D appear to be effectively decreasing catch and access of common carp (which was a restoration goal). However, these barriers may also be adversely impacting northern pike (and even largemouth bass), which is consistent with previous research (French et al., 1999), although not in all systems (see Wrubleski et al., 2024). Specifically, the vertical bar spacing of the exclusion barrier could be blocking other large-bodied native species (such as northern pike among others such as bigmouth buffalo, *Ictiobus cyprinellus*), which has been termed the connectivity conundrum (Zielinski et al., 2020). This connectivity conundrum could explain not only the decrease in habitat use for northern pike, but potentially the larger largemouth bass individuals as well, which could also explain the decreased presence after restoration at both sites. To mitigate the connectivity conundrum, exclusion barriers aimed at controlling invasive species such as common carp should be implemented with a selective fragmentation approach, whereby the barriers are designed and/or operated to allow passage for certain species, while undesirable species (i.e., invasives) are blocked (Rahel and McLaughlin, 2018). One such method is to operate barriers seasonally or based on within-year temperatures (Piczak et al., 2023c), which could permit northern pike unimpeded access to conduct their spawning movements since they occur earlier than common carp (Piczak et al., 2023b). Barriers could be left open earlier in the year to permit passage of northern pike and then lowered as common carp move into these sites, with the specific timing of closure linked to air temperatures, which has been found to be predictive of peak movements for both species (Piczak et al., 2023b). Broadly, our results are in agreement with previous research which has shown that exclusion barriers can be an effective tool to control common carp (reviewed in Piczak et al., 2023c; Wrubleski et al., 2024).

We determined that there were lag times after restoration where there was a delay in the detection of tagged fish at both restoration sites. It is well known that the timeline associated with species response to ecological restoration can be delayed (Watts et al., 2020), whereby it can take several years up to multiple decades to detect changes in biota (Ruiz-Jaen and Mitchell Aide, 2005; Marsden et al., 2016, 2023). This was evident within our study, reflected by the lag times, whereby all three species had minimal changes in catch and they also exhibited a delayed return to restoration sites after sites were hydrologically reconnected. Time lags have been suggested as one of the main factors limiting the apparent success of ecological restoration because of processes such as decreased availability of resources (Maron et al., 2015). Another explanation for delayed responses within TH could be disturbance or decreases in habitat quality associated with construction. It is also possible that the smaller changes that we did detect after restoration including decrease in catch for common carp in Embayment D and presence of largemouth bass at both sites during spring could indicate that the sites are in a state of transition, where the responses of fish are on-going and will increase as time continues after the ecological restoration. All of these factors associated with extended restoration timelines or lag times point to the importance of long-term monitoring to detect changes (i.e., 15–20 years; Choi, 2004).

Ecological restoration and assessment of efficacy are complex and biases could have arisen in our methods. For electrofishing, we had a limited number of runs for certain areas (i.e., Embayment D), which can introduce bias against certain species or smaller fish (Kanno et al., 2009; Polačik et al., 2008; Smith et al., 2015). Further, single runs in smaller areas or areas with high species richness can also increase the chance for bias or missing rare species (Glover et al., 2019). Although we did detect some changes in community traits across restoration, we only had electrofishing data from the summer and fall (i.e., no winter or spring), therefore potentially adding a seasonal bias. Next, the acoustic telemetry array in TH was deployed in 2010, building up coverage into 2011, which limited the amount of before time for Embayment D, where ecological restoration was initiated in 2012. On the other hand, we had a limited number of years for the after restoration period for Cell 2. It would have been beneficial to have multiple years for each restoration period: more time before restoration for Embayment D to establish a better baseline and more time after restoration for Cell 2 to allow more time for fish response. We also had challenges associated with small sample sizes for the number of fish acoustically tagged, particularly after restoration, which was mainly due to limited field work during the pandemic. Finally, while we examined sites and time periods associated with major restoration actions, there were other projects undertaken at the control site across the 13 year period (e.g., 1.5 ha of created wetlands, Barnes et al., 2020) which could have confounded our findings.

There are many potential avenues for future studies to advance our knowledge regarding ecological restoration. First, as previously discussed, the response of fishes to ecological restoration can occur on the order of decades (Ruiz-Jaen and Mitchell Aide, 2005), so it will be imperative to continue monitoring with both electrofishing and acoustic telemetry to continue to examine changes. Expanding trophic guilds beyond piscivore and non-piscivore species (e.g., invertivore, planktivore) and adding spawning guilds would add more depth to the study of restoration effects on community structure, as many restoration measures target declining specialist species with different life-history requirements (Jude and Pappas, 1992). In our study, we only tagged three large-bodied species, which together represent a small part of the fish community. These larger-bodied fishes are likely most affected by the presence of exclusion barriers, making their movements important to study, however, a focus on these larger fishes may underestimate the utility of the restored systems to the whole fish community. It would therefore be beneficial to examine the responses of other life stages and species groups (e.g., wetland resident or transitory fishes) to the restoration works. Further, due to transmitter size limitations, we did not track small species (e.g., other Centrarchid species) or juvenile fishes. Including these other species and life stages would provide a more holistic assessment of how fishes are responding to the restoration actions. The latter group, juvenile fishes, are particularly important given the critical role coastal wetlands play as nursery habitat for many Great Lakes fishes (Wei et al., 2004). As technological advances contribute to the miniaturization of acoustic transmitters (Lennox et al., 2017), we recommend tracking these smaller fishes in future studies. Further, although we did use acoustic transmitters with depth sensors, we had small sample sizes after restoration. In addition to depth sensors, using acoustic transmitters with other environmental sensors such as water temperature provide more information regarding habitat use to inform ecological restoration techniques including altering bathymetry and shoreline complexity, which are designed to provide thermal refugia (Crossin et al., 2017). While powerful, acoustic telemetry cannot confirm the specifics of animal behavior (i.e., explicit use), such as spawning activity, which could be examined with other complementary methods (e.g., diver surveys, Binder et al., 2016; Marsden et al., 2016). Further, it would be beneficial to investigate egg, larval, and age-1 production to assess if the restored habitat is positively contributing to recruitment (i.e., the attraction-production debate; see Lindberg, 1997), which could be achieved with methods such as egg mat surveys (Schwinghamer et al., 2019; Fischer et al., 2018). Finally, the spatial resolution of the acoustic telemetry array could not disentangle which restoration techniques or habitat features were directly being used by fish in the restored system. There are, however, finer-scale positioning array designs (e.g., Vemco Positioning Systems, VPS) that can be deployed to provide sub-meter positioning of fishes, as well as analytical techniques (e.g., Baktoft et al., 2017), which hold great promise for determining exactly how and when tagged fishes may utilize specific habitat restoration features (e.g., submerged logs, added substrates).

5. Conclusion

Conducting ecological restoration is challenging and assessing the efficacy of such efforts can presents even greater complexities. Our study highlights the importance of ecological monitoring for assessing the efficacy of ecological restoration for freshwater fish habitat and there were several lessons learned that can be applied to future evaluations. By combining two long-term datasets throughout all phases of ecological restoration and across both treatment and control sites, this study was able to contribute a more comprehensive understanding of fish response to ecological restoration. Our study also emphasized the importance of monitoring multiple species, as we found species-specific responses to ecological restoration. Specifically, largemouth bass responded the most positively to ecological restoration, and the goal of decreased access to common carp was also achieved, although potentially at the cost of negatively impacting movements of northern pike. We also underscore the incorporating multiple decades of monitoring data proved useful as there can often be lag times associated with ecological restoration and/or the introduction of potential confounding factors (e.g., the flood experienced during our study). Results from this study add to the growing literature base of restoration ecology, providing evidence for environmental managers to mitigate the adverse impacts associated with habitat alteration and invasive species for the benefit freshwater biodiversity.

Funding sources

Funding for this project came partially from grants held by J. Midwood under the Great Lakes Action Plan, which is administered by Environment and Climate Change Canada.

CRediT authorship contribution statement

M.L. Piczak: Conceptualization, Methodology, Data curation, Formal analysis, Visualization, Validation, Writing – original draft, Writing – review & editing. **Sebastian Theis:** Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Formal analysis, Conceptualization. **Rick Portiss:** Writing – review & editing, Resources, Project administration, Methodology, Data curation, Conceptualization. **Jonathan L.W. Ruppert:** Writing – review & editing, Writing – original draft, Visualization, Project administration, Methodology, Investigation, Formal analysis. **Jonathan D. Midwood:** Writing – review & editing, Supervision, Resources, Project administration, Methodology, Investigation, Funding acquisition, Formal analysis, Data curation, Conceptualization. **Steven J. Cooke:** Writing – review & editing, Supervision, Project administration, Methodology, Investigation, Funding acquisition, Conceptualization.

Declaration of competing interest

The authors do not have any competing interests to declare.

Data availability

Data will be made available on request.

Acknowledgements

We thank the field crews of the Toronto and Region Conservation Authority, namely Brynn Coey, Matthew Fraschetti, and Brian Graham. We appreciate the enduring support from Nancy Nate and Chris Holbrook of the Great Lakes Acoustic Telemetry Observation System. Thank you for all of the support from the larger Toronto acoustic telemetry team including Nicholas Lapointe, Maxime Veilleux, Andrew Rous, Liset Cruz-Font, Susan Doka, and Mathew Wells.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.scitotenv.2024.176088.

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