Fish community structure varies by location and presence of artificial islands: a case study in Hamilton Harbour, Lake Ontario

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Abstract Artificial islands and reefs have been used in the Laurentian Great Lakes for over 40 years as a means of improving aquatic habitat; but research on their efficacy has primarily focused on their ability to increase the abundance of specific sportfish, top predators, or other keystone species. To understand the importance of islands in structuring the whole fish community, we took a holistic approach and analysed the effect of islands, location, and the interaction effect between the two in structuring fish communities in Hamilton Harbour, Lake Ontario using a 30-year electrofishing dataset. The effect of islands varied by location within the harbour, with some species showing a preference for islands in some locations while avoiding them in others. Island communities also tended to have significantly different species compositions, with higher index of biotic integrity scores and species richness, greater numbers of pollution intolerant fishes, centrarchids, and fewer generalist species. However, these results paled in comparison to the level of inter-annual variation in the fish community of the harbour, which has changed markedly over the 30-year time span. Taken together, our results highlight that while island creation can influence the fish community, the type and magnitude of effect will vary based on their distance to other suitable habitats (i.e., location within the system) and the design of the island itself. Further, the noted inter-annual variability emphasizes the importance of considering long time scales (>10–20 years) when exploring fish community responses to habitat creation. Collectively, these results will help the design of more effective management strategies for restoring fish communities.

Keywords Freshwater · Assemblage · Reef · Aquatic habitat · Restoration · Remediation

Introduction

The creation of artificial reefs and islands as a management tool to enhance fish habitat and promote recovery of fish communities has recently garnered increased attention (Lima et al. 2019; Paxton et al. 2020a, b). The addition of these features can enhance existing habitat by sheltering littoral areas, reducing erosion, and allowing for the establishment of aquatic vegetation (Randall et al. 1996; CCB et al. 2007). They can also serve to replace habitat that has been lost due to human activities (McLean et al. 2015; Becker et al. 2018). Artificial reefs, which are underwater structures comprised of rocks or other...
aggregate material, may also initiate positive feedback loops that result in continued habitat improvements through the accretion of nutrients (Layman et al. 2016). From an aquatic habitat perspective, islands are functionally similar to reefs, provided they also have submerged aggregate areas that act to increase habitat complexity. Islands may, however, have added benefits by providing shelter from wind and wave action to aquatic areas on the lee of the island (Cooper et al. 2019; Galbraith et al. 2021).

For both reefs and islands, their creation can result in more heterogeneous habitat, with a greater diversity of plant species and substrate types, which in turn can provide valuable spawning and nursery habitat and lead to greater diversity of fish species (Leslie and Timmins 1992; Smokorowski et al. 1998; McLean et al. 2015).

The creation of islands or reefs can have positive benefits on aquatic communities, leading to increases in fish abundance and diversity (Bohn sack et al. 1994; Arena et al. 2007; Paxton et al. 2017) and increases in predator density (Binkowski 1985; Paxton et al. 2020a, 2020b); however, in some instance, they can also have negative impacts on fish communities. For example, lower species richness and genetic diversity have been documented on some artificial reefs (Carr and Hixon 1997; Froehlich and Kline 2015; Marranca et al. 2015) as have different (Koeck et al. 2014) or increasingly homogeneous fish communities (Daf f orn et al. 2015), and invasive species such as round goby (Neogobius melanostoma; Creque et al. 2006).

Collectively, these results suggest that the benefits or drawbacks of island or reef creation will be project and site-specific, and while research on the effects of artificial reefs has been substantial (Paxton et al. 2020a, b), it has primarily focused on marine ecosystems, and on how artificial reefs can be used to increase fishes targeted by recreational and commercial fisheries (Marsden and Chatkowski 2001; Lee et al. 2018; Lima et al. 2019). Comparatively, little research has been performed on the effects of lake-based reefs or islands and their impact on freshwater fish communities (Lee et al. 2018; Lima et al. 2019).

Freshwater ecosystems are at-risk globally (Strayer and Dudgeon 2010) with habitat loss being one of the primary threats to ecosystem integrity and function (Dudgeon et al. 2006; Reid et al. 2019). Degradation of remaining freshwater habitat continues to impair ecosystem function and survival of freshwater species (Richter et al. 1997; Magurran 2009). For freshwater fishes, loss or degradation of habitat has been identified as one of the primary drivers of population declines and species extirpation (Arthington et al. 2016). In the Laurentian Great Lakes in particular, loss and degradation of fish habitat has similarly been identified as the main driver behind changes in fish community assemblages (Whillans 1979; Beeton et al. 1999) and populations are further threatened by aquatic invasive species and climate change (Collingsworth et al. 2017). With all these threats manifest, considerable effort has been made to restore and remEDIATE fish habitat in the Great Lakes (e.g., Har tig et al. 2020). Artificial islands and reefs have been used for over 40 years in the Great Lakes for habitat creation or enhancement and are widely regarded as a useful means of improving populations of recreational and commercially valued fishes (Creque et al. 2006; Roseman et al. 2011; Kelch 2012; McLean et al. 2015).

To date, research on artificial reefs and islands has primarily focused on their efficacy for increasing the abundance of individual species including important top predator species (e.g., lake trout [Salvelinus namaycush]; Fitzsimons 1996; smallmouth bass [Micropterus dolomieu]; Creque et al. 2006); or other keystone species (e.g., lake sturgeon [Acipenser fulvescens]; Marranca et al. 2015). Less attention has been paid to how reefs and islands may shape the overall fish community, or what role they may play in remediating degraded ecosystems.

To better understand what effect artificial reefs and islands may have on fish communities, we turned to a 30-year electrofishing dataset from Hamilton Harbour, Lake Ontario. The harbour has a long history of anthropogenic disturbance dating back more than 150 years and was listed as a Great Lakes Area of Concern (AOC) in 1985 due to its degraded fish and wildlife populations, poor water quality, and industrial legacy contaminants, among other impairments (Smokorowski et al. 1998; BARC 2019). As a result, a Remedial Action Plan (RAP) was developed to help guide remediation of the harbour back to a more natural state. A key goal of these efforts was to improve the fish community by shifting from one indicative of eutrophic environments (i.e., one dominated by gizzard shad [Dorosoma cepedianum], brown bullhead [Ameiurus nebulosus], common carp [Cyprinus carpio], and other degradation tolerant species) to a more balanced mesotrophic community with a higher
proportion of top predators and greater native species richness (BARC 2019). Impairments to the fish community are thought to be due in part to habitat loss and degradation — both anthropogenic (e.g., nearly 70% of wetlands were filled to support industrial and urban development and > 70% of coarse substrates were removed from the harbour) and through the action of invasive species (e.g., common carp disturb vegetation used by native species; Holmes and Whillans 1984; Lougheed et al. 1998; BARC 2019). In order to combat this habitat loss, numerous habitat creation and remediation efforts have been implemented, including the creation of small islands within the harbour (Smokorowski et al. 1998).

Shortly after the construction of the islands, electrofishing and habitat surveys suggested indices of fish community health had improved slightly (Smokorowski et al. 1998). Similarly, across all altered areas in the harbour (including both created islands and other remediated areas) where aquatic vegetation became established, Boston et al. (2016) found an increase in the index of biotic integrity (IBI; an integrated measure of fish community health from Minns et al. 1994) and a higher percentage of top predators (e.g., largemouth bass [Micropterus salmoides] and other Centrarchid species such as pumpkinseed [Lepomis gibbosus]). The objectives of the present study were to focus solely on island/reef creation and thus to determine: (1) if the presence of artificial islands resulted in changes in the overall structure of nearby fish communities and, if so, whether changes were consistent among the different artificial islands; (2) if changes were detected, which species drove these changes and were changes consistent through time; and (3) if changes had a positive effect on the overall condition of the fish community. For the final objective, community condition was assessed based on IBI scores as well as some of the metrics presented in Fausch et al. (1990) as indicative of how fish communities change with environmental degradation. Specifically, higher IBI scores and increases in native species, top predators, and specialists and a decrease in invasive species, generalists, and pollution tolerant species (among other metrics) were interpreted as positive changes to the fish community. Based on past studies that involved the addition of aggregate materials to create islands and artificial reefs and the potential for the islands to create new sheltered habitat, we predict that: (1) communities near islands will be distinct from nearby areas with more exposed shorelines; (2) there will be increased catch of species that are more commonly found in protected habitats (e.g., largemouth bass and pumpkinseed), rather than species that are more representative of exposed or pelagic habitats (e.g., alewife [Alosa pseudoharengus] and gizzard shad); and (3) there will be increased catch at islands of warm-water fishes that prefer rocky substrates (e.g., walleye [Sander vitreus], round goby, smallmouth bass, rock bass [Ambloplites rupestris], and logperch [Percina caprodes]; Gannon et al. 1985; Kelch et al. 1999; Creque et al. 2006). The results from this study will directly support the assessment of fish populations and habitat within the Hamilton Harbour AOC and will also contribute to the literature on the efficacy of this type of island creation as a means of improving freshwater fish community assemblages.

Methods

Study site

Hamilton Harbour (43°14′N, 79°51′W) is located in southern Ontario, Canada, at the western end of Lake Ontario. A roughly triangular-shaped body of water, it is approximately 8 km along the east–west axis and 5 km along the north–south axis with a surface area of 22 km² (see Fig. 1). The harbour has an average depth of 13 m and a maximum depth of 26 m, and it is connected to Lake Ontario via the Burlington Shipping Canal (Smokorowski et al. 1998). During the summer, oxygen is depleted in the hypolimnion leading to frequent wind-driven upwellings of anoxic waters into nearshore areas (Flood et al. 2021). Substrate in the nearshore (<7 m) is dominated by sand, with smaller pockets of silt, clay, and larger aggregate material interspersed throughout (Doolittle et al. 2010). In total, 24 islands or reefs were constructed at six sites within the harbour between 1996 and 2001. These construction projects created reefs, emergent shoals, variegated shorelines, pebble beaches, and promontories (Fig. 1).

The islands generally fall into three groups based on their shape, size, site characteristics, and location in the harbour: the Eastern, LaSalle, and Western islands (Fig. 1). The eight Eastern islands are located on the northeastern shore of the harbour, close to the
Burlington Shipping Canal and Lake Ontario. They are the largest islands (mean: 134 m$^2$, range: 36–287 m$^2$) and thus provide the most shelter from wave action, with evidence of aquatic vegetation establishing on their sheltered side shortly after completion in 1996 (Smokorowski et al. 1998). The Eastern islands were primarily built to provide protected nesting habitat for ground- and tree-nesting colonial waterbirds (Quinn et al. 1996) and, while this has been largely successful (Zanchetta et al. 2016), soft substrates (silt or clay) are now pervasive on the sheltered side of the islands, likely as a result of these colonies. These islands are the most isolated from shallow, protected wetland areas, which are primarily located at the western end of the harbour (Fig. 1); consequently, the islands are one of the few areas that provide shallow sheltered habitat on the east end. The waters on the eastern end of the harbour can experience high turbidity as Indian Creek enters the harbour just to the north of the islands. Situated along the north-central shore of the harbour, the five LaSalle islands created in 1996 are intermediate between Eastern and Western locations in terms of their size (mean: 57 m$^2$, range: 53–62 m$^2$) and the degree of shelter they provide. These islands are rockier than the Eastern islands, and the LaSalle sampling locations generally have a high density of submerged aquatic vegetation (Doolittle et al. 2010). LaSalle transects are also situated farther from harbour tributaries and as such tend to have the lowest levels of turbidity of the three locations. Finally, the 11 Western Islands created in 2001 are the smallest (mean: 34 m$^2$, range: 24–40 m$^2$) and provide the lowest level of shelter with the lowest vegetation cover among the three areas. The islands are subject to turbid water flowing in from Grindstone Creek to their north (Fig. 1). The Western islands are also situated close to the connection point between Cootes Paradise Marsh, a 320-ha shallow protected wetland, and the harbour proper.

**Sampling methodology**

Electrofishing was conducted along the shoreline of the harbour and transect locations were fixed. This
meant that following island creation, some transects were situated on the sheltered side of the islands, which allowed us to assess the nearshore habitat that has been created as a result of island construction. Sampling followed protocols outlined in Valere (1996) and Brousseau et al. (2005) using a 6.1 m long Smith-Root SR20E electrofishing boat equipped with two anodes, each with a terminal six wire umbrella array. Current was produced by a 16 HP gas motor that powered a 7.5 KW generator. DC output ranged from 170 to 340 Volts in four steps and 8–10 Amps (standardized to 8 Amps after 1992), and output pulse frequency was set at 120 pulses per second. Fish were collected along 100 m transects that were run parallel to the shoreline, roughly following the 1.5 m depth-contour. All individuals were identified to species and were counted and weighed (Smokorowski et al. 1998; Brousseau et al. 2005).

Analysis was restricted to samples collected in the evenings between June and September from 1988 to 2019 to control for diel and seasonal effects, respectively (Midwood et al. 2016; Larocque et al. 2020). Following this filtering, the final dataset contained data from 463 sampling events collected at 13 transects over 17 years (Supplemental Tables 1 and 2). Transects were placed into one of six groups based on their location in the harbour, site characteristics, and whether an artificial island was constructed adjacent to them at some point. The groups were Eastern locations with islands (transects HH10, HH11, and HH12), Eastern locations without islands (transects HH14 and HH16), Western locations with islands (transect HH34), Western locations without islands (transects HH26, HH28, and HH36), locations near LaSalle Park with islands (transect HH19), and locations near LaSalle Park without islands (transect HH18, HH22, and HH24; Fig. 1).

Statistical analyses

To account for differences in sampling effort among groups, all metrics were standardized as catch per unit effort (CPUE)—unless otherwise specified—where effort was the number of sampling transects conducted per site, per year. Relative abundances were calculated using the decostand function in the vegan package (Oksanen et al. 2020) in R (v4.0.3; R Development Core Team 2020). As a preliminary assessment of fish community structure (Obj. 1), non-metric multidimensional scaling (NMDS) ordination was used to identify differences among groups, and determine which groups differed most in their species compositions (Clarke 1993; Bakker 2008; Midwood et al. 2016). NMDS axes were plotted against year for each group to explore temporal changes in the fish community (Obj. 2; De Caceres et al. 2010).

A permutational analysis of variance (PERMANOVA) was run with location (Eastern, Western, LaSalle) and whether the location had an island at the time of sampling (True, False) as a full factorial analysis; this allowed us to test the significance of location and island creation independently at all three locations (Obj. 1). Year was included as a random effect in the model to account for repeated sampling among the different island/location combinations. The PERMANOVA was run in PRIMER (v 7.0.17; Clarke and Gorley 2015) on square root transformed community data to control for skewed species abundances with a Bray–Curtis similarity index applied to the square root transformed data as recommended by Anderson et al. (2008). Permutations were set at 9,999 with a sum of squares type II estimation, the permutation method used permutations of the residuals under a reduced model, and the fixed effects of the mixed terms were not set to sum to zero. Analyses were restricted to years where data were present for both islands and non-islands at each location (1996–2019 for East and LaSalle locations and 2006–2019 for Western locations). Significant relationships from the PERMANOVA were assessed via a test for homogeneity of multivariate dispersion (PERMDISP) in PRIMER to ensure that differences in variance within groups were not responsible for the significant differences between groups.

A multivariate ANOVA from the mvabund package (Wang et al. 2020) was used to identify individual species that contributed significantly to the differences between locations and island and non-island areas (Obj. 2). Due to the requirements of the ANOVA, analyses were restricted to years where data were present for both islands and non-islands at each location (1996–2019 for Eastern and LaSalle locations and 2006–2019 for Western locations). Analyses were performed assuming unconstrained correlation and p values were calculated using 9,999 iterations via pit trap resampling to account for correlation in testing (Warton 2011; Warton et al. 2012).
catch rather than CPUE was used in these analyses, with the number of sampling events used as the offset (Haple and Gallagher 2021). This method operates similarly to CPUE, in that it corrects for unequal sampling among sites; however, the use of offsets is prescribed by the mvabund package authors, while Primer does not have this capability — and CPUE is used instead (Wang et al. 2020). The model for the ANOVA included location, presence of an island, and the interaction effect between the two as fixed effects; the factors of the model were blocked by year to account for repeated sampling (Wang et al. 2020). To further explore the responses of species contributing to differences among groups, we summarized where species were more likely to occur for each location both with and without an island for all species showing a significant interaction effect in the ANOVA.

A second set of analyses was run, which included only nearshore species in the model as per the methods of Smokorowski et al. (1998). This was done to assess whether offshore species may have been obscuring some significant effects of the islands on the nearshore fish community and was of interest because the islands were intended to provide additional habitat for nearshore species and thus their construction may have had the largest impact on this group. Results from the nearshore and whole community analyses were similar; therefore, we present only results for the whole community.

To determine whether islands were having positive effects on the local fish community (Obj. 3), we derived a variety of fish community measures proposed by Fausch et al. (1990) as indicators of environmental health including the number of species/individuals; number of pollution intolerant individuals/species; number of native, generalist, specialist, piscivore, and offshore species/individuals; and the IBI and adjusted IBI (which does not include pelagic fishes; Minns et al. 1994). The numbers of centrarchid species/individuals were also assessed since many species in this warmwater family rely on protected nearshore habitats at some point in their life history. Statistical analyses were performed using a paired T-test comparing island sites to non-island sites within the same location (Eastern, Western, LaSalle) and year for the period that islands were present.

Rejection of the null hypothesis (α) for all tests was p ≤ 0.05, and all values reported represent the mean ± standard error (SE) where appropriate. Results are reported without a correction factor for multiple comparisons applied as recommended by Rothman (1990). Throughout the results and discussion, we use the terms positively associated or negatively associated to indicate situations when a species had a significantly higher or lower CPUE for a fixed effect (i.e., island/non-island or one of the three assessed locations). For example, if we were to say that species X was positively associated with island sites, we mean that species X has a significantly higher CPUE at island locations compared to non-island locations. Similarly, if we were to say that species Y was negatively associated with Eastern sites, this means that species Y had a significantly lower CPUE at Eastern sites compared to LaSalle and Western sites.

Results

In total, 11,342 fish were caught across all sampling events representing 40 different species, 31 of which were classified as being nearshore species (Supplemental Table 3). During the period where islands were present (1996–2019), white perch (Morone americana) had the highest rates of catch for any species across all group/year combinations. Infrequently captured species, which only appeared in the catch data in one year/location combination, included longnose gar [Lepisosteus osseus], northern pike [Esox lucius], and trout-perch [Percopsis omiscomaycus].

Several analysis explored whether the presence of an island and its location resulted in changes in the fish community (Obj. 1). Based on the NMDS output (Fig. 2), there was extensive overlap among the different groups, but there was some evidence of clustering within groups (i.e., points within the same group are closer together than would be expected based on chance; Fig. 2, K=3, stress=0.14). For the PERMANOVA, year, location, presence of an island, and the interaction effect between location and island presence were all significant (p ≤ 0.0002 in all cases), supporting the finding that there was significant clustering in the NMDS plots of island-location groups (Table 1). The significant interaction effect indicated that the species communities at island sites varied by location. Subsequent pair-wise analyses run on the location*island interaction effect revealed that Eastern island and non-island fish communities
differed significantly from LaSalle and Western island and non-island communities ($ p \leq 0.033 $ in all cases). LaSalle and Western island and non-island communities were not significantly different from one another at the $ \alpha \leq 0.05 $, but there was the indication of an effect ($ p \leq 0.0771 $ in both cases). This may have been due to the fact that the LaSalle sites were closer to the nearest Western transect (1.5 km) than the nearest Eastern transect (2.1 km). Most relevant to our objectives, Eastern, Western, and LaSalle islands were significantly different from Eastern, Western, and LaSalle non-islands ($ p = 0.003 $, $ p = 0.002 $, and $ p = 0.043 $, respectively; see Table 1 for a full breakdown of the relationships). Differences in dispersion as measured through PERMDISP were not significant for presence of an island ($ F = 2.63 $; $ p = 0.15 $) or location ($ F = 0.05 $; $ p = 0.96 $) indicating that differences in within group variance were not responsible for the significant results from the PERMANOVA that we observed. Despite a significant effect of all of the fixed effects in the model, they explained considerably less variance within the data than the random effect of year (location: 5.3%; presence of an island: 3.1%; interaction effect between location and presence of an island: 4.2% compared to 49.7% for year; Table 1), indicating that inter-annual variation in factors not considered by this model played a larger role in shaping the fish communities at the various sites in the harbour.

While the ANOVA produced similar results to the PERMANOVA and found a significant effect of location, presence of an island, and an interaction effect between the two ($ p \leq 0.011 $ in all cases), it also allowed us to determine which species were driving the apparent changes in the communities (Obj. 2). Post hoc univariate analyses examining which species contributed to these significant effects were performed using a multivariate ANOVA. In total, 13 species were found to have a significant relationship with location (i.e., higher or lower rates of capture per unit effort at a specific location), 11 species were found to have a significant island/no island relationship, and 18 were found to have a significant relationship with the interaction effect between the island and location factors (Supplemental Table 4). Yellow perch had the largest location effect as measured through the Wald test statistic ($ W = 20.2 $, $ p \text{ value} \leq 0.001 $) and were predominantly found at LaSalle sites. Rock bass had the largest island/no island effect ($ W = 20.7 $, $ p \text{ value} \leq 0.001 $) and were primarily found at island sites. Finally, brown bullhead had the largest location/interaction effect ($ W = 992.8 $, $ p \text{ value} < 0.001 $) with positive associations at Eastern island sites, but negative associations at LaSalle and

| Main test | df | SS  | $ \eta^2 $ | Pseudo-$ F $ | $ P(\text{perm}) $ |
|-----------|----|-----|----------|--------------|------------------|
| Island    | 1  | 2,872 | 3.06%    | 4.071        | 0.0002           |
| Location  | 2  | 5,000 | 5.34%    | 3.5434       | 0.0001           |
| Island*Location | 2  | 3,891 | 4.15%    | 2.7575       | 0.0002           |
| Year      | 11 | 46,604 | 49.72%   | 6.0048       | 0.0001           |
| Residuals | 47 | 33,161 | 35.38%   |              |                  |
| Total     | 63 |      |          | 93,717       |                  |

| Pair-wise comparison | df | $ t $ | $ P(\text{perm}) $ |
|----------------------|----|------|------------------|
| No island            |    |      |                  |
| East, LaSalle        | 11 | 1.7677 | 0.0095          |
| East, West           | 7  | 2.2419 | 0.0033          |
| LaSalle, West        | 7  | 1.4501 | 0.0771          |
| Island               |    |      |                  |
| East, LaSalle        | 11 | 1.4821 | 0.0332          |
| East, West           | 7  | 1.633 | 0.0231          |
| LaSalle, West        | 7  | 1.4808 | 0.0742          |
| Island vs no island  |    |      |                  |
| East                 | 11 | 2.1328 | 0.0029          |
| LaSalle              | 11 | 1.4953 | 0.0427          |
| West                 | 7  | 1.7262 | 0.0239          |
Western island sites; see Supplemental Table 4 for a full list of the univariate tests.

In general, species that showed a significant interaction effect were either positively associated with Eastern island sites and negatively associated with Western island sites or positively associated with Western island sites and negatively associated with Eastern island sites. This was true for 72% of the species showing a significant interaction effect. The remaining 28% of species showed either a positive or negative association with all island sites. Species at LaSalle sites followed the same patterns as at Eastern or Western sites (e.g., a species never showed a positive association with LaSalle islands and a negative association at both Eastern and Western islands; Table 2).

To evaluate species-specific predictions, we conducted a focused review of each species showing either a significant response to the presence of an island or an interaction effect between location and the presence of an island. For warmwater fishes that prefer more sheltered habitat, the majority showed positive associations (i.e., higher CPUE) with Eastern and LaSalle islands, but negative associations with Western islands (e.g., white perch, largemouth bass, and pumpkinseed; Table 2). More pelagic fishes like alewife and gizzard shad were positively associated with the Western islands (LaSalle as well for gizzard shad), but emerald shiner showed a negative relationship with all island locations. Finally, commonly encountered lithophilic species like round goby and rock bass were positively associated with all island locations, logperch were positively associated with islands in general (interaction term was not significant), and walleye were negatively associated with all island locations (Table 2).

To explore temporal changes in fish community structure (Obj. 2), a plot of the NMDS axes described previously over time revealed that the different islands-location groupings appeared to be experiencing similar community changes from 1988 to 2019 (Fig. 2). The most dramatic change was arguably in NMDS 1, which showed a steady increase over time across all sampling locations. NMDS 1 had statistically significant positive loadings from gizzard shad, brook silverside [Labidesthes sicculus], rock bass, walleye, goldfish [Carassius auratus], and bluegill [Lepomis macrochirus]; and negative loadings from alewife, white perch, pumpkinseed, logperch, emerald shiner (Notropis atherinoides), yellow perch (Perca flavescens), brown bullhead, largemouth bass, common carp, spottail shiner [Notropis hudsonius], white sucker [Catostomus commersonii], and round goby indicating that these species were increasing and decreasing, respectively, over time (Supplemental Table 5).

Finally, overall fish community improvements were assessed using derived metrics indicative of ecosystem and community condition (Obj. 3). Mean IBI scores were higher, and pollution intolerant fish, centrarchids, and offshore fishes were significantly more numerous at island sites relative to non-island sites, while the total number of generalists captured,
the number of generalist species, and total number of species were significantly higher at non-island sites relative to island sites. Other indicators of community health (e.g., number of individuals, number of top predators, and number of specialists) were not significantly different between the two groups (Table 3).

**Discussion**

With increasing stress on aquatic ecosystems (Strayer and Dudgeon 2010), habitat protection, remediation, and creation are critical elements in the management of aquatic resources. Interventions, including the reclamation of lost and degraded habitat, are often viewed as a means of restoring biotic communities to their former state. Thus, assessing the efficacy of habitat interventions by documenting the response of targeted biotic communities, such as fish, is essential. The artificial islands in the Hamilton Harbour AOC were created to provide habitat that could support diverse wildlife and fish communities. For colonial waterbirds, the success of island creation has been documented (Zanchetta et al. 2016). From a fisheries perspective, earlier works undertaken shortly after the creation of some of the islands showed short-term improvements in some fish community metrics (Smokorowski et al. 1998; Boston et al. 2016). In the present study, however, we found more mixed results with spatially distinct fish communities among locations, strong temporal effects throughout the system, and distinct island-focused communities at the Eastern and Western locations, and to some extent at LaSalle. These more mixed results reinforce the importance of monitoring habitat creation or enhancement projects for extended periods of time to ensure that apparent short-term gains are persistent. This allows the area to experience different climatic conditions (Kondolf and Micheli 1995; Jeppesen et al. 2007) and allows multiple generations of fish to use the area, making effects more apparent (Niemi et al. 1990; Johnson et al. 2005). Our results also serve to highlight that habitat creation efforts are often not as straightforward as we may think — the creation of suitable habitat will not guarantee the reestablishment of previous communities, and the communities that do arise will often be shaped by their place in the broader landscape.

**Island- and location-specific differences**

Fish communities varied by presence of an island and their location within the harbour. Eastern sites had significantly different species assemblages than

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**Table 3** Paired T-test results comparing measures of habitat quality between islands and non-islands for each year where samples were available for both island and non-island sites at each location (East/ LaSalle: 1996–2019; West: 2006–2019). IBI is the index of biotic integrity score, a measure of fish community health from Minns et al. (1994). IBI* is the adjusted index of biotic integrity score which only considers nearshore species. For details on which species is in which group, see Supplemental Table 3. Significant p values appear in bold.

| Comparison          | Island mean ± SE | Non-island mean ± SE | t-stat | p value |
|---------------------|------------------|-----------------------|--------|---------|
| IBI                 | 43.86 ± (1.80)   | 38.47 ± (1.53)        | 2.89   | 0.01    |
| IBI*               | 22.89 ± (1.86)   | 20.78 ± (1.54)        | 1.43   | 0.16    |
| # Species           | 11.12 ± (0.54)   | 12.84 ± (0.72)        | −2.00  | 0.05    |
| # Individuals       | 23.58 ± (2.43)   | 20.65 ± (1.68)        | 1.42   | 0.16    |
| # Native species    | 8.06 ± (0.43)    | 9.12 ± (0.56)         | −1.57  | 0.13    |
| # Native individuals| 16.62 ± (1.82)   | 13.50 ± (1.27)        | 1.70   | 0.10    |
| # Intolerant species| 1.50 ± (0.12)    | 1.70 ± (0.15)         | −0.80  | 0.43    |
| # Intolerant individ| 1.87 ± (0.28)    | 1.16 ± (0.17)         | 2.77   | <0.01   |
| # Centrarchid species| 2.34 ± (0.21)    | 2.22 ± (0.25)         | 0.43   | 0.67    |
| # Centrarchid individuals| 4.13 ± (0.91) | 1.89 ± (0.45)         | 2.76   | <0.01   |
| # Piscivore species  | 1.19 ± (0.14)    | 1.47 ± (0.19)         | −1.34  | 0.18    |
| # Piscivore individ| 1.54 ± (0.38)    | 1.06 ± (0.26)         | 1.48   | 0.15    |
| # Generalist species| 2.44 ± (0.18)    | 3.09 ± (0.19)         | −2.64  | 0.01    |
| # Generalist individ| 2.84 ± (0.41)    | 3.80 ± (0.52)         | −2.06  | <0.05   |
| # Specialist species| 7.53 ± (0.43)    | 8.28 ± (0.53)         | −1.32  | 0.19    |
| # Specialist individ| 19.20 ± (2.09)   | 15.79 ± (1.43)        | 1.94   | 0.06    |
| # Offshore species  | 3.38 ± (0.20)    | 3.75 ± (0.22)         | −1.79  | 0.08    |
| # Offshore individ| 12.37 ± (1.39)   | 9.80 ± (0.77)         | 2.36   | 0.03    |
Fig. 2 Top Non-metric multidimensional scaling (NMDS) of the six different island groups (EI, Eastern islands; ENI, Eastern non-islands; LSI, LaSalle islands; LSNI, LaSalle non-islands; WI, Western islands; WNI, Western non-islands). Each point represents the sample score based on catch per unit effort for a single group in a given year after correcting for relative abundance. Ellipses are the 95% confidence intervals for each group. Despite overlapping ellipses, island groups are closer together than would be expected based on chance. See Supplemental Table 5 for details on species loadings along the axes. Bottom NMDS 1 plotted over time for each of the six island groups. The temporal pattern indicates that there have been system-wide changes in the fish community that would make detecting an effect of island creation challenging without suitable control sites. See Supplemental Fig. 1 for NMDS 2 and NMDS 3 versus time, respectively.
LaSalle or Western sites, indicating the presence of strong location-based differences within the harbour. These differences have been well documented in past studies focused on fish communities in Hamilton Harbour with generally higher density and biomass of fish in the western harbour relative to the north and east (Boston et al. 2016; Midwood et al. 2019). Of all the terms in the model islands had the smallest effect, explaining only 3.1% of the variation. This indicates that in Hamilton Harbour, island construction only plays a small role in shaping fish communities. Instead, specific local habitat conditions and the availability of alternative habitat are more likely to drive differences in fish communities between island and nearby non-island locations.

We detected significant differences in fish communities between island and non-island sites for all three locations, though LaSalle island and non-island fish communities were not significantly different from Western island and non-island communities, respectively. This could be partially related to where the LaSalle sites are located in that they fall between Eastern and Western sampling locations and their slightly closer proximity to Western sites may influence their community structure (see Fig. 1). Fish community structure varies with distance from an artificial reef (Rosemond et al. 2018) and thus proximately situated sampling transects may be more similar. A more in-depth assessment of movement and home ranges for warmwater species would help to determine appropriate distances between sampling transects, with estimates for some warmwater fishes (e.g., 2.1 hectares for largemouth bass and 1.1 hectares for pumpkinseed [Fish and Savitz 1983]) suggesting that some of the non-island LaSalle and Western transects may be too close to be fully distinct. This may have contributed to the observed similarity in the fish communities at island and non-island site at the LaSalle and Western locations. If these locations were more spatially distinct, it is possible that we may have observed significantly distinct fish communities.

Species-specific differences

The effects of artificial reefs or islands on fish communities have been shown to vary by location; water temperature (Creque et al. 2006); reef structure (Granneman and Steele 2015); and distance to other reefs (Lowry et al. 2014). In the present study, we similarly identified species that showed a positive association with islands (i.e., had a higher CPUE) in some locations while appearing to avoid them in other locations (e.g., largemouth bass). The interaction effect between location and presence of an island explained a similar level of variation in the model as the effect of the island, or the location effect by itself. Such location-island interactions can generally be divided into four groups of species: those showing a positive association with Eastern islands while avoiding Western islands (white perch, brown bullhead, largemouth bass, yellow perch), those showing a positive association with Western islands while avoiding Eastern islands (gizzard shad, alewife), those showing a negative association with islands in general (emerald shiner), and those that were always positively associated with islands (rock bass, round goby). Species-specific associations with LaSalle islands matched patterns seen at Eastern or Western island sites, but were more muted. These groupings align well with our predictions, such that some island locations do show increased catch of nearshore and lithophilic species and decreased catch of pelagic fishes; however, environmental conditions at locations where these predictions were not met may partially explain these discrepancies.

Opposing species associations for the Eastern and Western islands are intriguing and may be related to both the design of the islands and their location within the harbour. As noted previously, the Eastern islands are more isolated from shallow wetland areas compared to the Western location, which is adjacent to Cootes Paradise Marsh and marshes in lower Grindstone Creek. The Eastern islands may therefore provide the only sheltered, shallow habitat on the east end of the harbour resulting in an “island effect” that serves to attract nearshore-oriented species like largemouth bass and pumpkinseed (MacArthur and Wilson 1967; Boada et al. 2018). Additionally, the larger footprint of the Eastern islands results in more sheltered habitat in their lee and the smaller Western islands may simply not afford sufficient protection from wind and wave action or the frequent upwellings of hypolimnetic waters that occur in the harbour during the summer (Flood et al. 2021). As a result, the lack of a positive association of nearshore-oriented species at the Western islands is likely a combination of the limited area of suitable habitat they create and the proximity of equally suitable habitat. Finally,
the presence of nesting bird colonies at the Eastern islands may also influence local fish communities since prey fishes tend to avoid areas of high predation risk (Fauchald 2009). While species such as the double-crested cormorant (*Phalacrocorax auritus*) tend to forage within a few kilometres of their nesting area, the entire harbour falls well within their observed foraging range (Coleman et al. 2005). Prey fishes throughout the harbour are thus likely affected by recent increases avian predators (Zanchetta et al. 2016), but further study of the spatial distribution of avian foraging efforts is required to determine their location-specific influence on fish community composition.

Pelagic species were either negatively associated with islands in general (emerald shiner) or showed the opposite pattern as more nearshore fishes, with positive associations with Western islands and negative association with the Eastern islands (e.g., gizzard shad and alewife). These species generally prefer areas of open water and typically only use the nearshore environment for spawning and nursery habitat (Page and Burr 2011), so a lack of association with islands for some is not surprising. For those associated with Western islands, conditions at these small, rocky, exposed islands are more closely aligned with the offshore environment than more sheltered areas, like the Eastern islands. Past surveys of pelagic fishes in the harbour have found the highest density and biomass of these fishes in the western part of the harbour as well as a concentration of individuals above the thermocline due to hypolimnetic anoxia (Midwood et al. 2019). Increased capture in the nearshore of the west end may therefore be related to relatively higher abundance of these fishes in this region as well as their reliance on comparatively oxygen-rich surface waters. Upwellings of anoxic hypolimnetic waters (as outlined in Flood et al. 2021) may further act to push these fishes into even shallower waters where they can be captured using electrofishing. Finally, alewife in particular have been shown to form loose associations with artificial reefs that lie in the path of zooplankton-rich currents (Creque et al. 2006). While no information on the density of zooplankton at the sites sampled here was available, a spatially limited comparison of zooplankton biomass within the harbour between nearshore and offshore areas suggested high predation by planktivores in the nearshore (Currie et al. 2018). Assessing lower trophic production and foraging opportunities throughout the harbour would aid in our understanding of the drivers behind observed fish distributions, particularly for pelagic fishes that move readily to track their prey.

Finally, we predicted that lithophilic fishes would be more prevalent at island sites given the presence of aggregate materials along their margins. This was true for some species (e.g., round goby, logperch, and rock bass); however, there were no island associations for other species like smallmouth bass and walleye that have been linked to islands in previous studies (Kelch et al. 1999; Creque et al. 2006). Smallmouth bass had overall low CPUE, which may have masked any island effect, and while CPUE was slightly higher for walleye, they were actually negatively associated with islands, particularly at the Eastern and LaSalle locations. Walleye in Hamilton Harbour are somewhat unique because they have been stocked into the system as fingerlings since 2012 (OMNRF 2020) and, as such, adults did not show up in the catch data in large numbers until after 2015. In previous studies, they have been positively associated with artificial reefs (Kelch et al. 1999); however, summer sampling in the current study does not align with their primary period of use of these types of habitats (i.e., during spawning; Raabe and Bozek 2012) and as such they may make use of islands more in the spring (OMNRF 2020). Further exploration of electrofishing data from other seasons will help to determine if the apparent differences in island use by species such as walleye in the present study are a function of the timing of sampling and/or the type of habitat that has been created as part of island creation in Hamilton Harbour.

Temporal effects

There were strong temporal trends in the fish community data with year explaining a considerable amount of the variation in our model. The trends also appeared to show consistent shifts in community structure across years with changes in NMDS axes 1, 2, and 3 all appearing to be consistent among islands/locations, while also showing consistent directional changes over time in NMDS 1, particularly since 2010. This suggests that there are likely large harbour-wide changes in community structure occurring in Hamilton Harbour that are not being fully captured within our models. There are multiple potential factors that could be acting independently or
in concert including ongoing walleye stocking efforts that have yielded strong year classes of this important top predator (biannually starting in 2012; OMNRF 2020); establishment of novel invasive species (e.g., round goby were predicted to have arrived in 1999; Balshine et al. 2005 and rudd [Scardinius erythrophthalmus] were first detected in 2013; C. Boston, unpublished data); declines in invasive common carp since management activities started excluding them from their spawning habitat in Cootes Paradise Marsh in 1996 (Boston et al. 2016); increases in populations of invasive species already present in the system (e.g., increasing catch of goldfish starting in 2006; Boston et al. 2016); increasing populations of avian predators like double-crested cormorants (Zanchetta et al. 2016); and inter-annual variation in nutrients, water quality, and primary productivity (Hiriart-Baer et al., 2016; Munawar et al. 2017) among a myriad of other factors. Many of these factors are also manifest in Lake Ontario in general and have been linked to declining fisheries productivity as nutrients are shifted from the pelagic to the benthic system (Mills et al. 2003; McKenna et al. 2017). However, without more information on the potential drivers behind the observed temporal changes, it is unclear whether the observed shift in the fish community over the past 10 years in Hamilton Harbour is related to changes in local conditions or driven by regional lake-wide factors. For the present work, the inclusion of control transects helped incorporate some of these more systemic changes into our analysis by providing a dynamic baseline with which to compare the communities at the island sites. Regardless, conclusions on the magnitude of changes in the fish community solely from island creation are somewhat confounded by the underlying change in the fish community. Future research should work to uncover the factors that are driving these system-wide trends as it is possible that the addition of these other explanatory variables may serve to increase or diminish the amount of variation in community structure explained by location or island creation. There are a small number of similar long-term datasets available in Lake Ontario (e.g., Bay of Quinte; Boston et al. 2016, and Toronto Harbour; Midwood et al. 2022) and exploring trends in fish community structure among all these datasets may help partition the amount of variation that is specific to Hamilton Harbour and island creation compared to more systemic lake-wide changes. Additionally, by improving our understanding of the source, magnitude, and direction of these unknown effects, we may uncover new targets for effective community management and improvement.

Many studies assessing habitat creation or remediation efficacy have been limited by datasets with shorter temporal scales, often only spanning 2 to 4 years (Creque et al. 2006; Folpp et al. 2011; Lowry et al. 2014). Given that diversity and abundance are often expected to increase following artificial reef construction but are likely to take a year or more to stabilize, these limitations may be pronounced (Folpp et al. 2011; Becker et al. 2018). These short temporal scales also mean that year-to-year variation or underlying systemic shifts, as observed in the present study, have the potential to obscure the effects of island creation itself. In addition to short-term natural variation, fish community changes from habitat creation can take multiple generations of target fishes to fully manifest (Niemi et al. 1990) and such species-specific lag-times in response to a change can be challenging to predict (Nilsson et al. 2017). This further emphasizes the need for long-term monitoring both before and after the implementation of a restoration project, with the duration of such monitoring linked to the life history of the target species.

Fish community improvement

From a fisheries management perspective, islands in Hamilton Harbour had a more desirable species community, in that pollution intolerant fishes and centrarchids were more numerous at island sites compared to non-island sites, and island sites also had higher IBI scores. Comparatively, non-island sites had communities more indicative of degraded ecosystems with a higher number of generalist species and individuals (Fausch et al. 1990; Smokorowski et al. 1998). Interestingly, island sites had a greater number of individuals associated with the offshore environment, which was unexpected since the presence of the islands should provide some shelter from wind and wave action; however, the adjusted IBI score, which only includes nearshore species, was not significantly different between island and non-island sites. This could indicate that offshore species may congregate around islands (as noted for alewife and gizzard shad for some locations), and their abundance drives some of the community-level differences between island and non-island sites.
Despite some indication of improved fish communities at island sites, the effectiveness of islands as a means of habitat remediation and for changing species composition remains to be seen. As has been discussed in previous studies, without specific targeted analyses of the biological processes taking place at the islands, it is impossible to disentangle the question of whether the islands are increasing species-specific abundance, or merely attracting fish from elsewhere in the system (Simon et al. 2011). Future work examining year-to-year recruitment of individual species and age-class associations with islands may prove useful in disentangling this issue. Additionally, while the overall effect of islands on the structure of the fish community appeared small — as the presence of an island only accounted for 3.1% of the variation in the model — consideration must be given to the fact that the islands represented a comparatively small habitat modification (0.0017 km²) in a large harbour (22 km²), accounting for less than 0.01% of the total area. Such disparity in area highlights the extent to which harbour-based processes are likely a far greater influence on fish communities than small-scale habitat changes at the islands.

Conclusion

Artificial islands or reef creation has been used as a means to restore aquatic habitat in a range of ecosystems with mixed evidence of efficacy for improving fish community condition. Reef creation is the more common approach in a fisheries context, but island creation has the potential to benefit the fish community in addition to avian or terrestrial species. While limited by only three islands, our study suggests that the effect of island creation on the fish community is likely to vary based on their location within the larger ecosystem, distance to other suitable habitat (and thus proximate species pool), and the size or footprint of the island itself. This was particularly evident at the Eastern locations where islands appeared to provide important and unique sheltered habitat, which resulted in many species showing a positive association with the island sites. Additionally, responses to island creation were species-specific (as well as location specific), with the individual species’ unique biology and habitat requirements dictating their response. Finally, these works draw attention to the importance of considering long time scales (>10–20 years) when exploring the effect islands or artificial reefs may have on the fish community. Considerable inter-annual variability in fish community assemblages was evident within Hamilton Harbour and thus short-term assessments (<5 years) of the response of a fish community could be masked by natural variation or unrelated environmental or systemic factors. Collectively, by improving our understanding of species-specific and community relationships to artificial islands, we can design more effective management strategies that work towards improving fish communities.

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Data availability The datasets generated during and/or analysed during the current study are available from the corresponding author on reasonable request.

Declarations

Ethics approval All fish community monitoring work conducted by Fisheries and Oceans Canada followed Canadian Council for Animal Care guidelines including internally developed Standard Operating Procedures for electrofishing and fish handling (GWACC-111).

Conflict of interest The authors declare no competing interests.

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