Effect of a Fish Ladder on Fish Community Similarity Inside and Outside a Diked Great Lakes Marsh

by

Geoffrey B. Steinhart[1], Sierra Bode[2,3], Jennifer Thieme[4], and Christopher J. Winslow[2]

10/29/2015

[1] Author to whom correspondence should be addressed (geoffrey.steinhart@dnr.state.oh.us). Ohio Department of Natural Resources, Sandusky Fisheries Research Station, 305 E. Shoreline Dr., Sandusky, OH 44870

[2] Ohio Sea Grant College Program and F. T. Stone Laboratory, The Ohio State University, 1314 Kinnear Road, Columbus, OH 43212

[3] Department of Biology and Geology, Baldwin Wallace University, 275 Eastland Rd., Berea, OH 44017

[4] The Nature Conservancy, Kitty Todd Nature Preserve, 10420 Old State Line Rd., Swanton, OH 43558

Abstract

Great Lakes wetlands are important habitat for many fishes. Over 40% of Great Lakes fish species are wetland-dependent, while more than 70% benefit from wetlands periodically. As the critical role these wetlands play in the life cycles of Great Lakes fish is better understood, the rate of restoration and rehabilitation of Great Lakes coastal wetlands has increased dramatically. This necessitates a balance among varied management objectives, including fish passage. The presence of water control structures commonly found on diked and managed wetlands can limit fish passage. We sampled three diked Lake Erie coastal wetlands to assess how different structures affected fish passage through each structure type. Water control structures included a pool-and-weir fish ladder, a swing gate, and an open culvert. Fyke nets were set five to seven times at each site between April and October 2015. One net was set outside the wetland and the other was set inside the wetland, with the net wings enclosing the opening of the structure. Catches were compared for catch rate, species richness, three measures of community similarity and the Wetland Fish Index (WFI). Fish communities on either side of the culvert-drained wetland had the highest community similarity, suggesting that the open culvert provided the greatest opportunity for fish passage. The fish ladder and swing gate had low community similarities and low catch rates inside the wetlands signifying that few fish were moving into or out of the wetlands. Of the three wetlands, the Culvert wetland had a WFI most similar to wetlands connected to Lake Erie hydrology as examined in previous studies. Overall, the culvert provided the most opportunity for fish movement into the wetland; however, unlike the other structures, it did not provide a means for precisely managing wetland water levels.

Introduction

Wetlands are some of the most productive and diverse habitats providing habitat for plant and animals and serving as important links between terrestrial and aquatic ecosystems. Even though wetlands now make up only 3.5% of the land area in the United States, almost half of the listed endangered species depend on or use wetland habitat during their life (Mitsch and Gosselink 1993). Warm, shallow, and nutrient rich wetlands around the Great Lakes are believed to provide more food and shelter for wildlife than any other habitat around basin (Herdendorf 1987). In fact, 42% of Great Lakes fishes examined were considered wetlands species and 73% periodically used wetland habitats (Jude and Pappas 1992). This heavy reliance on wetland habitats suggests that poorly functioning wetlands have the potential to impact a majority of fish species in the region and emphasizes their widespread importance and potential risks of wetland loss. In fact, wetland fishes represent approximately half the biomass, 60% of the economic value of commercial fishes, and 80% of the recreational harvest in the Great Lakes (Trebitz and Hoffman 2015). Unfortunately, approximately 90% of the natural wetlands around Lake Erie have been degraded or destroyed due to human activities (Herdendorf 1987, Jude and Pappas 1992). Furthermore, when wetlands have poor connectivity to the broader ecosystem, both fish species richness and abundance can decline (Bouvier et al. 2009).

Many natural wetlands along western Lake Erie have been replaced by intensively managed and diked wetlands that impair some natural functions. Managed wetlands may meet some management goals, but trade-offs with other factors can lead to degraded water quality (Mitsch and Wang 2000), lower productivity (Cronk and Mitsch 1994), less desirable plant (Herrick and Wolf 2005, Mitsch et al. 2014) and have ambiguous effects on avian communities (Monfils et al.

2014). Because diked wetlands are often managed for vegetation and wildlife, wetland fish diversity often suffers (Herdendorf 1987, Bouvier et al. 2009). Wetland fishes include permanent residents (e.g., Longnose Gar and bullhead species; all scientific names are provided in Table 1), as well as migratory species using the wetlands seasonally for spawning or as nurseries (e.g., Northern Pike and Gizzard Shad; Jude and Pappas 1991). Typical diked wetland management actions include a spring draw-down to expose mudflats and promote vegetative growth, followed by refilling in fall (Wells et al. 2002). Diked wetlands often use pumps to fill and drain, which may allow very small fish to enter but provide no path for juvenile and adult fish to enter or leave the wetland. If proper connectivity is not maintained, fishes using wetlands seasonally or permanently may not be able to colonize newly created wetlands or move among existing wetlands. Therefore, fish passage in and out of wetlands is an ongoing concern for agencies seeking a balance between water, vegetation, waterfowl, and fish management objectives.

A notable exception where fish passage was a targeted objective was the management of Metzger Marsh on Ohio's shore of Lake Erie (Wells et al. 2002). The Metzger Marsh restoration included construction of a 2,348-m long dike to mimic a historical, but eroded, barrier beach. Five water control screw gates allowed connectivity to nearshore Lake Erie. To facilitate fish passage and assess the effects on the marsh and the fish community, the gates were required to remain open for four years after construction. Two gates were equipped with traps to capture immigrating and emigrating fishes. The remaining gates were equipped with steel grates to limit the migration of non-indigenous Common Carp while allowing desirable native fishes (French et al. 1999). With the open connection to the lake, over 100,000 fishes from 45 species and 16 families were captured moving in or out of the wetland between 1999-2002 (Wells et al. 2002). However, actions like this are rare for Great Lakes diked wetlands in part because leaving

connections open year-round often prevents water level management desired to meet some wetland objectives.

There are, however, other alternatives to promote fish passage while also maintaining some control of wetland water levels. Pool-and-weir fish ladders have been used to pass migratory species in many river systems, and have recently been applied to Great Lakes coastal wetlands. However, their utility for species common to these low-gradient systems has not been well studied. We aimed to determine the effectiveness of different structures for fish passage in three wetlands connected to Lake Erie hydrology. Our sites included three diked wetlands: one with a pool-and-weir fish ladder designed to promote fish movement and allow water level control; one with an open culvert allowing continuous passage but no control of water flow; and one with swing gates on manually adjustable sluice gates designed to control water levels. Sluice gates are typically metal gates that can be manually raised or lowered to allow for water exchange on a seasonal basis. Swing, or flap, gates open in one direction; when water pressure on the interior side of the gate reaches sufficient levels, the gate will swing open to allow water to flow out. The gate cannot swing open in the opposite direction even when under pressure from exterior water, thus preventing water from flowing back into the wetland. Because fish ladders are designed to aid fish passage by providing constant upstream access with resting pools, we hypothesized that fish movement (measured as similarity of the fish communities inside and outside the wetlands) would be greatest at the wetlands with the fish ladder and open culvert and lowest at the wetland with swing gates.

Methods

Study sites

We sampled three wetland sites each with a different structure that influenced water level and fish passage (Figure 1). The fish ladder site, known locally as the Blausey Tract, was a 40.4 ha wetland created in 2013 by raising levees around a former agricultural field, installing a pump within the levee adjacent to an agricultural ditch, and constructing a two-bay box culvert (each bay was 1.57-m wide) between the impoundment and the Toussaint River (a tributary to Lake Erie). One bay of the box culvert contained stop logs and the other a pool and weir fish ladder with three steps and a carp-exclusion grate on the bottommost weir. A fish ladder was selected to allow continuous fish movement between the wetland and the Toussaint River while maintaining control of wetland water level. The open culvert site was a 21-ha impoundment at the Great Egret Marsh Preserve, acquired by the Nature Conservancy in 2013. This diked impoundment was connected to Wet Harbor, a protected bay of Lake Erie, via a 5-m long, 91-cm diameter metal corrugated culvert. Water was free to flow in either direction through the culvert and flow volume and direction were dependent on lake and wetland levels (a function of rainfall wind direction and velocity). Finally, we sampled at Winous Point, which had two sluice gates each with a top-hinged swing gate to separate Sandusky Bay from a canal draining a series of wetlands and agricultural lands. The swing gates could open when water pressure in the canal was sufficient to force open the flaps (e.g., during spring run-off). The sluice gates could be manually opened to allow water movement in either direction. Because the swing and sluice gates were generally closed during our sampling (see results), we considered the swing gate site as a control because the closed gates should have prevented most fish passage. Because the emphasis in this study was on the structures separating the wetlands from Lake Erie, the sites

will be referred to by the structure type (fish ladder, culvert and swing gate) instead of their geographic names.

Sampling protocol

Sampling generally occurred bi-weekly during three seasons in 2015 (Table 2): spring (27 April to 28 May, three samples), summer (11 July to 20 July, two samples) and fall (21 September to 7 October, two samples). The summer sampling protocol was modified due to strong winds and heavy rains that dramatically increased water levels and created a safety concern. As a result, the first summer sampling date was abandoned and only the fish ladder and culvert sites were sampled on consecutive weeks (Table 2). In the end, the fish ladder site had seven complete samples, the culvert site had six complete samples, and the swing gate site had five complete samples.

On each sampling date, we collected surface water quality data with a YSI Pro Plus Multiparameter Instrument (YSI Inc., Yellow Springs, OH) including: water temperature, dissolved oxygen, specific conductivity, and pH. These data were collected to detect any unusual conditions, but values were within reasonable ranges (mean temperature = 20.6 °C (range: 9 - 28.4); mean dissolved oxygen = 7.0 mg/l (3.2 - 13.3); mean conductivity = 497 µS (277 - 678); mean pH = 7.9 (7.4 - 8.5)). Turbidity was measured with a Hach 2100Q Portable Turbidmeter (Hach Company, Loveland, CO; mean turbidity = 37.1 ntu (4.3 - 89.7)). Because no anomalous data were observed, the full water quality data are not presented. We also measured water velocity at approximately 0.3 m depth on both sides of the control structure with a FP111 Global Water Flow Probe (Global Water Instrumentation, College Station, TX).

At each site, two fyke nets were set: one to collect fish from the bay or river to which the wetland was connected (external set) and one to capture fish moving through the structure into the wetland (internal set, Figure 2). The external set was perpendicular to shore with the lead stretching to shore and the wings set at approximately a 45° angle from the lead. The internal set was inside the wetland, in front of the connecting structure with the wings completely enclosing the potential passage. At the fish ladder and swing gate sites, the nets within the wetland were completely submerged. As a result, fish could potentially swim over the wings. Two styles of fyke nets were used to collect fish. Small fyke nets (0.91 x 0.91 m frames, 0.76 m diameter hoops, 5 mm bar mesh, 15.2 m lead and 6.1 m wings) were used on all but three sampling dates. Large fyke nets (0.91 x 1.82 m frames, 0.76 m diameter hoops, 12.7 mm bar mesh, 21.3 m lead and 6.1 m wings) were used on three occasions: outside the fish ladder site on 12 May and outside the swing gate site on 29 April and 28 May. Nets were left overnight for 14 to 27 hr, but we considered the sample measurement to be a net-night, irrespective of soaking time.

All fish caught were identified to species. Total length (nearest mm) and wet weights (nearest g) were taken for the first 30 randomly-selected fish of each species. If more than 30 fish of a single species were caught, the remainder was weighed in bulk and the number of fish estimated based on mean weight of the 30 randomly-selected individuals. Fish were returned to the habitat from which they were caught.

Data analysis

To assess fish passage through the structures, we compared numbers and types of fish captured and fish community similarity inside and outside the wetlands. We used all data and ignored net type (large or small) because there was no difference in catches between the small

and large fyke nets. Catch rates of the two types of nets were not different at either site where both net types were used (fish ladder: $t = 2.01$, $df = 5$, $p = 0.95$; swing gate: $t = 2.35$, $df = 3$, $p = 0.90$), excluding 1,195 Emerald Shiners caught in a small fyke net at the fish ladder site on 28 April 2015, the only site and date where we caught more than 70 Emerald Shiners. Furthermore, the similarity indices when one large and one small fyke net were set were well within the 95% confidence intervals of the means from dates when only small fyke nets were set.

In addition to raw catch data, we calculated the catch similarity between internal and external net sets on each sampling date. While the internal sets were not likely to be completely effective at catching fish that had just passed through the connecting structure, we assumed a high similarity value (a function of species presence/absence or abundance) was indicative of movement through the connecting structures during or prior to sampling. We used four different species richness and similarity measures to compare fish catches inside and outside the wetlands on each sampling date. Species richness (the total number of species) and Jaccard's Coefficient both use species presence/absence data, avoiding biases in the number of individuals caught due to net sizes or how nets were set. Jaccard's Coefficient (J) was calculated as

$$J = \frac{a}{a + b + c}$$

where a was the number of species caught on either side of the structure, b was the number of species unique to the internal net, and c is the number of species unique to the external net. Similarity Ratio (SR) and Percentage Similarity (PS) both included the species abundance and were calculated as

$$SR = \frac{\sum_1^k y_{k,i} \cdot y_{k,e}}{\sum_1^k {y_{k,i}}^2 + \sum_1^k {y_{k,e}}^2 - \sum_1^k y_{k,i} \cdot y_{k,e}}$$

and

$$PS = \frac{200 \cdot \sum_1^k \text{minimum}(y_{k,i}, y_{k,e})}{\sum_1^k y_{k,i} + \sum_1^k y_{k,e}}$$

where y was the abundance of the kth species in the internal (i) or external (e) set. We used Kruskal-Wallis tests to assess if the fish community metrics were different among the three different structures. Because of our small sample sizes (5-7 sample periods for each site and no more than 20 species captured in any net), we considered $\alpha \leq 0.05$ to be highly significant and $0.05 < \alpha \leq 0.1$ to be moderately significant.

We also calculated Wetland Fish Index (WFI) scores for inside and outside each wetland. The WFI is an indicator of overall health of Great Lakes coastal wetlands and is highly correlated with water quality (Seilheimer and Chow-Fraser 2006, 2007). We used the presence absence version of the index, WFI (PA), in order to compare our findings with values of other Lake Erie wetlands reported in Wells et al. (2002) and Seilheimer and Chow-Fraser (2007) and to avoid any potential bias in catch numbers due to net sizes or how they were set . For species for which water-quality optimum and tolerance values were not available in Seilheimer and Chow-Fraser (2007), we estimated our own values based on similar species and preference and tolerance information from Becker (1983). In total, there were 10 of 52 species for which we estimated our own values, which made up an average of 13% of our species richness at each location (range: 7-19%). We do not believe estimating values influenced the results because 1) we are confident in our preference and tolerance values given the data available from closely related species (e.g., Yellow Bullhead values were estimated based on closely related Black and Brown Bullhead), 2) the low number of species with estimated values compared to the total number of species at each site, and 3) the estimated values were the same for all sites, thereby applying any error to each site where that species was captured.

Results

Water flow in or out of the wetlands varied across the sites. At the fish ladder site, water always was flowing out of the wetland, although the flow rates at the top of the first weir were low (mean = 0.04 m/s, range 0 – 0.12) and there was only measurable flow below the last weir of the ladder on 23 September 2015 (0.06 m/s). At the culvert site, flows were variable in direction and strength (mean flow = 0.33 m/s, range 0 – 1.07). On the first sampling date, water was flowing into the wetland when the nets were set and out of the wetland when the nets were pulled. On the second sampling date, flow directions were reversed. On the third sampling date water was flowing out both days. During July, water was observed flowing in but was not measurable (i.e., < 0.03 m/s). In fall, water was always flowing out of the wetland through the culvert but not at measurable velocities. At the swing gate site, flows were generally low and unmeasurable, although some water was observed leaking through cracks and holes in the sluice gates. The swing and sluice gates were fully closed on all but one sampling date (13 May 2015) when one sluice gate was partially opened and we measured a flow of 0.24 m/s out of the wetland.

In total, we captured 11,192 fish weighing a total of 847 kg (Table 1). Fish catch rate (fish per net night) was highest at the culvert site, followed by the fish ladder site and the swing gate site (Table 2). Catch rate was lower within the wetland than outside the wetland at the fish ladder (paired t-test, $t = 1.84$, df = 6, $p = 0.05$) and the swing gate ($t = 3.16$, $df = 4$, $p = 0.02$) sites. Low catch rates in these wetlands could indicate a lack of fish moving into the wetlands, but also could be due to effectiveness of setting the nets to enclose the structure on the interior side (e.g.,

Figure 2). However, catch rates inside and outside the wetland were not significantly different at culvert site (t = 1.34, df = 5, p = 0.11) despite setting the nets in a similar fashion.

Catch rates at all netting locations were relatively high in spring, declined in summer, and highest in fall (Table 2). The changes in catch rates were linked to species and life-stages captured. In spring, adult bullhead, crappie, Longnose Gar, and Quillback were frequently captured. In summer, young-of-the-year Gizzard Shad were abundant, and by fall, young-of-the-year centrarchids were the dominant species. The change is species and ages captured was reflected in the mean lengths of fish captured, which was highest in spring and lowest in fall (data not shown).

A total of 40 different species were captured during sampling (Table 1). At the fish ladder, 30 different species were captured in the river and only 14 in the wetland. At the culvert site, 28 species were caught in the bay and 23 were caught in the wetland. The swing gate site had 22 species in the bay and only 15 species above the swing gates. In comparison, at Metzger Marsh, 38 species were captured entering or exiting the marsh, but in 410 d of sampling (Table 1). The most common fish species captured varied. Phytophillic species like bullhead were among the most numerous in our sampling, but not in the top five for Metzger Marsh. In addition, centarchids (Bluegill, Pumpkinseed and crappie) and Gizzard Shad were very common at many of our netting locations.

All measures of community similarity were generally highest at the culvert and lower (but similar) at the swing gate and fish ladder (Figure 3). Species richness was higher outside the wetland than within on all but two occasions, both at the culvert site (Table 1). We then compared difference in species richness (outside – inside) as a measure of community similarity. In general, the difference in species richness was lowest (i.e., similarity highest) at the culvert

(Figure 3), although the site effect was only moderately significant (Kruskal-Wallis: $\chi^2 = 5.40$, $df = 2$, $p = 0.067$). There was no difference in Jaccard's Coefficient among sites ($\chi^2 = 2.95$, $df = 2$, $p = 0.229$), but Jaccard's Coefficient only considers species presence or absence. Both Similarity Ratio and Percent Similarity were highly significant ($\chi^2 = 7.51$, $df = 2$, $p = 0.023$) and moderately significant ($\chi^2 = 4.98$, $df = 2$, $p = 0.083$), respectively, with the highest similarities at the culvert site (mean *SR* = 0.33, mean *PS* = 36.7) and lower similarities at the fish ladder (0.03, 9.1) and swing gate (0.07, 16.7) sites.

The Wetland Fish Index, a measure of wetland health, using presence/absence data was highest inside the wetland at the culvert site and lowest inside the wetland at the swing gate site (Figure 4). Furthermore, WFI (PA) was lower inside than outside the wetland at the fish ladder and swing gate sites. Only the culvert site had a higher WFI (PA) inside the wetland. Overall, WFI (PA) values in this study were lower than those calculated for the community of fishes immigrating and emigrating from Metzger Marsh through open screw gates (Figure 4, Wells et al. 2002). In addition, the wetlands separated from Lake Erie by the fish ladder and swing gates had significantly lower WFI (PA) values than the mean of eight Lake Erie coastal wetlands that had no human-made structures blocking their connections to the lake (Seilheimer and Chow-Fraser 2007; Figure 4).

There were several qualitative observations of the effectiveness of the passage structures. At the fish ladder, both Emerald Shiners and Gizzard Shad were seen inside the ladder on different dates. Emerald Shiner were caught on both sides of the ladder, but Gizzard Shad were only caught on the river side during this study. Two wetland fishes, Longnose Gar and Northern Pike, were caught on the river side but never captured or seen in the wetland. At the culvert, Bullhead and Quillback were abundant both inside and outside the wetland on the same sampling dates,

but less common or absent at both locations on other sampling dates. Furthermore, adult Gizzard Shad were captured on the wetland side of the culvert in spring and juveniles were captured both inside and outside the wetland in summer and fall. At the swing gate, White and Black Crappie were far more abundant on the bay side (233 captured) than on the wetland side (3 captured) of the swing gates, as were bullhead (197 captured on bay side and 17 caught on wetland side).

Discussion

Our results suggest that the culvert at Great Egret Marsh allowed the most fish passage of any the control structures we studied. The catch rates and species richness, at the culvert site were similar inside and outside the wetland resulting in the highest overall community similarity. Furthermore, the WFI inside the culvert site was similar to outside and similar to the mean WFI of eight Lake Erie wetlands without human barriers to the lake, suggesting that passage was not significantly affected by the culvert. In contrast, the fish ladder and swing gate sites had low catch rates and species richness in the wetland, their fish communities on either side of the structures had low similarity, and they had lower WFI values inside than outside the wetland and in comparison to other Lake Erie coastal wetlands. These results were supported by qualitative observations of catches which suggest that several common wetland species (e.g., Longnose Gar Gizzard Shad, crappie and bullhead) were not moving through the ladder or swing gates in significant numbers. We did not expect to find fish moving through the swing gates at Winous Point, but it was surprising that few fish appeared to use the Blausey Tract fish ladder during our study.

Diked wetlands, like those in this study, have been shown contain fewer species than more natural wetlands for two main reasons: hydrologic isolation and proximity to other habitats. For

example, three of the five most commonly caught species at Metzger Marsh can be classified as lake species (Emerald Shiner, Spottail Shiner, and White Perch; Table 1). These species were relatively uncommon at our sites, especially within the wetlands, due to both distance and connectivity to Lake Erie. A comparison of wetlands near the Winous Point site found more species in undiked wetlands, presumably because the diked wetlands were spatially and hydrologically separated from the open lake (Johnson et al. 1997). Additional data on growth and condition of fishes in these wetlands supported the hypothesis that the diked wetland was biologically isolated despite management that included high volumes of periodic water exchange (Johnson et al. 1997, Markham et al. 1997). Others have reported similar findings that decreased connectivity reduces species diversity in diked Great Lakes coastal wetlands (Bouvier et al. 2009, Kowalski 2010, Kowalski et al. 2014). It has also been proposed that diversity in undiked wetlands may be higher because they may contain more non-phtyophillic species than diked wetlands that are heavily vegetated and offer very little habitat suitable for species that tend to be more pelagic. While open water species may not be dependent on wetlands or serve as major components of their foodweb, these fishes may still benefit from even temporary access to coastal wetlands.

Fish move in and out of coastal wetlands on daily, seasonal, and lifetime time scales (Jude and Pappas 1992). In Crane Creek, a Lake Erie wetland complex located near our sample sites, more than 30,000 fish entered and approximately 24,000 left the wetland each day during summer (Kowalski 2010). At Metzger Marsh, nearly 300,000 fish entered or exited over 410 d in just two of the five screw gates (Wells et al. 2002). The motivation for daily fish movements may be related to spawning, foraging or seeking shelter. Seasonally, fishes like Northern Pike, Gizzard Shad, and Common Carp use wetlands for spawning and/or nurseries before retreating to

the lake (Becker 1983, Jude and Pappas 1992). Outside the wetlands we studied, catches of adult fish that were possibly seeking spawning sites, were highest in spring. But relatively few adults were observed inside the wetlands hydrologically separated by the fish ladder or the swing gates. In contrast, at the culvert site where adults could move into the wetland, we found the highest catch rates for juvenile fish in summer (e.g., young-of-the-year Gizzard Shad) and fall (e.g., young-of-the-year centrarchids). Still other fishes may visit infrequently during their lifetime. But if connectivity to the larger aquatic ecosystem is broken, as can be for diked wetlands, it results in wetlands functioning as "inland wetlands adjacent to lakes" rather than fully functioning coastal wetlands (Wilcox and Whillans 1999). And even if an individual diked wetland appears to maintain a healthy and diverse fish community, the lack of connection with the lake and other wetlands could limit fish movement and gene flow (Midwood and Chow-Fraser 2015).

Unfortunately, we found very limited evidence that fish passed through the fish ladder during our sampling. Furthermore, the passing through the ladder were not representative of the potential source community. Pool-and-weir fish ladders create resting areas for fish passing upstream but still require that the fish leap or pass through very shallow but fast current from pool to pool. These acts may not be common for Midwestern species used to low gradient streams and rivers common to the Great Lakes basin. While fish ladders have been used with some success on coastal streams and rivers particularly for anadromous fishes, a review of fish passage success suggests that average passage rates for salmonids was actually lower than expected, around 60% (Noonan et al. 2011). For non-salmonids, passage rates near or far below 20% have been reported (Mallen-Cooper and Brand 2007, Noonan et al. 2011). Pool-and-weir fish ladders, especially those with little elevation change like the one at the Blausey Tract, are

one of the most effective types of passages for some fishes. But they still may not be successful at passing non-salmonids especially if flow is high, water depth in the weir orifices is low (as was observed in this study) or the species are not attracted to the ladder (Bunt 2001, Pratt et al. 2009, Noonan et al 2011). Therefore, it is not surprising that phytophillic species like Longnose Gar and Northern Pike were apparently not using the Blausey Tract fish ladder to move upstream.

While the culvert afforded the best possibility of fish movement, passage through culverts is variable and dependent on flow, species, and size (Warren and Pardew 1998, Briggs and Galarowicz 2013). At Great Egret Marsh, the direction and velocity of flow was observed to change frequently, so fish could have been actively swimming through the culvert or simply being pushed through with the current. We need a better understanding of fish passage requirements for species other than salmonids and to engineer passage devices appropriate for fishes of different sizes, behaviors, and physical abilities (Cooke and Hinch 2013). Fortunately, if a connection can be established fish may move into a wetland, reproduce, and have their offspring emigrate to grow and contribute to the at large fish community (Oele et al. 2015), as was seen at Metzger Marsh during the period when an open connection to Lake Erie was maintained (Wells et al. 2002).

Many authors have argued that reconnecting wetlands will improve fish diversity and, possibly, production (e.g., Jude and Pappas 1992). Yet there is a history of disregard fish passage in wetland management because waterfowl and vegetation are often the traditional targets for wetland restoration and management (Rogers et al. 1994, Monfils et al. 2014). Opening wetland connections for fish migration can be optimal when fish are highly valued, especially in comparison to waterfowl (Bloczynski et al. 2010). Given that the fisheries of Lake Erie have an

economic impact of more than $1 billion USD (Southwick Associates 2013), the value of fish is certainly high. Furthermore, it is clear that wetlands are important in maintaining a healthy and diverse fishery (Trebitz and Hoffman 2015). In addition to direct effects on fisheries, recent research has demonstrated other benefits of a properly connected and functioning wetland including: improved water quality through nutrient sequestration (Mitsch and Wang 2000), increased productivity (Cronk and Mitsch 1994), and more diverse vegetation (Herrick and Wolf 2005, Mitsch et al. 2014), all of which could benefit both fish and birds (Monfils et al. 2014). Consequently, there has been recent interest and action in restoring Great Lakes coastal wetlands. During the first five years of the Great Lakes Restoration Initiative (GLRI), more than 40,000 ha (100,000 acres) of Great Lakes coastal wetlands have been protected or restored (GLRI 2015).

But reconnecting Great Lakes coastal wetlands depends on more than successfully allowing fish move freely. The GLRI has made wetland conservation and restoration a focus area and has removed more than 500 barriers to fish movement, but mostly from tributaries, not wetlands (GLRI 2015). In part this may be because of uncertainty about the consequences and management of reconnecting formerly diked wetlands. While it certainly would be ideal to allow complete movement of water and biota between and within wetlands, there are risks that are not fully understood. Open connections could accelerate invasion by non-indigenous plants and animals. Some of these invaders could subsequently accelerate degradation of the connected wetland (e.g., Common Carp; Lougheed et al. 1998, Bajer et al. 2009). On the other hand, even diked wetlands are still vulnerable to invasion (Herrick and Wolf 2005) and stability in diked wetlands can result in the loss of habitat diversity if invasive macrophytes become more dominant (Monfils et al. 2014). The best alternative may be to maintain a more continuous and

passable connection that helps limits potential nuisance species (e.g., French et al. 1999) and also allows the ability to close off the wetland and apply various management tools (e.g., water manipulation, chemical application, planting) as needed to periodically reset the system (Rogers et al. 1994, Kowalski 2010, Boys et al. 2012, Monfils et al. 2014). Continuous and systematic management of wetlands may prevent beneficial use of the wetland by fishes (Kowalski et al. 2014), but the effects of only periodic management deserve further study. Toward this end, fish ladders offer some promise, but work needs to be done to determine the best designs that provide attraction and passability for Great Lakes fishes.

In conclusion, our data suggest that open culverts facilitate fish movement into Great Lakes coastal wetlands to a greater degree than currently implemented pool-and-weir fish ladder designs, and that pool-and-weir fish ladders provide limited connectivity similar to structures that are frequently closed (i.e., swing and sluice gates). Future reconnection, creation, or restoration of wetlands should include hydrologic connections that promote movement and reduce required management effort (Wilcox and Whillans 1999, Wells et al. 2002, Boys et al. 2012, Kowalski et al. 2014). Where promoting fish access is a priority, this may include limiting intensive water level management, a tool used to promote desirable vegetation and waterfowl communities, and instead maintaining an open connection throughout a greater period of the year. In the long run, promoting more natural and connected wetland systems, if done properly, should support natural function of the whole food web and benefit the entire ecosystem.

Acknowledgements

Funding for this project came from the Lake Erie Protection Fund. The Ohio Lake Erie Commission administers Ohio's Lake Erie Protection Fund, which was established to finance

research and on-the-ground projects aimed at protecting, preserving and restoring Lake Erie and its watershed. The Fund is supported through donations to the Lake Erie Commission or through purchase of a Lake Erie license plate featuring the Marblehead Lighthouse or Lake Erie life preserver. SB's participation in the F.T. Stone Laboratory's Research Experience for Undergraduates Program came from the F.T. Stone Laboratory Research Endowment, the Thomas Huxley Langlois Research Fellowship, and the John L. Crites Research Fellowship. The Nature Conservancy aided in project development, and provided staff and volunteers to assist fish sampling. Ohio Sea Grant and Ohio State University's F.T. Stone Laboratory helped secure funding, provided equipment and staff time for collecting data. Finally, the authors would like to thank Tory Gabriel, Dale Matox, Erin Monaco, Matt Thomas and all our field assistants and volunteers for their help. Frank Lopez provided a review of an earlier version of this manuscript. John Simpson and Brendan Shirkey (Winous Point Marsh Conservancy) graciously allowed access to Winous Point and assisted with sampling.

References

Bajer, P. G., G. Sullivan, and P. W. Sorensen. 2009 Effects of a rapidly increasing population of common carp on vegetative cover and waterfowl in a recently restored Midwestern shallow lake. Hydrobiologia 632: 235-245.

Becker, G. C. 1983. Fishes of Wisconsin. The University of Wisconsin Press, Madison, WI.

Bloczynski, J. A., W. T. Bogart, B. F. Hobbs, and J. F. Koonce. 2000. Irreversible investment in wetlands preservation: Optimal ecosystem restoration under uncertainty. Environmental Management 26: 175-193.

Bouvier, L. D., K. Cottenie, and S. E. Doka. 2009. Aquatic connectivity and fish metacommunities in wetlands of the lower Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 66: 933–948.

Boys, C. A., F. J. Kroon, T. M. Glasby, and K. Wilkinson. 2012. Improved fish and crustacean passage in tidal creeks following floodgate remediation. Journal of Applied Ecology 49: 223-233.

Briggs, A. S., and T. L. Galarowicz. 2013. Fish passage through culverts in central Michigan warmwater streams. North American Journal of Fisheries Management 33: 652-664.

Bunt, C. M. 2001. Fishway entrance modifications enhance fish attraction. Fisheries Management and Ecology 8: 95-105.

Cooke, S. J., and S. G. Hinch. 2013. Improving the reliability of fishway attraction and passage efficiency estimates to inform fishway engineering, science, and practice. Ecological Engineering 58: 123-132.

Cronk, J. K., and W. J. Mitsch. 1994. Aquatic metabolism in four newly constructed freshwater wetlands with different hydrologic inputs. Ecological Engineering 3: 449-468.

French, J. R. P. III, D. A. Wilcox, and S. J. Nichols. 1999. Passing of Northern Pike and Common Carp through experimental barriers designed for use in wetland restoration. Wetlands 19: 883–888.

Great Lakes Restoration Initiative. 2015. Great Lakes Restoration Initiative Report to Congress and the President, Fiscal Years 2010-2014. Available: http://greatlakesrestoration.us/pdfs/21050720-report_to_coongress.pdf. (Accessed: October 2015).

Herdendorf, C. E. 1987. The ecology of the coastal marshes of western Lake Erie: a community profile. U.S. Fish and Wildlife Service, Office of Biological Services, Washington, DC. Biological Report 85 (7.9).

Herrick, B. M., and A. T. Wolf. 2005. Invasive plant species in diked vs. undiked Great Lakes wetlands. Journal of Great Lakes Research 31: 277-287.

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

Jude, D. J., and J. Pappas. 1992. Fish utilization of Great Lakes coastal wetlands. Journal of Great Lakes Research 18: 651–672.

Kowalski, K. P. 2010. Overcoming Barriers to Coastal Wetland Ecosystem Rehabilitation: Strategies for the Great Lakes. Ph.D. Dissertation. University of Michigan, Ann Arbor, MI.

Kowalski, K. P., M. J. Wiley, and D. A. Wilcox. 2014. Fish assemblages, connectivity, and habitat rehabilitation in a diked Great Lakes coastal wetland complex. Transactions of the American Fisheries Society. 143: 1130-1142.

Lougheed, V. L., B. Crosbie, and P. Chow-Fraser. 1998. Predictions on the effect of common carp (*Cyprinus carpio*) exclusion on water quality, zooplankton, and submergent

macrophytes in a Great Lakes wetland. Canadian Journal of Fisheries and Aquatic Science 55: 1189-1197.

Mallen-Cooper, M., and D. A. Brand. 2007. Non-salmonids in a salmonid fishway: What do 50 years of data tell us about past and future fish passage? Fisheries Management and Ecology 14: 319-332.

Markham, C. A., W. E. Lynch Jr., D. L. Johnson, and R. W. Petering. 1997. Comparison of white crappie populations in diked and undiked Lake Erie. Ohio Journal of Science 97: 72–77.

Midwood, J. D. and P. Chow-Fraser. 2015. Connecting coastal marshes using movements of resident and migratory fishes. 2015. Wetlands 35: 69-79.

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

Mitsch, W. J., and J. G. Gosselink. 2007. Wetlands, 4th edition. John Wiley & Sons, Hoboken, NJ.

Mitsch, W. J., L. Zhang, E. Waletzko, and B. Bernal. 2014. Validation of the ecosystem services of created wetlands: Two decades of plant succession, nutrient retention, and carbon sequestration in experimental riverine marshes. Ecological Engineering 72: 11-24.

Monfils, M. J., P. W. Brown, D. B. Hayes, G. J. Soulliere, and E. N. Kafcas. Breeding bird use and wetland characteristics of diked and undiked coastal marshes in Michigan. The Journal of Wildlife Management 78: 79-92.

Noonan, M. J., J. W. A. Grant, and C. D. Jackson. 2011. A quantitative assessment of fish passage efficiency. Fish and Fisheries 13: 450-464.

Oele, D. L., J. D. Hogan, and P. B. McIntyre. 2015. Chemical tracking of northern pike migrations: If we restore access to breeding habitat, will they come? Journal of Great Lakes Research 41: 853-861.

Pratt, T. C., L. M. O'Connor, A. G. Hallett , R. L. McLaughlin , C. Katopodis , D. B. Hayes and R. A. Bergstedt. 2009. Balancing aquatic habitat fragmentation and control of invasive species: Enhancing selective fish passage at sea lamprey control barriers. Transactions of the American Fisheries Society 138: 652-665.

Rogers, D. R., B. D. Rogers, and W. H. Herke. 1994. Structural marsh management effects on coastal fishes and crustaceans. Environmental Management 18:351–369.

Seilheimer, T. S., and P. Chow-Fraser. 2006. Development and use of the Wetland Fish Index to assess the quality of coastal wetlands in the Laurentian Great Lakes. Canadian Journal of Fisheries and Aquatic Sciences 63: 354-366.

Seilheimer, T. S., and P. Chow-Fraser. 2007. Application of the Wetland Fish Index to northern Great Lakes marshes with emphasis on Georgian Bay coastal wetlands. Journal of Great Lakes Research 33 (Special Issue 3): 154-171.

Southwick Associates. 2013. Sportfishing in America: An Economic Force for Conservation. Produced for the American Sportfishing Association, Alexandria, VA.

Trebitz, A. S. and J. C. Hoffman. 2015. Coastal wetland support of Great Lakes fisheries: Progress from concept to quantification. Transactions of the American Fisheries Society 144: 352-372.

Warren, M. L., and M. G. Pardew. 1998. Road crossings as barriers to small-stream fish movement. Transactions of the American Fisheries Society 127: 637-644.

Wells, S. E., J. R. McClain, and T. D. Hill. 2002. Fish passage between Lake Erie and Metzger Marsh: Monitoring of an experimental fish passage structure, 1999-2002. Final Report. U.S. Fish and Wildlife Service. Alpena, Michigan.

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

Table 1. Fish catches for four wetland sites sampled fish movement. Blausey Tract, Great Egret Marsh, and Winous Point were sampled April to October 2015, while the Metzger Marsh data were collected from 199-2002 (Wells et al. 2002). Each site had a different passage structure listed below the site name and further described in the text.

		Blausey Tract Fish Ladder		Great Egret Marsh Culvert		Winous Point Swing Gates		Metzger Marsh Open Screw Gates
Common Name	Scientific Name	Marsh	River	Marsh	Bay	Marsh	Bay	Marsh
Alewife	*Alosa pseudoharengus*	0	1 (<0.1)	0	0	0	0	2,258 (1.9)
Bigmouth Buffalo	*Ictiobus cyprinellus*	0	0	0	0	0	0	21 (<0.1)
Black Bullhead	*Ameiurus melas*	3 (0.6)	2 (0.1)	33 (0.6)	10 (0.5)	**8 (11.0)**	23 (4.1)	0
Black Crappie	*Pomoxis nigromaculatus*	**92 19.2)**	101 (3.8)	27 (0.5)	27 (1.4)	2 (2.7)	**62 (11.1)**	52 (<0.1)
Bluegill	*Lepomis macrochirus*	**118 (24.6)**	**119 (4.4)**	**183 (3.3)**	**296 (15.7)**	**10 (3.7)**	**25 (4.5)**	372 (0.3)
Bluntnose Minnow	*Pimephales notatus*	0	0	6 (<0.1)	**210 (11.1)**	0	0	134 (0.1)
Bowfin	*Amia calva*	0	15 (0.6)	1 (<0.1)	2 (0.1)	2 (2.7)	1 (0.2)	909 (0.8)
Brown Bullhead	*Ameiurus nebulosus*	**200 (41.8)**	31 (1.2)	**306 (5.5)**	106 (5.6)	2 (2.7)	**131 (23.5)**	15 (<0.1)
Channel Catfish	*Ictalurus punctatus*	0	1 (<0.1)	0	0	1 (1.4)	10 (1.8)	42 (<0.1)
Common Carp	*Cyprinus carpio*	**27 (5.6)**	5 (0.2)	3 (<0.1)	9 (0.5)	**4 (5.5)**	7 (1.3)	627 (0.5)
Creek Chub	*Semotilus atromaculatus*	1 (0.2)	0	0	0	0	0	1 (<0.1)
Emerald Shiner	*Notropis atherinoides*	4 (0.8)	**1,304 (48.7)**	2 (<0.1)	**78 (4.1)**	0	0	**37,735 (31.8)**
Freshwater Drum	*Aplodinotus grunniens*	0	10 (0.4)	0	31 (1.6)	0	5 (0.9)	286 (0.2)
Gizzard Shad	*Dorosoma cepedianum*	0	**370 (13.8)**	**1,862 (33.8)**	**231 (12.3)**	**16 (21.9)**	14 (2.5)	**59,270 (49.9)**
Golden Redhorse	*Moxostoma erythrurum*	0	1 (<0.1)	0	0	0	0	0
Golden Shiner	*Notemigonus crysoleucus*	1 (0.2)	3 (0.1)	27 (0.5)	10 (0.5)	0	0	68 (0.1)
Goldfish	*Carassius auratus*	1 (0.2)	8 (0.3)	2 (<0.1)	7 (0.4)	0	0	1,395 (1.2)
Green Sunfish	*Lepomis cyanellus*	0	2 (0.1)	7 (0.1)	0	0	0	3 (<0.1)
Johnny Darter	*Etheostoma nigrum*	0	0	0	8 (0.4)	0	0	0
Largemouth Bass	*Micropterus salmoides*	6 (1.3)	8 (0.3)	20 (0.4)	55 (2.9)	0	1 (0.2)	**2,803 (2.4)**
Longear Sunfish	*Lepomis peltastes*	0	0	2 (<0.1)	0	0	0	0
Longnose Gar	*Lepisosteus osseus*	0	**271 (10.1)**	1 (<0.1)	6 (0.3)	0	1 (0.2)	18 (<0.1)
Mimic Shiner	*Notropis volucellus*	0	0	0	0	0	0	1 (<0.1)
Northern Brook Silverside	*Labidesthes sicculus*	0	14 (0.5)	0	5 (0.3)	0	0	0
Northern Logperch	*Percina caprodes semifascita*	0	0	0	0	0	0	53 (<0.1)
Northern Pike	*Esox luscius*	0	1 (<0.1)	0	0	0	0	14 (<0.1)

Orange Spotted Sunfish	*Lepomis humilis*	0	1 (<0.1)	0	0	0	1 (0.2)	51 (<0.1)
Pumpkinseed	*Lepomis gibbosus*	**10 (2.1)**	16 (0.6)	**2,907 (52.7)**	**589 (31.3)**	3 (4.1)	11 (2.0)	434 (0.4)
Quillback	*Carpiodes crypinus*	0	0	3 (<0.1)	47 (2.5)	0	2 (0.4)	252 (0.2)
Rainbow Smelt	*Osmerus mordax*	0	0	0	0	0	0	8 (<0.1)
Rainbow Trout	*Oncorhynchus mykiss*	0	0	0	0	0	0	10 (<0.1)
Rockbass	*Amploplites rupestris*	0	0	0	0	0	0	72 (0.1)
Round Goby	*Neogobius melanostomus*	0	3 (0.1)	2 (<0.1)	3 (0.2)	0	0	2,108 (1.8)
Sand Shiner	*Notropis stramineus*	0	0	0	3 (0.2)	0	0	986 (0.8)
Sea Lamprey	*Petromyzon marinus*	0	0	0	0	0	0	3 (<0.1)
Shorthead Redhorse	*Moxostoma macrolepidotum*	0	0	0	0	0	0	2 (<0.1)
Silver Chub	*Macrohybosis storeriana*	0	0	0	0	0	0	1 (<0.1)
Silver Redhorse	*Moxostoma anisurum*	0	0	0	0	0	0	1 (<0.1)
Smallmouth Bass	*Micropterus dolomieu*	0	0	0	0	0	0	33 (<0.1)
Spotfin Shiner	*Cyprinella spiloptera*	0	1 (<0.1)	2 (<0.1)	3 (0.2)	0	0	87 (0.1)
Spottail Shiner	*Notropis hudsonius*	1 (0.2)	**156 (5.8)**	0	24 (1.3)	**4 (5.5)**	6 (1.1)	**4,286 (3.6)**
Spotted Sucker	*Minytrema melanops*	0	11 (0.4)	0	0	0	22 (3.9)	0
Tadpole Madtom	*Noturus gyrinus*	0	0	0	0	1 (1.4)	0	11 (<0.1)
Troutperch	*Percopsis omiscomaycus*	0	0	0	0	0	0	26 (<0.1)
Walleye	*Sander vitreus*	0	0	0	0.0	0	0	133 (0.1)
Western Banded Killifish	*Fundulus diaphanus menona*	0	0	0	1 (0.1)	0	0	0
White Bass	*Morone chrysops*	0	43 (1.6)	0	2 (0.1)	0	8 (1.4)	1,238 (1.0)
White Crappie	*Pomoxis annularis*	5 (1.0)	60 (2.2)	31 (0.6)	53 (2.8)	1 (1.4)	**171 (30.7)**	97 (0.1)
White Perch	*Morone americana*	0	110 (4.1)	**54 (1.0)**	22 (1.2)	2 (2.7)	11 (2.0)	**2,603 (2.2)**
White Sucker	*Catostomus commersoni*	0	0	0	0	1 (1.4)	0	70 (0.1)
Yellow Bullhead	*Ameiurus natalis*	10 (2.1)	7 (0.3)	26 (0.5)	12 (0.6)	**16 (21.9)**	**43 (7.7)**	35 (<0.1)
Yellow Perch	*Perca flavescens*	0	5 (0.2)	9 (0.2)	34 (1.8)	0	2 (0.4)	78 (0.1)

Table 2. Fish catch data and community similarity for three wetland sites sampled (April to October 2015) above and below a potential barrier to fish movement. Each site had a different passage structure listed below the site name and further described in the text. High water in July prevented some sampling.

Site	Date	Total Catch Wetland	Total Catch Bay/River	Species Richness Wetland	Species Richness Bay/River	Jaccard's Coefficient	Similarity ratio	Percentage similarity
Blausey Tract	28-Apr	20	1,288	7	16	0.28	0.004	2.8
Fish Ladder	12-May	282	222	9	14	0.28	0.6	12.7
	27-May	113	153	6	18	0.26	0.03	15.0
	12-Jul	0	16	NA	7	NA	NA	NA
	20-Jul	21	261	7	12	0.19	0.0002	2.1
	23-Sep	7	427	3	18	0.17	0.04	3.2
	7-Oct	36	315	7	18	0.26	0.06	18.6
Mean		**68**	**383**	**7.5**	**14.7**	**0.24**	**0.03**	**9.1**
Great Egret Marsh	29-Apr	184	405	6	17	0.16	0.02	5.8
Culvert	13-May	83	167	11	12	0.64	0.54	54.8
	28-May	47	81	11	10	0.24	0.22	32.8
	13-Jul	No set	72	No set	14	---------- No wetland set ----------		
	20-Jul	1,795	169	11	11	0.53	0.06	13.5
	22-Sep	217	238	13	20	0.46	0.96	81.3
	6-Oct	3,200	753	16	17	0.65	0.20	32.2
Mean		**921**	**269**	**11.3**	**14.4**	**0.45**	**0.33**	**36.7**
Winous Point	29-Apr	14	159	6	8	0.27	0.01	4.6
Swing Gates	13-May	13	16	6	8	0.30	0.19	37.8
	28-May	15	75	4	12	0.33	0.14	28.9
	Jul	---------- No nets set due to high water ----------						
	23-Sep	16	111	7	11	0.29	0.01	6.3
	7-Oct	15	187	1	16	0.06	0.02	5.9
Mean		**15**	**110**	**4.8**	**11**	**0.25**	**0.07**	**16.7**

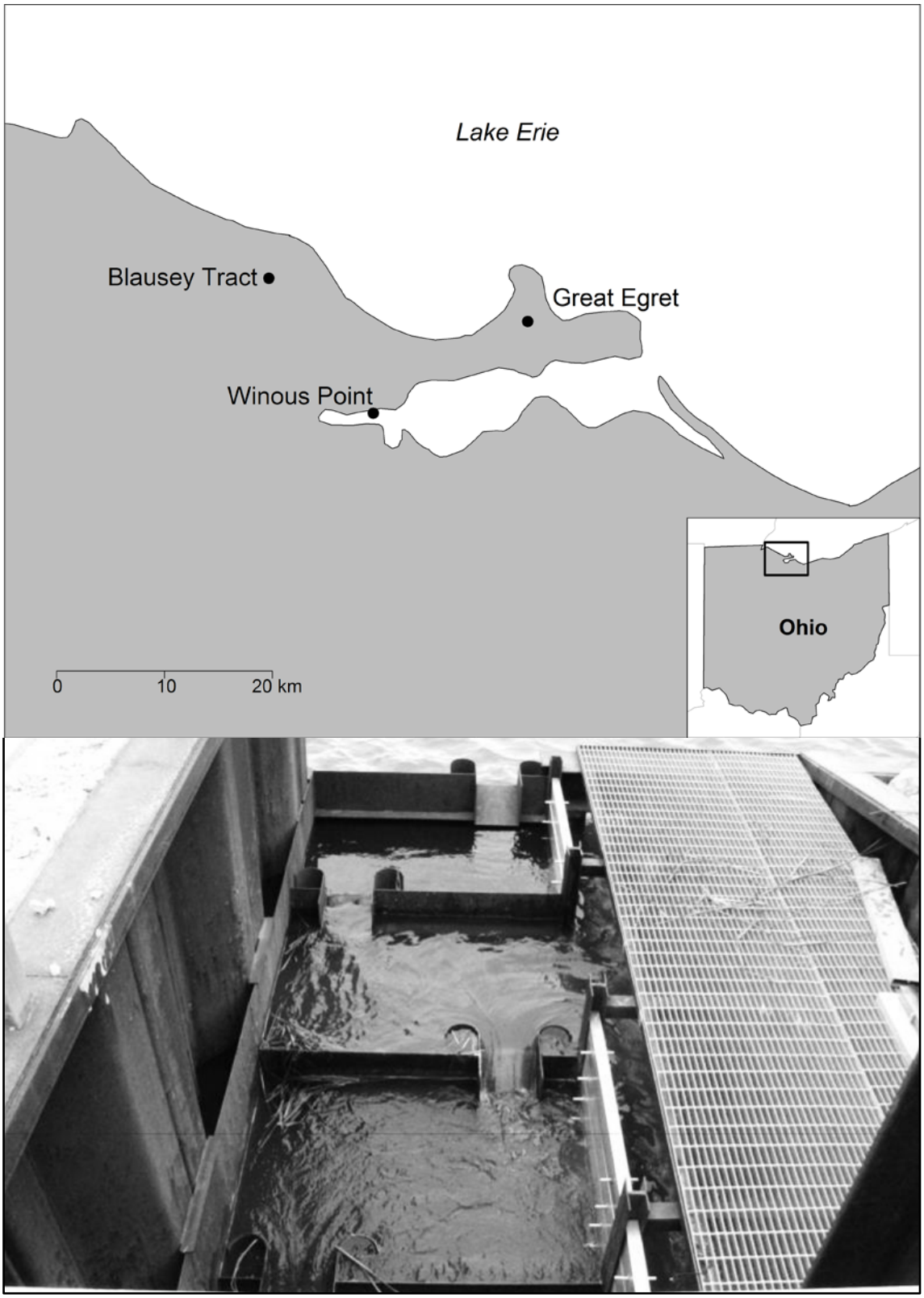

Figure 1. Wetland locations (Top) and photograph of the Blausey Tract fish ladder (Bottom).

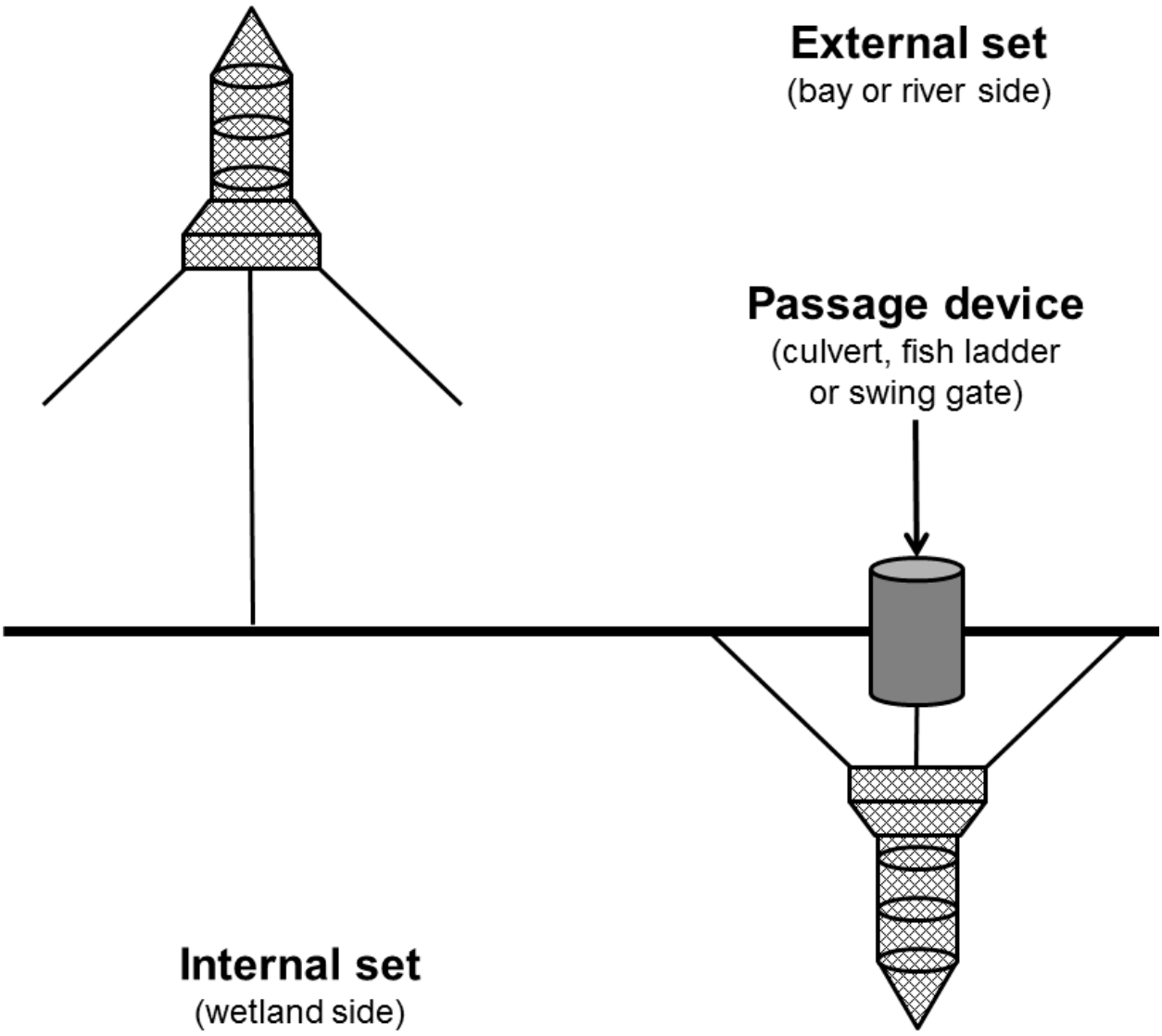

Figure 2. Two fyke nets were set at each site: one net was set perpendicular to shore outside the wetland with the lead extending to the shore and one net was set inside the wetland with the wings completely enclosing the opening of the passage/water control structure.

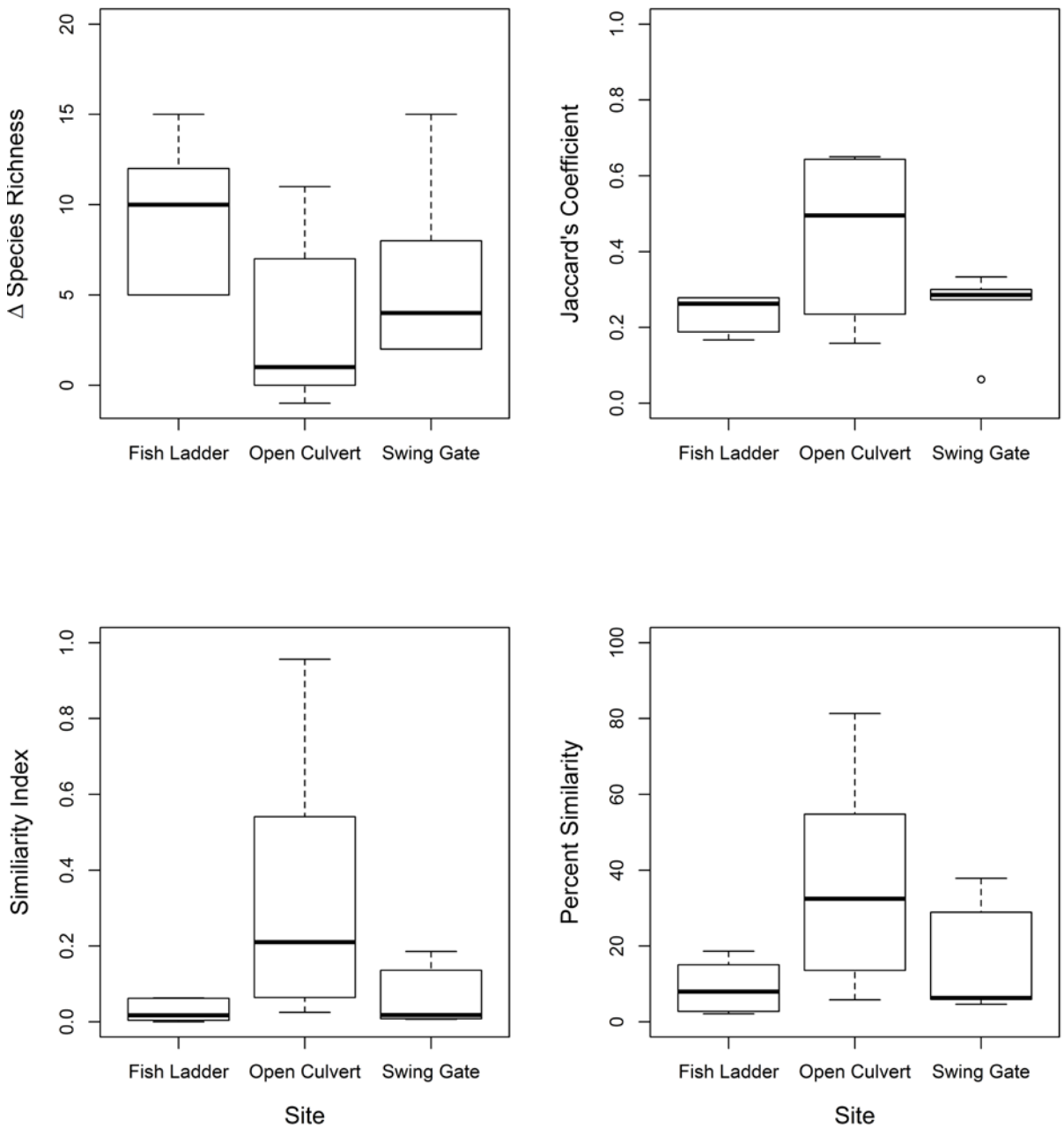

Figure 3. Catch results for the three sites with different structures sampled between April and October 2015: fish ladder (Blausey Tract), culvert (Great Egret Marsh), swing gates (Winous Point). Box plots depict the median (thick horizontal line), the 2nd and 3rd quartiles (box) and the minimum and maximum (whiskers). Outliers are shown as open circles. Δ species Richness is the number of species caught in outside set minus the number of species caught in the inside net set.

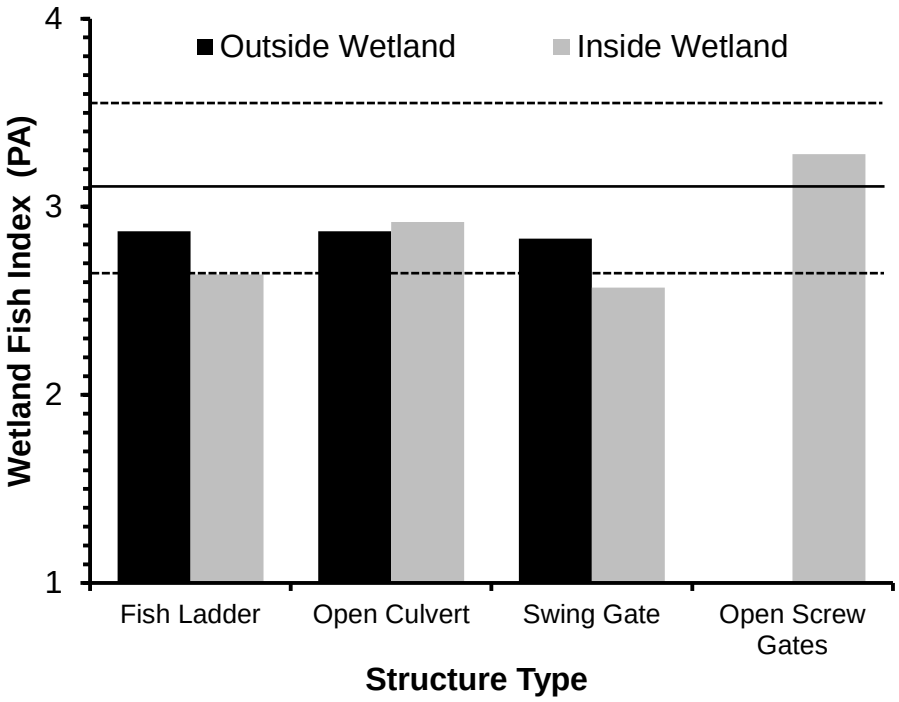

Figure 4. Wetland Fish Index values for wetland sites separated from natural waters by a fish ladder (Blausey Tract), culvert (Great Egret Marsh), swing gates (Winous Point), or open screw gates (Metzger Marsh). Wetland Fish Index for presence absence data, WFI (PA), was calculated as described in Seilheimer and Chow-Fraser (2006 and 2007) where high values are indicative of better ecosystem health. Horizontal lines indicate the mean (solid line) and 95% confidence intervals for WFI data from eight Lake Erie wetlands without structures affecting fish passage (Seilheimer and Chow-Fraser 2007). Data for Metzger Marsh (Wells et al. 2002) were pooled for fish emigrating from or immigrating to the wetland.

Acknowledgements

Funding for this project came from the Lake Erie Protection Fund. The Ohio Lake Erie Commission administers Ohio's Lake Erie Protection Fund, which was established to finance research and on-the-ground projects aimed at protecting, preserving and restoring Lake Erie and its watershed. The Fund is supported through donations to the Lake Erie Commission or through purchase of a Lake Erie license plate featuring the Marblehead Lighthouse or Lake Erie life preserver. SB's participation in the F.T. Stone Laboratory's Research Experience for Undergraduates Program came from the F.T. Stone Laboratory Research Endowment, the Thomas Huxley Langlois Research Fellowship, and the John L. Crites Research Fellowship. The Nature Conservancy aided in project development, and provided staff and volunteers to assist fish sampling. Ohio Sea Grant and Ohio State University's F.T. Stone Laboratory helped secure funding, provided equipment and staff time for collecting data. Finally, the authors would like to thank Tory Gabriel, Dale Matox, Erin Monaco, Matt Thomas and all our field assistants and volunteers for their help. Frank Lopez provided a review of an earlier version of this manuscript. John Simpson and Brendan Shirkey (Winous Point Marsh Conservancy) graciously allowed access to Winous Point and assisted with sampling.