EFFECTIVENESS OF CRITICAL LAKE TROUT (*SALVELINUS NAMAYCUSH*) AND COREGONID REEF SPAWNING HABITAT RESTORATION IN NORTHERN LAKE MICHIGAN: MITIGATING FROM ENVIRONMENTAL AND INVASIVE EGG PREDATOR IMPACTS

and

STATUS AND HISTORY OF CISCO (*COREGONUS ARTEDI*) IN MICHIGAN INLAND WATERS

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ABSTRACT

High-quality nearshore spawning reefs are a rare, critical habitat in Lake Michigan. Anthropogenic impacts including shoreline development, sedimentation, and the introduction of invasive species like Round Goby (*Neogobius melanostomus*) and Rusty Crayfish (*Orconectes rusticus*) have degraded many nearshore reef habitats, threatening three species that use them for spawning: Lake Trout (*Salvelinus namaycush*), Lake Whitefish (*Coregonus clupeaformis*), and Cisco (*C. artedi*). The conservation and restoration of high-quality habitat is critical to the recovery and sustainability of these species, as spawning fish tend to focus on small patches of high-quality habitat. A reef complex near Elk Rapids, Grand Traverse Bay, is the only known spawning reef complex used by Cisco in Lake Michigan; the reef is also used by Lake Trout and Lake Whitefish. A portion of the Elk Rapids reef complex is degraded as a result of a historic iron dock operation, and egg deposition and survival is subsequently low. Baseline rates of invasive egg predators, egg deposition, and egg survival for native reef spawners were quantified on both the adjacent highly productive site, and the degraded site of the reef complex from 2013-2015. Physical characteristics were also quantified on the reference and degraded sites. In August 2015, 450 tons of limestone gravel/rubble were added to improve interstitial depth and habitat quality of the degraded site with the goals of increasing native fish egg deposition and retention and reducing egg loss due to invasive species predation. We examined the effectiveness of the restoration by comparisons to a high-quality reference reef before and directly after restoration. The post-restoration habitat was found to be extremely similar to the reference habitat. Although we found higher seeded egg and bead retention within the restored reef when compared to the reference site and pre-restoration years, we anticipate that

determining the success of this restoration effort will require monitoring across multiple spawning seasons.

Cisco *Coregonus artedi* is a state threatened species that inhabit fewer than 200 inland lakes of Michigan. The majority of these lakes have not been recently evaluated, which has resulted in a lack of information regarding the current status of inland Cisco in Michigan. Latta (1995) examined and classified 153 inland Cisco lakes of Michigan. We have used and expanded on the work of Latta (1995) to examine the ecoregional distribution, trends, and status of Cisco lakes in Michigan. Lack of information on the remaining populations of Cisco is one of the largest impediments to their recovery. We suggest prioritizing sampling efforts within the Battle Creek / Elkhart Outwash Plain (56b) and Interlobate Dead Ice Moraines (56h) ecoregions. The majority of Cisco lakes in Michigan are contained within these two ecoregions, and lakes within these ecoregions are under the greatest threats of habitat degradation. Furthermore, the majority of the lakes in these regions have not been recently sampled. The Inland Cisco Lakes Wildlife Action Plan has suggested management considerations for the reestablishment of Cisco. We suggest an ecoregional approach prioritizing the assessment of the Cisco lakes within these ecoregions of higher risk, as the majority of the lakes within these ecoregions are of unknown status.

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INTRODUCTION

Nearshore reef habitats in Lake Michigan originated from Pleistocene glacial deposits of sedimentary rocks deposited into underwater beach-ridges through historical water-level change and wave action (Thompson and Baedke 1995, Janssen et al. 2005). Nearshore reefs are critical for spawning and development of many native species, yet high-quality reefs are becoming increasingly rare in Lake Michigan (Rutherford et al. 2009). Anthropogenic influences such as navigation, shoreline erosion and hardening, and excess nutrients and pollution in addition to invasive species have degraded nearshore reefs (McLean et al. 2015). These threats have resulted in the need to "protect and restore reef spawning habitats" as one of the six environmental objectives identified by the Lake Michigan Committee of the Great Lakes Fishery Commission (Rutherford et al. 2009).

The creation, rehabilitation, and restoration of reef habitats has occurred for over 40 years in the Great Lakes, and has been associated with attracting fish, improving recreational catch rates, and increased egg densities on the improved reef habitat (McLean et al. 2015). However, the majority of these reef projects focus on the biological outcomes (e.g. increased egg deposition, spawner abundance), and fail to quantitatively examine the physical habitat characteristics that relate to the success/failure of the new reef habitat (McLean et al. 2015); thus developing quantitative methods examining the microhabitat characteristics that influence egg deposition and retention are needed. Toward that end, we examined the effectiveness of a nearshore spawning reef restoration with the goal of determining the influence of reef microhabitat characteristics on egg deposition and retention rates, and invasive egg predator densities at the restoration site; and to gain insight on the microhabitat qualities that are associated with productive nearshore spawning reefs through the examination of a high-quality nearshore reference reef.

Lake Trout (*Salvelinus namaycush*), Lake Whitefish (*Coregonus clupeaformis*), and Cisco (*C. artedi*) are three native species that use nearshore reefs for spawning in Lake Michigan. Lake Trout were valuable to both the sport and commercial fisheries in Lake Michigan before 1950 and were extremely ecologically valuable as they provided a stabilizing effect to the fish community through the use of a wider variety of habitats and food resources when compared to other salmonids (Bronte et al. 2008). Cisco was once one of the most important and abundant prey fishes in Lake Michigan (Koelz 1929, Madenjian et al. 2011, Stockwell et al. 2009, Yule et al. 2012). In addition to supporting valuable commercial fisheries (Bronte et al. 2003, Madenjian et al. 2011, Wells and McLain 1973), the planktivorous Cisco were crucial in transferring energy to the predatory fish biomass (Madenjian et al. 2011, Stockwell et al. 2009, Yule et al. 2012). Lake Whitefish is another native, ecologically important, and commercially lucrative species that utilizes Lake Michigan nearshore reefs for spawning. Economically, it is the most important commercial fish in Lake Michigan (Madenjian et al. 2002), despite its reduced numbers in recent years. Although Lake Whitefish is not a species of concern like Cisco and Lake Trout, their spawning presence on Lake Michigan nearshore reefs add immensely to the value of this spawning habitat.

Spawners of these three species deposit eggs over the interstitial spaces of cobble/rubble substrates (Marsden et al. 1995). The selection of spawning habitat depends on many factors including: currents, reef slope, water quality and temperature, substrate size and cleanliness, interstitial space, and interstitial depth (Claramunt et al. 2012, Marsden et al. 1995). Relatively steep slopes are usually accompanied by stronger currents which help to congregate spawning fish and maintain water quality within the spawning reef (Marsden and Krueger 1991, Marsden et al. 1995, Riley et al. 2014). Substrate with interstitial spaces at least 1m deep is characteristic of high-quality nearshore spawning reefs, as shallow substrate depth results in deposited eggs being more susceptible to displacement from wave action (Eshenroder et al. 1995a, Fitzsimons 1996, Marsden et al. 1995). Eggs deposited on high-quality reefs settle within the many small crevices of the deep (1-2m) substrate and are protected from wave action and anoxic conditions. Multiple layers of small, rounded to sub-angular cobble/rubble is the optimal substrate to incubate eggs (Marsden et al. 1995).

The higher interstitial depth and smaller interstitial spaces of high-quality reefs could also protect eggs from invasive egg predators such as the Round Goby (*Neogobius melanostomus*) and Rusty Crayfish (*Orconectes rusticus*). These invasive predators use the interstitial habitat of reefs for protection and foraging (Chotkowski and Marsden 1999, Ray and Corkum 2001). Recruitment of Lake Trout, Lake Whitefish, and Cisco has been negatively impacted by these invasive interstitial predators through predation on their eggs, which has contributed to the lack of recovery of Lake Trout and Cisco in Lake Michigan (Bronte et al. 2003, Chotkowski and Marsden 1999, Claramunt et al. 2005, Jones et al. 1995). High-quality interstitial habitat could allow for eggs to filter down within the reef but minimize access to predators during the incubation period (Biga et al. 1998, Roseman et al. 2011).

A reef complex near Elk Rapids, Grand Traverse Bay, is currently the only known Cisco spawning location in Lake Michigan, and the only spawning reef complex used by Lake Trout, Lake Whitefish, and Cisco in Lake Michigan (Figure 1). Egg deposition for native reef spawners has been monitored at this reef complex from 2013-2015. Although spawning by all of these species has been documented at two locations in the Elk Rapids reef complex, one area has the lowest egg deposition and survival for Lake Trout, Lake Whitefish, and Cisco (Barton et al. 2011). This degraded section of reef habitat was an incidental reef created by historical iron dock construction and operation in the early 1900s, which resulted in differences in microhabitat characteristics when compared to the rest of the reef complex (i.e. unimpacted reference site). Restoration of the degraded site is vital as Lake Trout, Lake Whitefish, and Cisco use this reef complex, and high-quality habitat is important for egg incubation and survival (Claramunt et al. 2005, Marsden et al. 1995).

In August 2015, 450 tons of limestone gravel/rubble (similar to the gravel/rubble at the unimpacted reference site) was added to the sub-optimal reef habitat to determine the impact of increasing the interstitial depth and quality of interstitial spaces on egg deposition and retention rates and invasive egg predator densities. We assessed the effectiveness of the restoration by comparing a number of biotic and abiotic indicators at the degraded site before and directly after restoration and through comparisons of the indicators at the degraded site to the adjacent high-quality reference reef. The indicators were quantified yearly from 2013 – 2015. Abiotic indicators were slope, interstitial depth, interstitial space, substrate size, and current velocity. Biotic indicators included number of invasive egg predators, number of eggs, and egg survival. Our objectives were to quantify the differences in habitat, egg deposition and survival, and invasive egg predators both temporally and between the reference and restoration sites.

METHODS

Study sites

Two separate sites within the Elk Rapids reef complex have been identified as suitable spawning locations for Lake Trout, Lake Whitefish, and Cisco (Barton et al. 2011) (Figure 1). The reference site is a naturally occurring shoal of 2-25cm diameter substrate with interstitial depth of approximately 1m. The original reef at the degraded site was an incidental reef created by the dismantling of the crib structure of an iron company pier in 1918 (Abbot et al. 2011). The pier has been abandoned as of 2015, and has existed as a series of wooden pilings, debris, and old commercial waste (Figure 2). Before the addition of limestone gravel/rubble in August 2015, the pre-restoration habitat quality at the degraded site was extremely poor, with interstitial depth <0.5m. Both sites exist in approximately the same water depth (3-4m) and distance from shore (~850m). The reference site has an area of $350m^2$ and the degraded site area is $354m^2$. The reference site was approximately 400m from the restoration site.

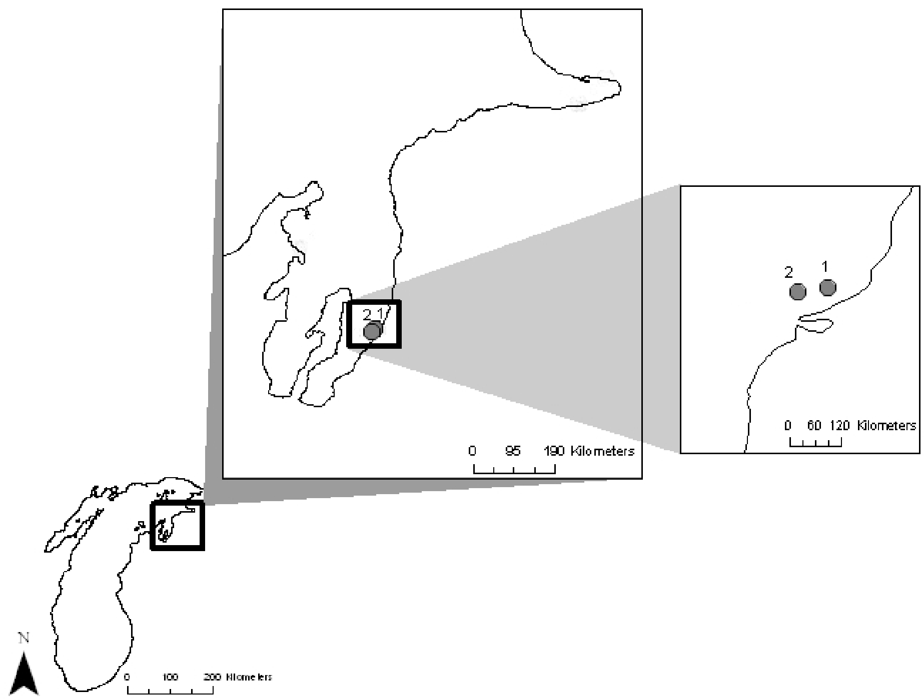

Figure 1. Location of the restoration site (1) and the high-quality reference site (2) in the east arm of Grand Traverse Bay, Lake Michigan.

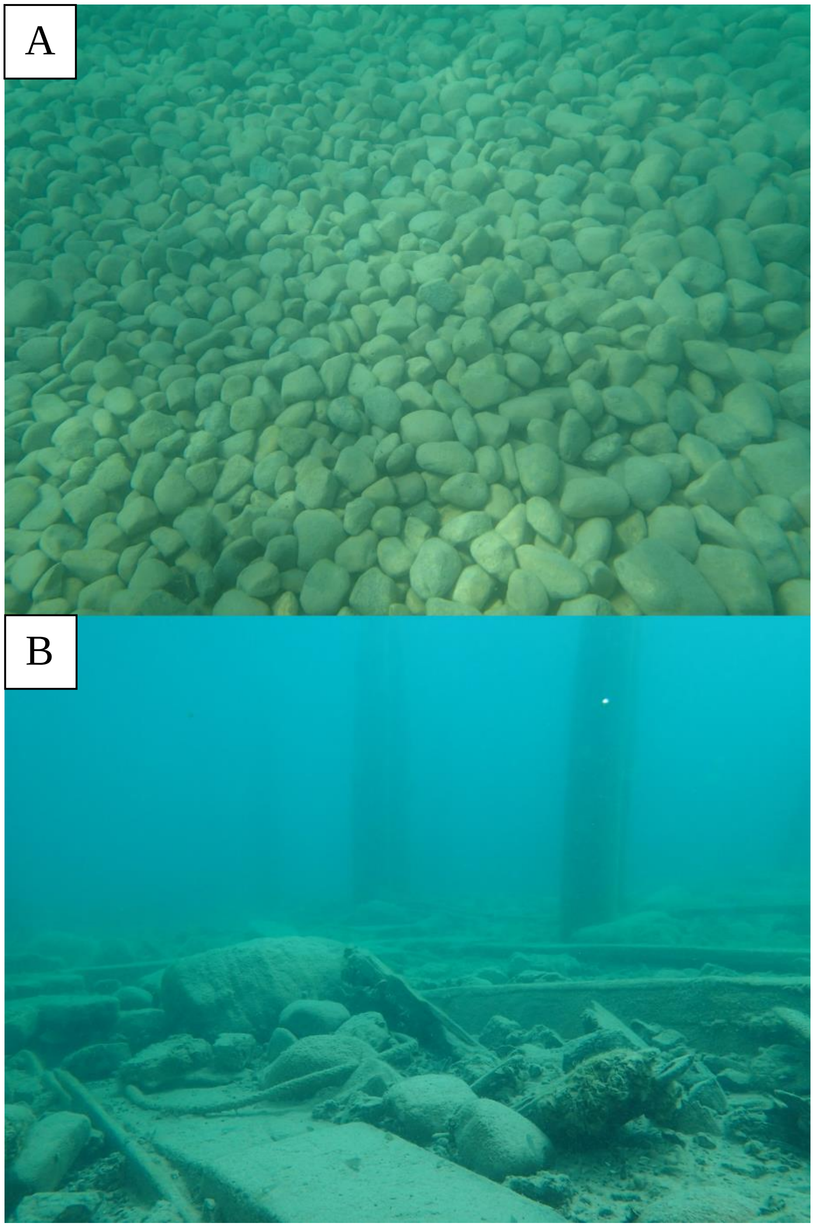

Figure 2. A) Picture of the high-quality reference site habitat. B) Picture of the low-quality habitat at the restoration site before restoration occurred. *Photo credit: Eric Calabro, CMU.*

Reef Restoration

The degraded habitat was a triangle-shaped area at the lake-ward end of the dilapidated Elk Rapids Iron Company pier. This area was once a large crib structure filled with stone and slag, potentially used for mooring and to add structure to the pier (Abbott et al. 2011). The pier was dismantled and abandoned in 1918, and the remaining fill within the crib structure at the end of the pier, along with the natural slope and currents in the area, has attracted Lake Trout, Lake Whitefish, and Cisco for spawning (Abbott et al. 2011, Barton 2010). In August 2015, 450 tons of limestone gravel/rubble were placed on the degraded restoration site. The 4 – 28cm gravel/rubble size and approximately 1 m interstitial depth that were selected for the restoration were based on recommendations from previous examination of the high-quality reference site (Barton 2010) and the literature. The limestone gravel/rubble originated from the Lake Michigan basin near Grand Traverse Bay.

Habitat Measurements

Substrate size and interstitial space were assessed at ten randomly selected locations at the degraded site and the reference site before and after restoration from June through October. A $1m^2$ quadrat was used for scale and to delineate each location. At each location, divers removed a maximum of five layers of rock within the reef and brought each layer to the surface. On the boat, ten randomly selected rocks from each layer were measured (mm) on the x, y, and z axes, and scraped of all *Dreissenid* mussels. Rocks too large to bring to surface were measured underwater with a metric measuring tape.

The volume of interstitial spaces were sampled through the collection of $1m^2$ of rocks at each layer in each random quadrat location. Rocks were collected from within each quadrat to fill a container measuring 0.61m x 0.30m x 0.30m. Rocks were placed in the container as compactly as possible to mimic their orientation on the reef. The container was filled with enough water to just fill the interstitial spaces. The displacement of water was measured to determine the percent interstitial space within the reef.

Reef slope was measured at each of five different locations on both the restoration and reference sites using an ACE Magnetic Angle Locator™ both before and after restoration. Slope measurements were taken 3-5m apart along the reef edges at each site. Interstitial depth was measured at the same five locations at each site by scuba divers. Divers removed layers of gravel/rubble until sand was visible. A meter/survey stick was placed on the sand, perpendicular to the reef face to measure interstitial depth. In areas where the rock layer was too deep to remove (usually > 50cm), a survey stick was extended off the slope of the reef, and the height was measured just off the reef face. Three dynamometers, similar to those described in Bell and Denny (1994), were deployed at each site during the first week of November 2015 to measure maximum current velocity (Figure 3). Before deployment, the dynamometers were calibrated to determine the velocity (m/s) that corresponded to the distance (mm) the stopper traveled as a result of the current. Wind speed data from the National Oceanic and Atmospheric Administration's National Data Buoy Center was used as a broad-scale indicator of environmental disturbance at both sites. Data from Station GTLM4 – Grand Traverse Light (45.211N, 85.550W) from October 15 through December 15 in 2013, 2014, and 2015 were used. Only wind speed data from 225^{o} (SW) to 45^{o} (NW) were used, as these directions have the largest fetch, and would produce the largest disturbance at the nearshore reef sites.

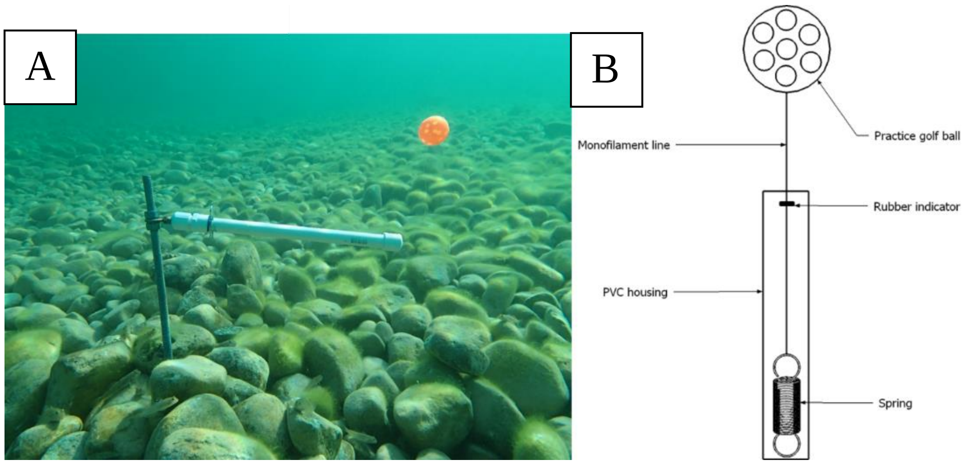

Figure 3. A) Picture of maximum wave velocity dynamometer deployed at the reference site. B) Schematic of the maximum wave velocity dynamometers, spring size 8.73mm x 4.76cm x 0.635mm. *Photo credit: Eric Calabro, CMU.*

Epibenthic Predator Monitoring

Invasive egg predators were monitored through the deployment of standard Gee minnow traps (23cm x 45cm with 0.64cm steel wire mesh) from 2013 – 2015. In 2013 and 2015, ten traps were set on the reference site, and six traps were set on the restoration site. Six traps were set at both the reference site and restoration site in 2014. At each site, half of the traps had a small (3 cm) opening and half of the traps had a large (6 cm) opening. Traps were set approximately 10m apart, with alternating large and small openings. Traps were baited with ~30g of fresh Lake Trout eggs and deployed for 1.5 hours during October, a time of peak activity for Round Goby and Rusty Crayfish (Robinson 2014). All fish and crayfish species were identified, enumerated, and immediately released.

Invasive egg predators were also monitored through the use of GoPro HERO 4® cameras from 2013 through 2015. Each camera was mounted to a 1” steel pipe, welded to a quad-pod base of 3/8” steel rods (height = 60 cm; base = 0.45m^2) (Robinson 2014). Cameras were mounted to the center pipe and pointed down toward the substrate. The cameras were placed 10m apart and set to take a picture once every minute for 10 minutes. The area covered in each image was ~0.27m^2. In 2015, both baited and unbaited cameras were deployed to compare with varying methods used in 2013 and 2014. In 2013, cameras were baited with ~30g of fresh Lake Trout eggs, and unbaited cameras were deployed in 2014. In 2015, the unbaited cameras were deployed first. Once all images were taken, they were pulled up, baited with ~30g of Lake Trout eggs and immediately redeployed. Each fish and crayfish in each image was identified and counted using MS Paint. A minimum of 50% of an individual in the viewing area was required to be included in the count. The maximum number of each species per m^2 during the 10 minute period was used for analysis.

Interstitial Monitoring

Egg deposition and invasive egg predator data were collected using egg bags at both the restoration and reference sites before and after the limestone addition. Egg bags used in this study were similar to those described in Perkins and Krueger (1994) and Barton et al. (2011). Ten egg bags were deployed at each site 1-2m apart and were pulled and re-set every three weeks. The first set of egg bags was deployed in mid-September, and the last set was recovered mid-December in 2013 – 2015. Divers carefully removed egg bags from the substrate and closed each egg bag to prevent loss of eggs. All egg bags were kept in fresh lake water and processed within 24 hours. In the lab, each egg bag was emptied into a clear, gridded, glass tray on a light table to be sorted. Eggs were identified as either Lake Trout or Coregonid.   Number of eggs each year was calculated and used for analysis. Predators in each egg bag were identified, measured (mm), and weighed (g) from 2013 – 2015.

Egg funnels, similar to those used by Barton et al. 2011, were also used to assess natural egg deposition. Five funnels were deployed at each site and were pumped a minimum of twice a month from October through November 2013 – 2015. Funnel samples were kept in fresh lake water and processed within 24 hours using the same methods described above for the egg bags.

Artificial and Natural Egg Seeding

Egg nets and funnels at both the reference and degraded sites were seeded with both artificial and natural eggs once each year in mid-October from 2013 – 2015. Divers seeded the gear by opening vials of eggs centered on each egg bag, approximately 5cm above the substrate. The beads mimicked natural egg deposition as they settled into the substrate contained in each egg bag and funnel. Twenty artificial Lake Trout eggs (6mm diameter, plastic beads) and twenty eyed Lake Trout eggs were seeded in each egg bag; and twenty artificial Lake Trout eggs and 100 eyed Lake Trout eggs were seeded in each funnel. Both artificial and natural eggs were seeded to examine the relative contribution of environmental disturbance and predation on egg mortality.

Statistical Analyses

Substrate size was calculated by taking the product of the x, y, and z measurements from each rock to attain an idealized volume for each individual rock. Substrate size was assessed between the reference site, the degraded site pre-restoration, and the restored reef post-restoration with a nested analysis of variance (ANOVA) with site as a factor and rock layer nested within each site to examine potential differences between sites and the rock layers. The number of Dreissenid from the ten rocks in each layer were summed, and sum of the volumes of the ten rocks were used to calculate the Dreissenid density per cubic meter of gravel/rubble. Dreissenid abundance within the reference site was examined with a generalized linear model.

The number of Dreissenid was the response variable and the rock layer was the independent factor. A negative binomial error distribution was used to account for overdispersion. Two-sample t-tests were used to assess the similarities in interstitial depth, interstitial space, and maximum current velocity between the reference site and the restored reef post-restoration. A one-way ANOVA was used to examine similarities in slope among the reference site, degraded site pre-restoration, and restored site post-restoration site. A two-way ANOVA was also used to test differences between sites and years, plus their interaction in the predator density data collected by the baited and unbaited predator monitoring cameras, which were analyzed separately. Minnow trap data were analyzed with generalized linear models with the predator CPE (number of individuals per minnow trap 1.5 hours) as the response, and the year, site, and the year*site interaction as factors. The quasi-poisson error distribution was used in these models as overdispersion prevented other error distributions from being employed (Cameron and Trivedi 1990, Crawley 2007). Predators collected from eggbags were also analyzed using generalized linear models with the number of predators collected as the response, and year, site, and the year*site interaction as factors. This data was not overdispersed, and the Poisson error distribution was used (Cameron and Trivedi 1990, Crawley 2007). Generalized linear models with negative error distribution were used to examine seeded egg and bead returns, as well as naturally deposited eggs, through time and between sites in both egg bags and funnels (Cameron and Trivedi 1990). The number of seeded eggs or beads returned after seeding was the response, and year, site, and the year*site interaction were used as factors. All analyses were performed in R (3.1.0). An interaction between year and site was taken to be a result of the habitat restoration, and significant interactions were examined using the post-hoc multiple comparison function ‘testInteractions’ of the ‘phia’ package in R (3.1.0). Results were considered significant when $P \leq 0.05$.

RESULTS

Habitat Measurements

The post-restoration habitat was similar to the habitat at the reference site (Table 1). Interstitial depth (P=0.12), interstitial space (P=0.66), and maximum current velocity (P=0.28) did not differ between the reference and restoration sites. Slope differed among the pre-restoration ($24.2^o \pm 2.4$) , post-restoration ($62.4^o \pm 3.2$), and reference ($40.0^o \pm 3.7$) habitats (P<0.01). The pre-restoration rock sizes were much larger than the reference (P<0.001) and post-restoration rocks (P<0.001). The post-restoration and reference rocks did not differ in size (P=0.84) (Figure 4a). Due to the extremely large differences between the pre-restoration rock sizes compared to the reference site and post-restoration rock, the nested ANOVA was ran a second time with the pre-restoration rock removed. When the pre-restoration rock was not included in the analyses, there was a significant difference in rock size between the reference and post-restoration sites and layers (P = 0.001) (Figure 4b, Figure 5). Layer 1 at the reference site was the same as layers 1, 2, and 3 in the post-restoration habitat. All of the layers (1-5) in the post-restoration habitat were the same as layer 2 in the reference habitat. Layers 2 through 5 in the reference habitat were the same as layers 4 and 5 of the post-restoration habitat. At both the reference and post-restoration sites, rock size was smaller in the deeper layers of the reef.

Dreissenid abundance increased with layer depth (Figure 6). Dreissenid in layers 1, 2, and 3 were very low and similar to one another but different from layers 4 and 5, which had higher numbers of Dreissenid (Figure 6). Dreissenid had not colonized the restored site as of December 2015.

Table 1. Average characteristics of the reference, pre-restoration, and post-restoration habitats. Interstitial depth, mean (± SE); interstitial space, mean (± SE); slope, mean (± SE); and maximum current velocity, mean (± SE).

Site	Interstitial depth (cm)	Interstitial space (%)	Slope (º)	Maximum current velocity (m/s)
Reference	71.67 (14.53)	44.43 (1.85)	40.00 (3.70)	2.18 (0.22)
Pre-restoration	0	0	24.20 (2.40)	NA
Post-restoration	109.6 (15.54)	46.67 (3.33)	62.40 (3.24)	4.05 (1.26)

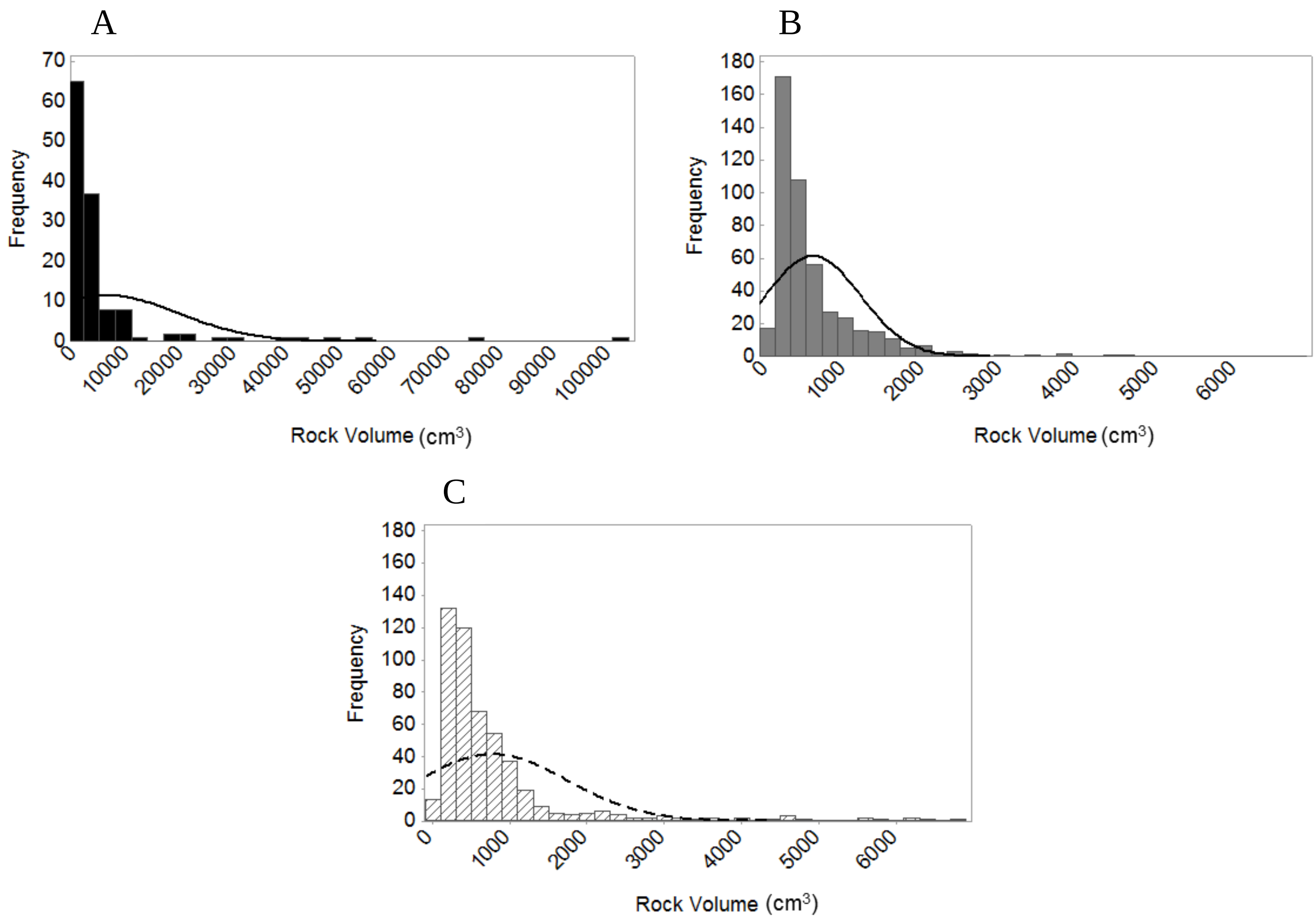

Figure 4. Occurrence frequency of the pre-restoration (A), reference (B), and post-restoration (C) individual rock volumes. Individual rock volumes were calculated by taking the product of the x, y, and z axes measurements from each rock.

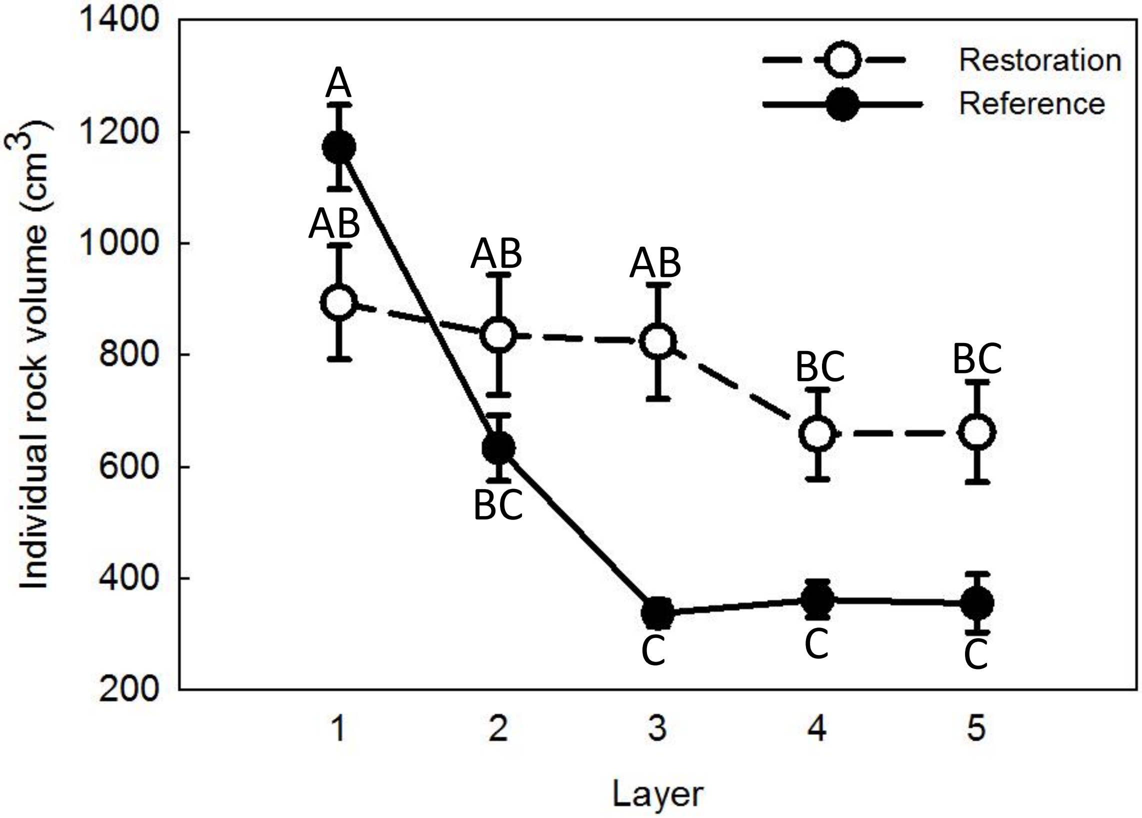

Figure 5. Average individual rock volume (cm^3) at the reference and post-restoration habitats collected in layers 1 (upper layer) through 5 (deepest layer) within the interstitial habitat of the reef. Points that share a letter are statistically equal (α=0.05).

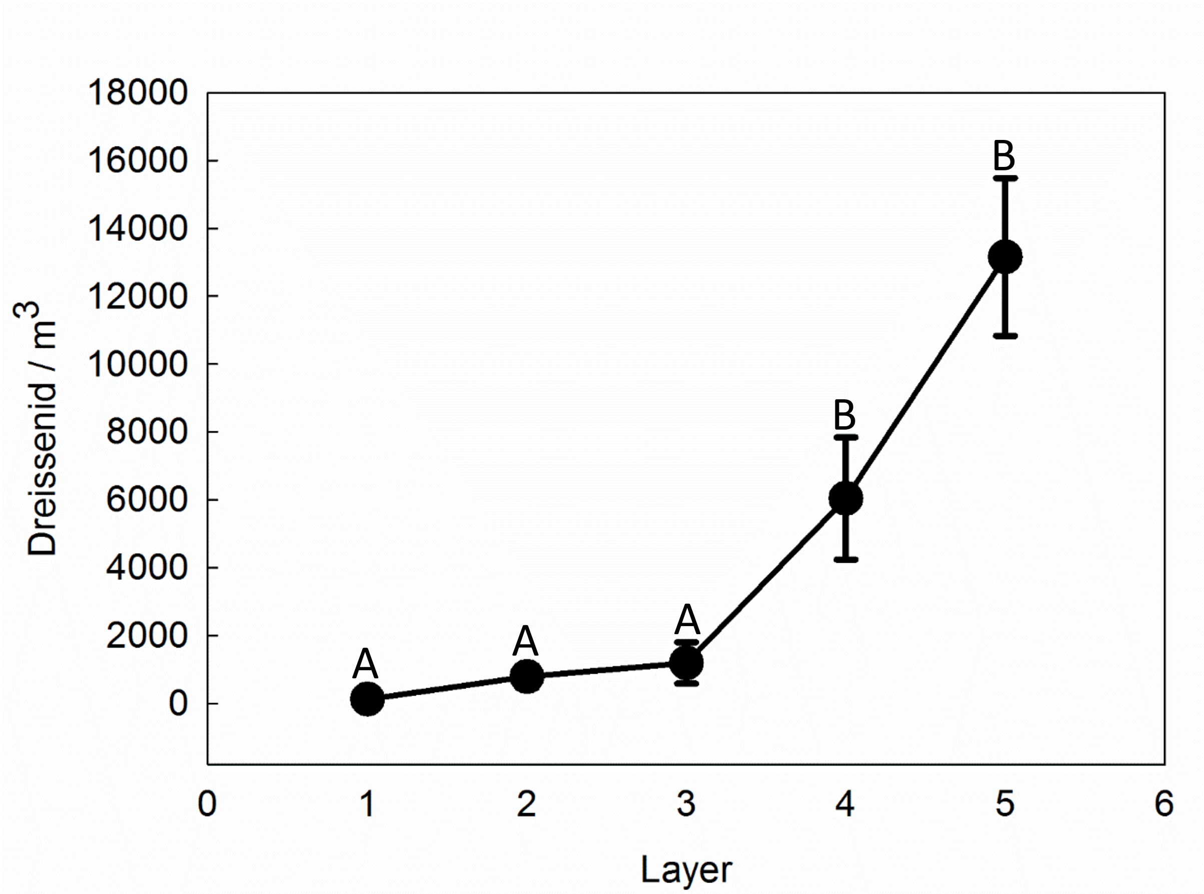

Figure 6. Mean (± SE) Dreissenid density in layers 1 (upper layer) through 5 (deepest layer) within the interstitial habitat of the reef (average density ± SE). Points that share a letter are statistically equal.

Epibenthic Predator Monitoring

Round Goby densities sampled by the unbaited predator monitoring cameras were lower in October 2015 than in October 2014 (P = 0.01), but there was no difference between sites (P = 0.07), nor an interaction between site and year (P = 0.09) (Figure 7a, Table 2). Data limitations prevented any analysis of Rusty Crayfish densities in the unbaited cameras, as no crayfish were captured at either site in 2015 (Figure 7b). Round Goby and Rusty Crayfish densities in 2013 and 2015 did not differ in baited cameras (Round Goby: Year: P=0.91, Site: P=0.77, Year*Site:

P=0.66; Rusty Crayfish: Year: *P*=0.87, Site: *P*=0.64, Year*Site: *P*=0.64) (Figures 7c & 7d) or minnow trap samples (Figures 8a & 8b).

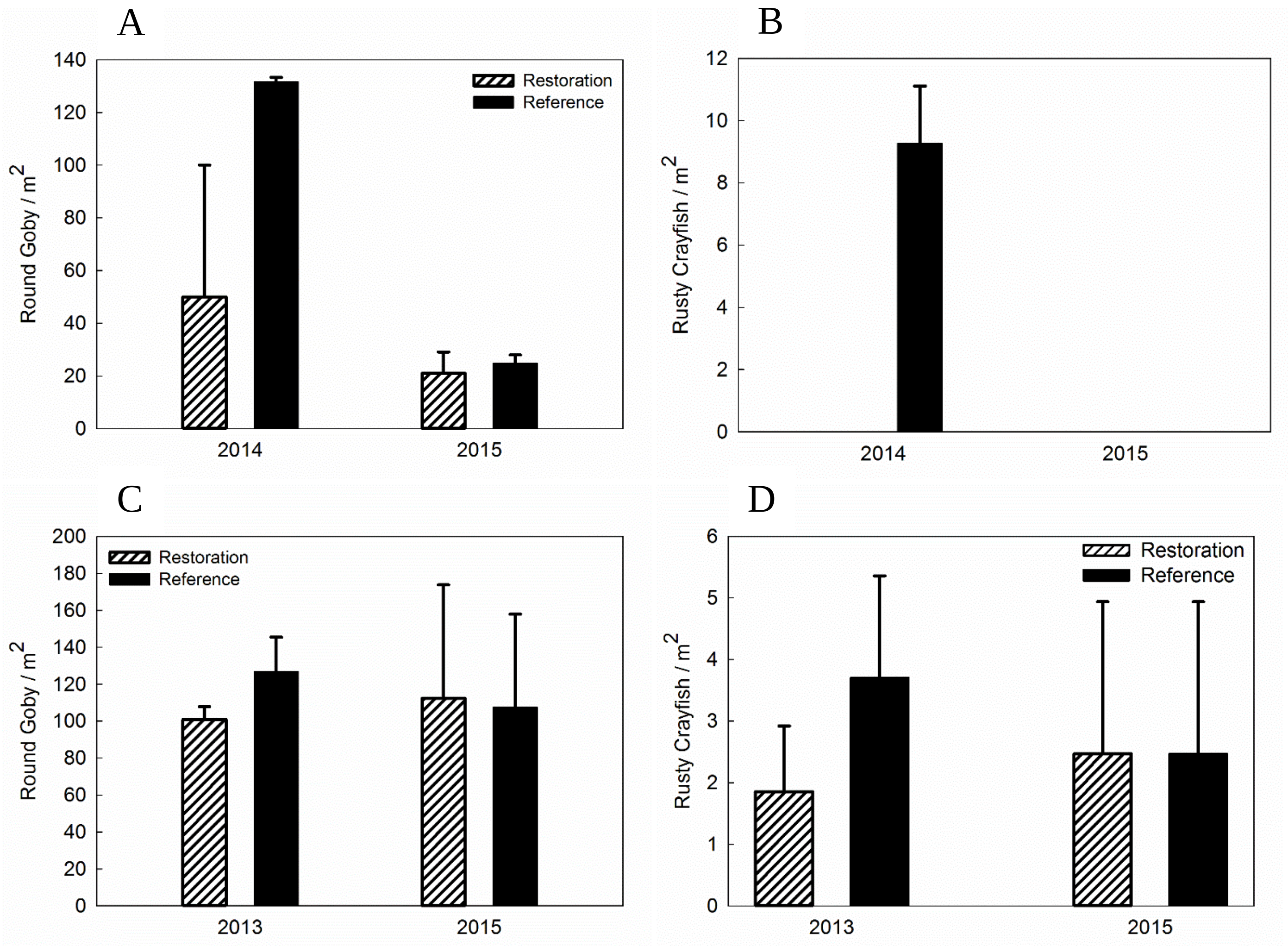

Figure 7. A) The maximum density of Round Goby (maximum number of individuals per m^2; max density ± SE) observed in the unbaited predator monitoring cameras in October of 2014 and 2015 at the reference and restoration sites. B) The maximum density of Rusty Crayfish (maximum number of individuals per m^2; max density ± SE) observed in the unbaited predator monitoring cameras in October of 2014 and 2015 at the reference and restoration sites. C) The maximum density of Round Goby (maximum number of individuals per m^2; max density ± SE) observed in the baited predator monitoring cameras in October of 2013 and 2015 at the reference and restoration sites. D) The maximum density of Rusty Crayfish (maximum number of individuals per m^2; max density ± SE) observed in the baited predator monitoring cameras in October of 2013 and 2015 at the reference and restoration sites.

Table 2. Summary of the invasive egg predator results

Species	Sampling method	Gear type	Yearly differences	Site differences	Interaction
Round Goby	Epibenthic	Unbaited cameras	Decreased from 2014-2015	None	None
Round Goby	Epibenthic	Baited cameras	None	None	None
Rusty Crayfish	Epibenthic	Unbaited cameras	N/A	N/A	N/A
Rusty Crayfish	Epibenthic	Baited cameras	None	None	None
Round Goby	Epibenthic	Minnow traps	None	None	None
Rusty Crayfish	Epibenthic	Minnow traps	None	None	None
Round Goby	Interstitial	Egg bags	Increased from 2013-2014, decreased from 2014-2015	Restoration site higher than reference site	Yes, densities were lower at restoration site between 2013 & 2015 and 2014 & 2015
Rusty Crayfish	Interstitial	Egg bags	None	None	None

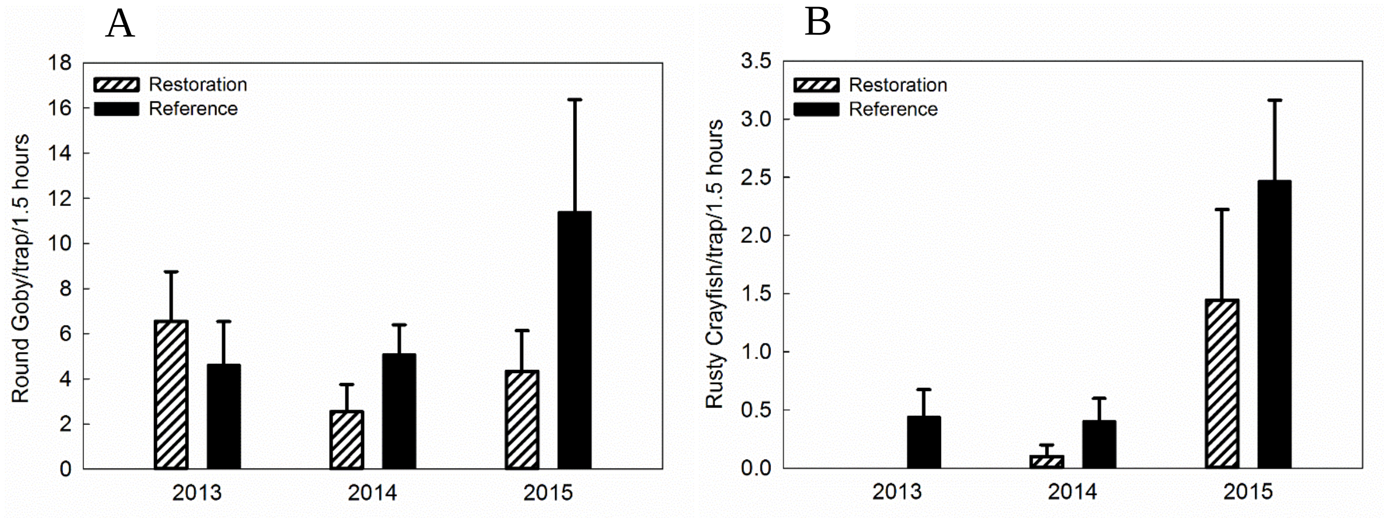

Figure 8. A) Round Goby catch-per-unit-effort (number of individuals per minnow trap 1.5 hours; CPUE ± SE) in October of 2013, 2014, and 2015 at the reference and restoration sites. B) Rusty Crayfish catch-per-unit-effort (number of individuals per minnow trap 1.5 hours; CPUE ± SE) in October of 2013, 2014, and 2015 at the reference and restoration sites.

Interstitial Predator Monitoring

There were higher numbers of Round Goby collected from the egg bags at the restoration site when compared to the reference site from 2013-2015 (P = 0.02) (Figure 9a, Table 2). Differences were also found between the years 2013 – 2014 (P = 0.02) and 2014 – 2015 (P < 0.001) with 2014 having higher numbers of Round Goby. There were significant interactions between the sites and years 2013 – 2015 (P = 0.004) and 2014 – 2015 (P < 0.001), as there was fewer Roundy Goby at the restoration site in 2015 than the previous years, and when compared to the reference site. There were no significant differences in Rusty Crayfish numbers between sites or years (Year: P=0.07, Site: P=0.06, Year*Site: P=0.41) (Figure 9b).

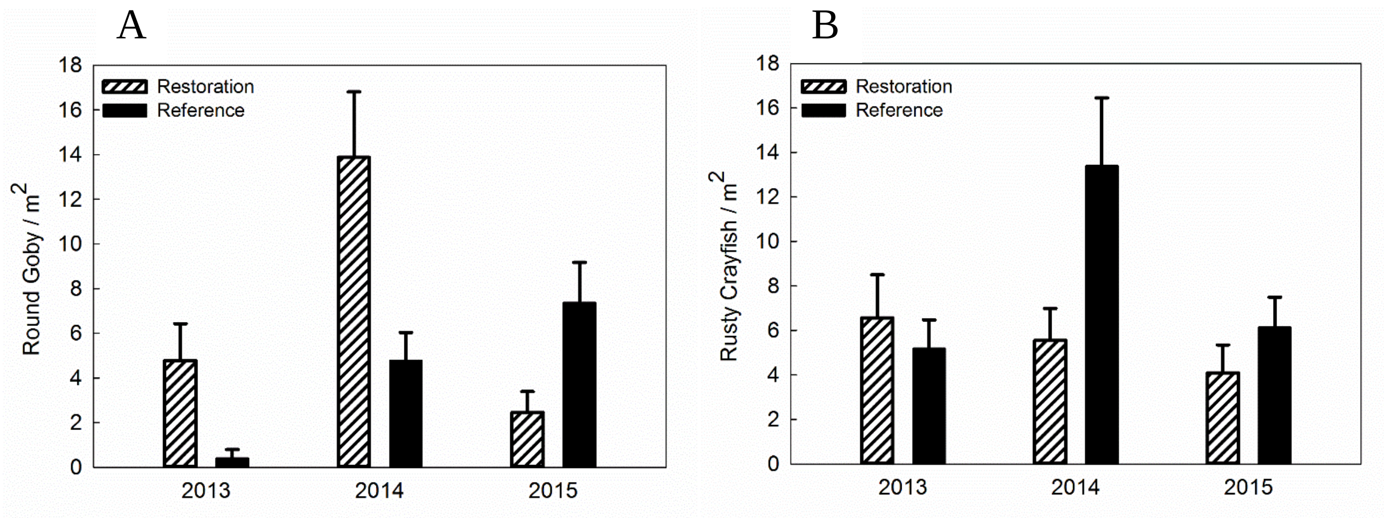

Figure 9. A) Interstitial Round Goby density (number of individuals per egg bag area; density ± SE) from October through December 2013 – 2015 at the reference and restoration sites. B) Interstitial Rusty Crayfish density (number of individuals per egg bag area; density ± SE) from October through December 2013 – 2015 at the reference and restoration sites.

Artificial and Natural Egg Seeding

More seeded eggs remained in the egg bags at the restoration site than at the reference site in 2015 (P<0.01). There were higher seeded egg returns in the egg bags at the restoration site in the post-restoration period (2015) compared to the pre-restoration site (Figure 10a, Table 3). There was also differences between 2013 – 2015 (P = 0.01) and 2014 – 2015 (P < 0.001), with higher returns in 2015; yet no differences between sites (P = 0.68). There were significant differences in seeded bead returns from 2013 – 2015 (P < 0.01) and 2014 – 2015 (P < 0.001) (Figure 10b). However, no interactions (P=0.19) and no differences between sites (P=0.63) were observed with the seeded beads in the egg bags (Figure 10b). Average wind velocity steadily decreased from 2013 to 2015 (Figure 10b).

There was no difference between seeded egg returns in in the funnels among all years (*P*=0.07) and between the reference and restoration site (*P*=0.50). The seeded eggs returns in the funnels were higher in 2015 (post-restoration) at the restoration site than in 2014 and when compared to the reference site, resulting in an interaction between 2014 – 2015 (*P*=0.02) (Figure 11a, Table 3). Seeded beads in funnels also showed higher returns in 2015 when compared to 2014 (*P* = 0.03). An interaction was also seen with the bead returns, as there was higher returns at the restoration site in 2015 than 2014 and when compared to the reference site (P = 0.02) (Figure 11b).

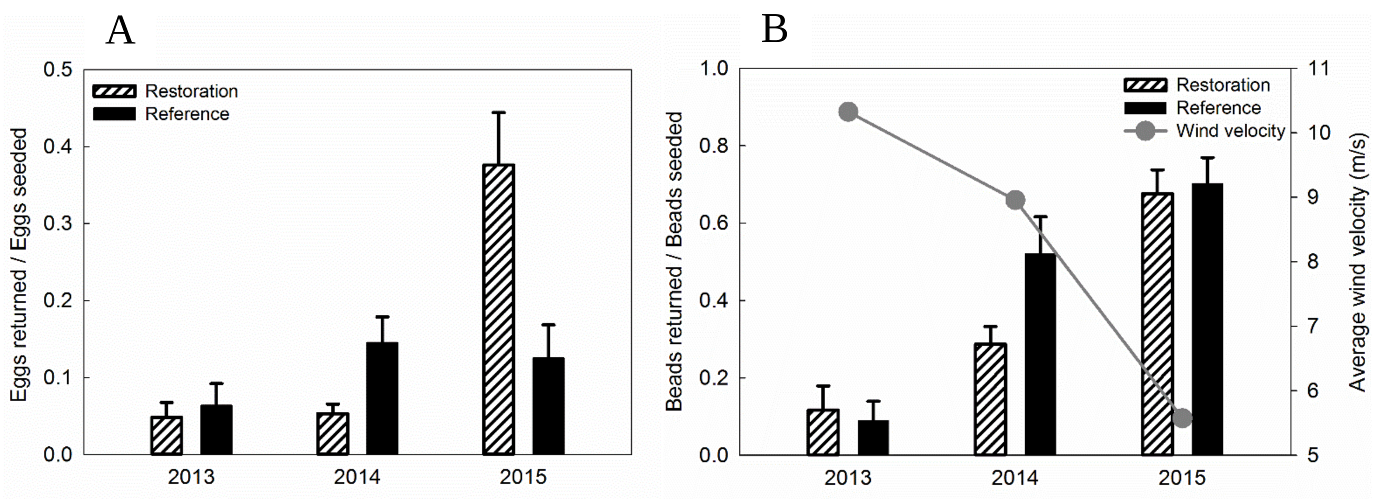

Figure 10. A) Seeded Lake Trout egg return ratio (eggs returned / eggs seeded ± SE) from egg bags in 2013, 2014, and 2015 at the reference and restoration sites. B) Seeded artificial egg return ratio (eggs returned / eggs seeded ± SE) from egg bags in 2013, 2014, and 2015 at the reference and restoration sites. Average wind velocity (m/s ± SE) for each year is represented by the solid line.

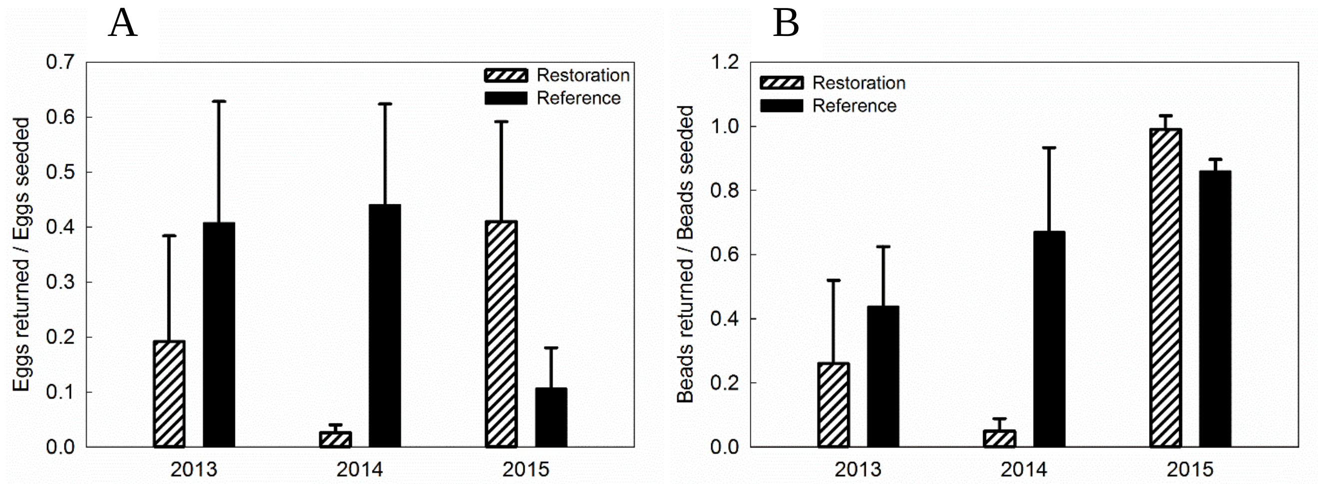

Figure 11. A) Seeded Lake Trout egg return ratio (eggs returned / eggs seeded ± SE) from funnels in 2013, 2014, and 2015 at the reference and restoration sites. B) Seeded artificial egg return ratio (eggs returned / eggs seeded ± SE) from funnels in 2013, 2014, and 2015 at the reference and restoration sites.

Table 3. Summary of the seeding experiment results.

Gear type	Seeding method	Yearly differences	Site differences	Interaction
Egg bags	Seeded eggs	Increased from 2013-2015 and 2014- 2015	None	Yes, returns were higher at restoration site between 2013 & 2015 and 2014 & 2015
Egg bags	Seeded beads	Increased from 2013-2015 and 2014- 2015	None	None
Funnels	Seeded eggs	None	None	Yes, between 2014 & 2015
Funnels	Seeded beads	Increased from 2014-2015	None	Yes, between 2014 & 2015

Natural Egg Deposition

There were no differences in naturally deposited Lake Trout eggs in the egg bags between years (*P*=0.19), sites (*P*=0.25), and year*site (*P*=0.31) (Figure 12a). Similarly, there were no differences between years (*P*=0.28) and no year*site interaction (*P*=0.08) in naturally deposited eggs in the funnels, however there was a difference between sites (*P*=0.02), with the reference site having higher returns from 2013-2015 (Figure 12b). Additional years of post-restoration data are needed to determine the effectiveness of the new habitat. Furthermore, the Lake Trout egg deposition in the funnels at the restoration site in 2015 was the second highest over the past three years, even compared to the deposition at the high-quality reference site. No coregonid egg deposition occurred on the restored reef in 2015, and low egg densities were seen in both egg bags and funnels at the reference site.

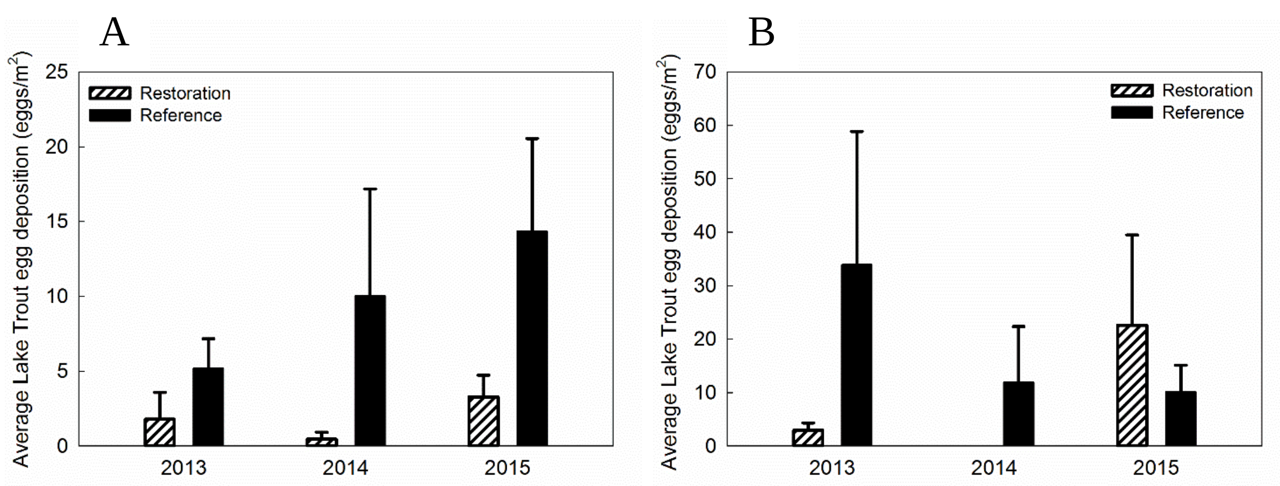

Figure 12. A) Natural Lake Trout egg deposition (eggs / m^2 ± SE) from egg bags in 2013, 2014, and 2015 at the reference and restoration sites. B) Natural Lake Trout egg deposition (eggs / m^2 ± SE) from funnels in 2013, 2014, and 2015 at the reference and restoration sites.

DISCUSSION

The primary goals of this reef restoration were to increase the interstitial depth and the quality of interstitial spaces to determine if that would result in increased egg retention and survival, and if habitat restoration would reduce epibenthic and interstitial invasive egg predator densities. The restoration was successful based upon measures of the microhabitat of the restored reef being similar to the reference reef, with some minor differences. Interstitial depth, percent interstitial space, and maximum current velocity was similar between the restored and reference reefs. The factors that were different (e.g. slope) may be even more favorable to reef quality. The increased interstitial depth and the high degree of similarity between the restored reef habitat characteristics and the high-quality reference site habitat resulted in improved seeded egg and bead retention when comparing pre and post restoration with trends at the reference site. Our preliminary egg deposition and invasive egg predator results are optimistic, but additional years of monitoring will be required to determine the success of this spawning reef restoration.

Through the assessment of the reference reef microhabitat, we gained additional insight on the characteristics of our reference site that attribute to the relatively high egg deposition and retention found here, compared to other nearshore spawning reefs in northern Lake Michigan. Two defining characteristics of our reference site are the (1) relatively deep interstitial depth and (2) the stratification of rock sizes within the reef (Table 1, Figure 5). We postulated that the combination of these two characteristics would result in a higher amount of egg protection from both environmental disturbance and invasive egg predators. Assuming there is no limitation to the depth eggs can settle within the reef, we contend that the smaller rock in the deeper layers of the reference site, in combination with the increasing habitat complexity with interstitial depth, limit Round Goby and Rusty Crayfish from gaining access to eggs, thereby increasing egg survival at this site. The steep slope and relatively high maximum current velocity at our reference site result in the reef being extremely clean and free of silt and debris. With our reference site also being located in relatively shallow (~3m) water, wave action causes the top two layers of gravel/rubble to be free of *Dreissenids*. We believe the "self-cleaning" qualities of our reference reef to be critical in not only attracting spawning fish, but also maintaining high interstitial water quality. Our reference reef maintains a delicate balance between having enough wave action and current velocity to keep incubating eggs well oxygenated, while the deep interstitial depth and relatively small interstitial spaces prevent deposited eggs from being displaced, except under the most extreme weather conditions. We found increased numbers of *Dreissenid* in the fourth and fifth (lower) layers compared to the first, second, and third (upper) layers. This indicates lower environmental disturbance, and limited access by interstitial predators, in the lower layers of the reference reef. Biga et al. (1998) found that rock size, and consequently the size of interstitial spaces, prevented Mottled Sculpin (*Cottus bairdi*) from accessing trout eggs in smaller substrates. Furthermore, they found as the size of rock increases, the size and range of sculpin movement through interstitial spaces also increases. Similar results were also found in a different study with Virile Crayfish (*Orconectes virilis*) (Savino and Miller 1991). Based upon increased *Dreissenid* abundance and smaller rock size in the deeper interstices, in combination with increasing habitat complexity with interstitial depth, we believe Round Goby and Rusty Crayfish can penetrate up to the third layer of rock within our reference site. With each layer of rock being approximately 10cm thick, we believe interstitial predators only have access to eggs within the upper 30cm of our reference site, meaning any eggs in the bottom 40cm of interstitial habitat could remain protected from predators. However, additional studies need to be conducted with Round Goby and Rusty Crayfish to examine the effects of varied substrate sizes, and substrate size stratification on interstitial predator movement and egg mortality.

We speculate that the restored reef will achieve a similar interstitial rock-size stratification to that of the reference site through time as wave energy and currents shift the reef and cause it to settle. This stratification appears to have already started at the time measurements were being taken, as rock size at the restored reef decreases with layer depth (Figure 5). We also anticipate Dreissenid mussel colonization on the restored reef will occur in a similar fashion as the reference site, however no colonization had occurred as of December 2015. Our results suggest that wave action and currents will prevent the restoration site from being colonized by Dreissenid mussels in the top two to three layers. Dreissenid colonization may occur in the deeper layers of the restored reef, but at relatively low densities, similar to the reference site. Because the interstitial depth, percent interstitial space, rock size, and maximum current velocity were the same between the reference site and post-restoration, we believe the restored reef will perform similar to the reference site in future years, and have a similar impact on predators and eggs

This spawning reef restoration was different from some of the other reef restoration projects throughout the Great Lakes, in that fish were spawning at the restoration site for multiple years prior to the habitat improvements, making this one of the few Lake Michigan spawning reef projects targeted at improving used, but sub-optimal, habitat. Lake Trout, Lake Whitefish, and Cisco were likely attracted to the natural slope and currents of degraded site (pre-restoration); however, adequate habitat for egg protection and incubation was lacking. The pre-restoration habitat was lacking interstitial depth and quality interstitial spaces, thus allowing predators, strong currents, and wave action to readily displace deposited eggs. The increased seeded egg and bead returns at the restoration site in 2015 indicate there is a higher level of protection from physical force, and lower predation post-restoration relative to pre-restoration and the reference site as a result of the habitat restoration. Differences in seeded bead retention can be explained by the decreasing average wind velocities from 2013 – 2015. Wind steadily decreases from 2013 to 2015, as seeded bead returns steadily increased. Additionally, water levels in Lake Michigan were at an all-time low in 2013, and increased to over 1m deeper in 2015. The relatively high wind velocity, combined with the low water levels in 2013 resulted in the lower bead returns that year. Epibenthic predator densities in the traps and baited cameras remained constant from 2013 – 2015, indicating the restoration did not influence epibenthic predators. Claramunt et al. 2005, Fitzsimons et al. 2006, and Claramunt et al. (*in preparation*) examined the influence and contribution of predation and environmental disturbance on egg retention and found interstitial predators can heavily influence egg survival. Interstitial Round Goby were impacted by the restoration, as interstitial Round Goby densities were lower post-restoration relative to pre-restoration and the reference site. The increased interstitial habitat complexity and smaller interstitial spaces may have prevented Round Goby movement into the deeper layers of the restored reef; however, more years of data and additional experimentation are needed to fully assess the impact of invasive egg predators on the restored reef. The lower predator densities at the restoration site in 2015 could have been a response to the instability and settling of the reef post-restoration. Additionally, the restoration site has an abundance of remaining structure from the large degrading pier directly adjacent to the site that attracts Smallmouth Bass (*Micropterus dolomieu*) and other piscivores. The higher numbers of centrarchids could be influencing the Round Goby and Rusty Crayfish numbers at this site, which could also contribute to its future success. Natural Lake Trout egg deposition at the restoration site was ~5 times higher post-restoration when compared to the two years before restoration. Although no Coregonid eggs were collected at the restored site, we anticipate they will utilize the reef in future years. The warmer water in 2015, combined with the exposed assessment gear as the newly restored reef was settling may have delayed and deterred Coregonid spawners (Barton et al. 2011).

Quantifying both physical habitat and biological responses with a long-term monitoring plan are crucial in determining the success of reef habitat improvements (McLean et al. 2015, Gannon, 1990). Our findings indicate that habitat objectives have been met and biological responses are promising, however long term monitoring will be required to determine whether increases in Lake Trout egg deposition and retention will be sustained. The ultimate goal of many reef restoration projects is to positively influence the population of target fish species that use the reef for spawning (Fitzsimons 1996, Dumont et al. 2011, Roseman et al. 2011, Houghton et al. 2013). However, there is currently little data showing how improved reef habitat influences fish population abundance (McLean et al. 2015). With Lake Trout, Cisco, and Lake Whitefish being relatively long-lived and late maturing, it may take at least a decade or more to observe an increase in abundance, especially in an area as large as Grand Traverse Bay or the entirety of Lake Michigan. Moreover, relatively large amounts of annual variation in environmental disturbance, water levels, invasive egg predators additionally complicate detecting a population-level response of the fish to the reef restoration, making long-term monitoring essential in any reef restoration (McLean et al. 2015).

Based upon our findings, interstitial microhabitat and interstitial predators should be a critical component in future spawning reef studies, as many important characteristics could be overlooked if only epibenthic habitat and biota are exclusively examined. Due to reef settling, water chemistry and sedimentation measurements were not able to be accurately measured, however in future years they will be monitored within the restored reef.

Minimizing egg loss from predation and environmental disturbance is critical for successful recruitment on nearshore spawning reefs (Eshenroder et al. 1995a, Marsden et al. 1995, Chotkowski and Marsden 1999, Claramunt et al. 2005). The increased interstitial depth and smaller interstitial spaces at the restored reef resulted in lower interstitial Round Goby abundance and a higher level of protection from environmental disturbance in 2015. We recommend focusing habitat improvement efforts in areas where spawning fish have already been spawning, but utilizing sub-optimal habitat. Restoration of used, sub-optimal areas will produce higher returns much faster than creating new habitat in an area that has historically not been used.

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STATUS AND HISTORY OF CISCO (*COREGONUS ARTEDI*) IN MICHIGAN INLAND WATERS

INTRODUCTION

Historically, Cisco *Coregonus artedi* were extremely abundant and widely distributed in the Great Lakes and in the inland waters of Michigan (Latta 1995, Scott and Crossman 1998). Cisco were not only valuable to commercial and recreational fishers, but they were ecologically valuable as they were a critical prey item for native piscivores (Stockwell et al. 2009, Derosier et al. 2015). However, habitat degradation, and invasive species have caused drastic declines of many Cisco populations in inland waters throughout the state (Colby and Brooke 1969, Hrabik et al. 1998, Derosier et al. 2015). The recovery of Cisco could stabilize and increase food web efficiency, provide a high-quality prey item for piscivores, and promote the recovery of other native species.

Cisco are an elongate, silvery fish with a blue-green to light green back, having an adult length commonly ranging between 254 – 381mm (Becker 1983, Derosier 2007). They have a pointed snout and a long lower jaw that extends slightly beyond the upper jaw (Becker 1983, Derosier 2007). The number of gill rakers ranges from 44 to 52 (Becker 1983, Derosier 2007). Multiple taxonomic schemes and variations are commonly found from one waterbody to another (Figure 1). Twenty-two subspecies were recognized by (Hubbs and Lagler 1964), 13 of which were reported in the inland lakes of Michigan. These "subspecies" likely represent variation among what are now considered morphotypes (Turgeon and Bernatchez 2001).

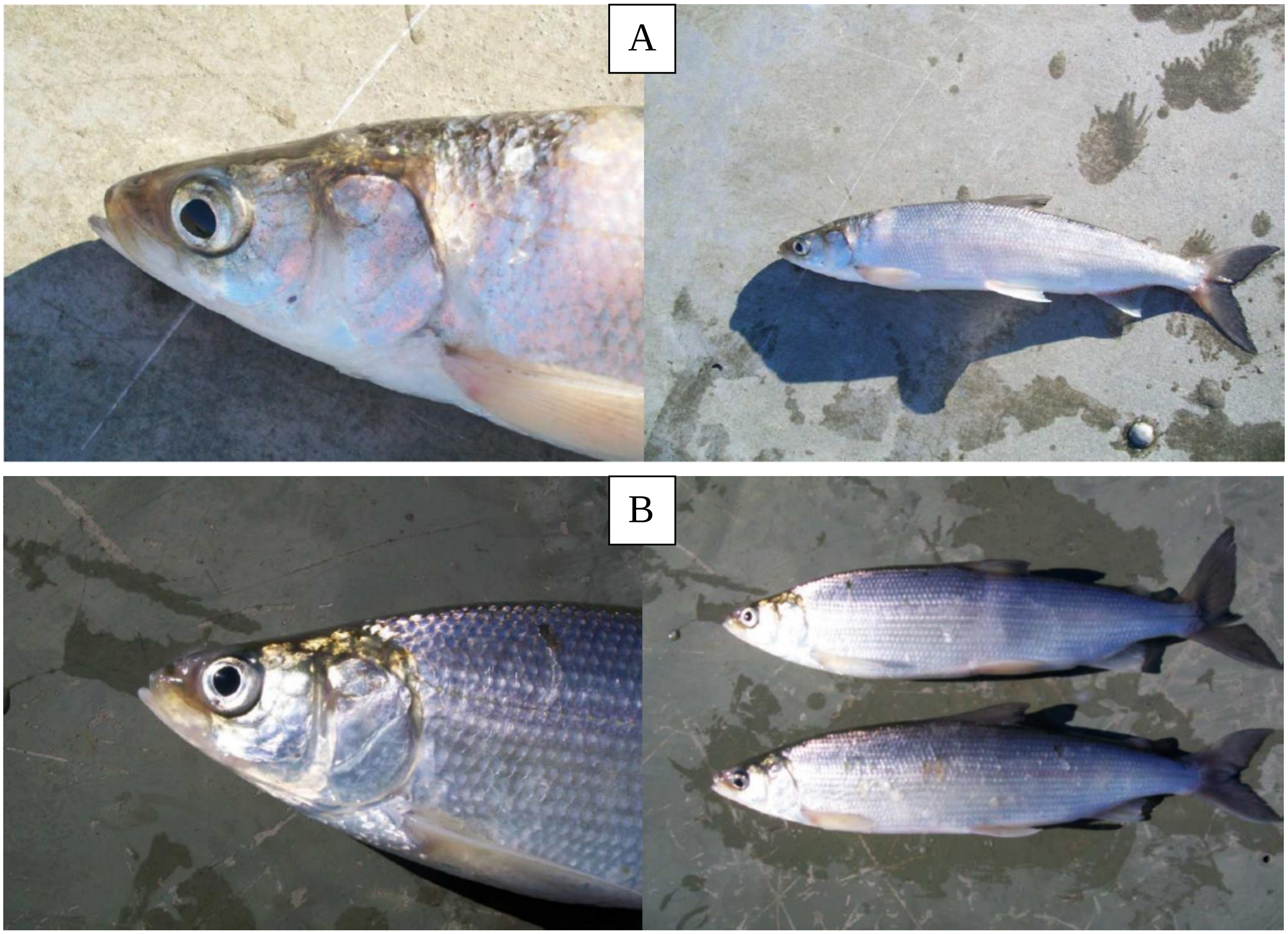


Figure 13. A) Cisco from Cedar Lake, Barry County, Mi. B) Cisco from Ziegunfuss Lake, Kent County, Mi. *Photo credit: Scott Hanshue, MDNR.*

Cisco are a cold-water fish that require temperatures less than 18ºC, and a minimum dissolved oxygen of 3-4mg/L (McLain & Magnuson 1988, Rook et al. 2013). Because this species is extremely sensitive to thermal and dissolved oxygen conditions, Cisco are excellent indicators of habitat degradation and environmental change (Latta 1995, Sharma et al. 2011). Other habitat features seem to be more important than lake size, as Cisco inhabit a multitude of

Michigan lakes varying in size from 0.02 – 76 km^2 with a median size of 0.83 km^2 (Latta 1995). However, as inland lake water temperatures increase in late summer, hypolimnetic oxygen levels can become dangerously low resulting in decreased growth rates and survival of Cisco (Aku et al. 1997, Aku et al. 1997, Becker 1983). Young of the year (YOY) Cisco are more tolerant than adult Cisco with respect increased temperature and dissolved oxygen (Edsall and Colby 1970, Sharma et al. 2011). The upper lethal temperature for adult Cisco is 20^{o}C, and 26 ^{o}C for young-of-year Cisco (Edsall and Colby 1970, Ebener et al. 2008). Low dissolved oxygen levels may push Cisco into lethal temperature waters (Becker 1983).

Spawning begins in late-November, and peak spawning occurs when water temperatures drop below 4^{o}C through December. In inland lakes, spawning takes place in 1-3m of water, and eggs are broadcast over sandy or gravel substrates. Cisco are mature at 3-4 years, and adults consume primarily zooplankton, but will also consume mollusks, insect larvae, small fish, and plant matter (Latta 1995, Ebener et al. 2008, Gamble et al. 2011a, Stockwell et al. 2014). Large variations in year class strength are common in Cisco populations, and very strong Cisco year classes have been produced from small parental stocks (Stockwell et al. 2009, Rook et al. 2013, Ebener et al. 2008).

Cisco are classified as a state threatened species in Michigan yet many inland waters have not been evaluated. One of the threats to Cisco, outlined in the Inland Cisco Lakes Wildlife Action Plan, was the lack of information regarding populations in Michigan (Derosier et al. 2015). Latta (1995) classified 153 inland Cisco lakes in Michigan, and evaluated their status. We have expanded on the work of Latta (1995) to update the state-wide status of Cisco in the inland waters of Michigan with the goal of narrowing management objectives and prioritizing conservation efforts across the state.

The objectives of this paper are to review the history and describe the current status of Cisco in Michigan inland water bodies and outline the current impediments and threats to conservation and restoration of Cisco in Michigan. We used data from Latta (1995) to examine the ecoregional distribution, trends, and status of Cisco lakes in Michigan. Additionally, we present goals and management objectives to protect and rehabilitate the remaining Cisco populations in Michigan inland waters.

METHODS

Locations and coordinates of each Cisco lake were identified using Google Earth and data from Latta 1995. The 29 additional Cisco lakes were added based upon updated MDNR catch data (*Scott Hanshue MDNR, personal communication, unpublished MDNR* data). ArcMap 10.2 was used to overlay the Cisco lake coordinates with the U.S. Environmental Protection Agency Level III and Level IV ecoregions of Michigan (EPA 2012)(Table 1). Level III and Level IV ecoregions were used as a course scale (Level III) and finer scale (Level IV) description of the Cisco lakes in Michigan. The Spatial Join function was used to determine the number of Cisco lakes in each ecoregion.

Table 4. EPA Level III and Level IV ecoregion codes and corresponding names.

Level III ecoregion code	Level III ecoregion	Level IV ecoregion code	Level IV ecoregion
50	Northern Lakes and Forests	50aa	Menominee-Drummond Lakeshore
50	Northern Lakes and Forests	50ab	Cheboygan Lake Plain
50	Northern Lakes and Forests	50ac	Onaway Moraines
50	Northern Lakes and Forests	50ae	Mio Plateau
50	Northern Lakes and Forests	50af	Cadillac Hummocky Moraines
50	Northern Lakes and Forests	50ag	Newaygo Barrens
50	Northern Lakes and Forests	50d	Superior Mineral Ranges
50	Northern Lakes and Forests	50i	Northern Wisconsin Highlands Lakes Country
50	Northern Lakes and Forests	50j	Brule and Paint River Drumlins
50	Northern Lakes and Forests	50k	Wisconsin/Michigan Pine Barrens
50	Northern Lakes and Forests	50u	Keweenaw-Baraga Moraines
50	Northern Lakes and Forests	50v	Winegar Dead Ice Moraine
50	Northern Lakes and Forests	50w	Michigamme Highland
50	Northern Lakes and Forests	50x	Grand Marais Lakeshore
51	North Central Hardwood Forests	51m	Manistee-Leelanau Shore
51	North Central Hardwood Forests	51n	Platte River Outwash
55	Eastern Corn Belt Plains	55a	Clayey High Lime Till Plains
56	Southern Michigan/Northern Indiana Drift Plains	56b	Battle Creek/Elkhart Outwash Plain
56	Southern Michigan/Northern Indiana Drift Plains	56d	Michigan Lake Plain
56	Southern Michigan/Northern Indiana Drift Plains	56g	Lansing Loamy Plain
56	Southern Michigan/Northern Indiana Drift Plains	56h	Interlobate Dead Ice Moraines

The status and abiotic characteristic data from Latta (1995) were used to explore the relationship between the measured abiotic characteristics and the status assigned to each lake. The status assigned to each lake was subjective, but based on the catch-per-unit effort of at least two samples from each lake (Latta 1995). Nonmetric multidimensional scaling (NMDS) was used to condense the abiotic variables into two dimensions. These dimensions were used to assess whether the status classifications from Latta (1995) represented discrete groups in the ordination space. Ninety out of the 182 lakes were used in the NMDS analysis, as many lakes lacked environmental variables and status. Differences in lake area (km^2) in relation status were examined with a Kruskal-Wallis test, as data did not meet the assumptions for a one-way analysis of variance (ANOVA) and were unable to be transformed to meet assumptions. Multiple comparisons were conducted using the Kruskal-Wallis post-hoc test proposed by Conover and Iman (1979). All analyses were performed in R (3.1.0) and results were considered significant when $P<0.05$.

RESULTS AND DISCUSSION

Latta (1995) identified 153 inland lakes in Michigan where Cisco were present. We have identified 29 additional inland waterbodies, for a total of 182 inland waterbodies that could potentially contain Cisco (Appendix 1, Figure 2). The 29 additional inland waterbodies (Figure 3) were included based on updated catch records (*unpublished MDNR data*), however no abiotic data was available to further examine these lakes. Additional sampling will also be required to determine the status of the Cisco residing in these lakes.

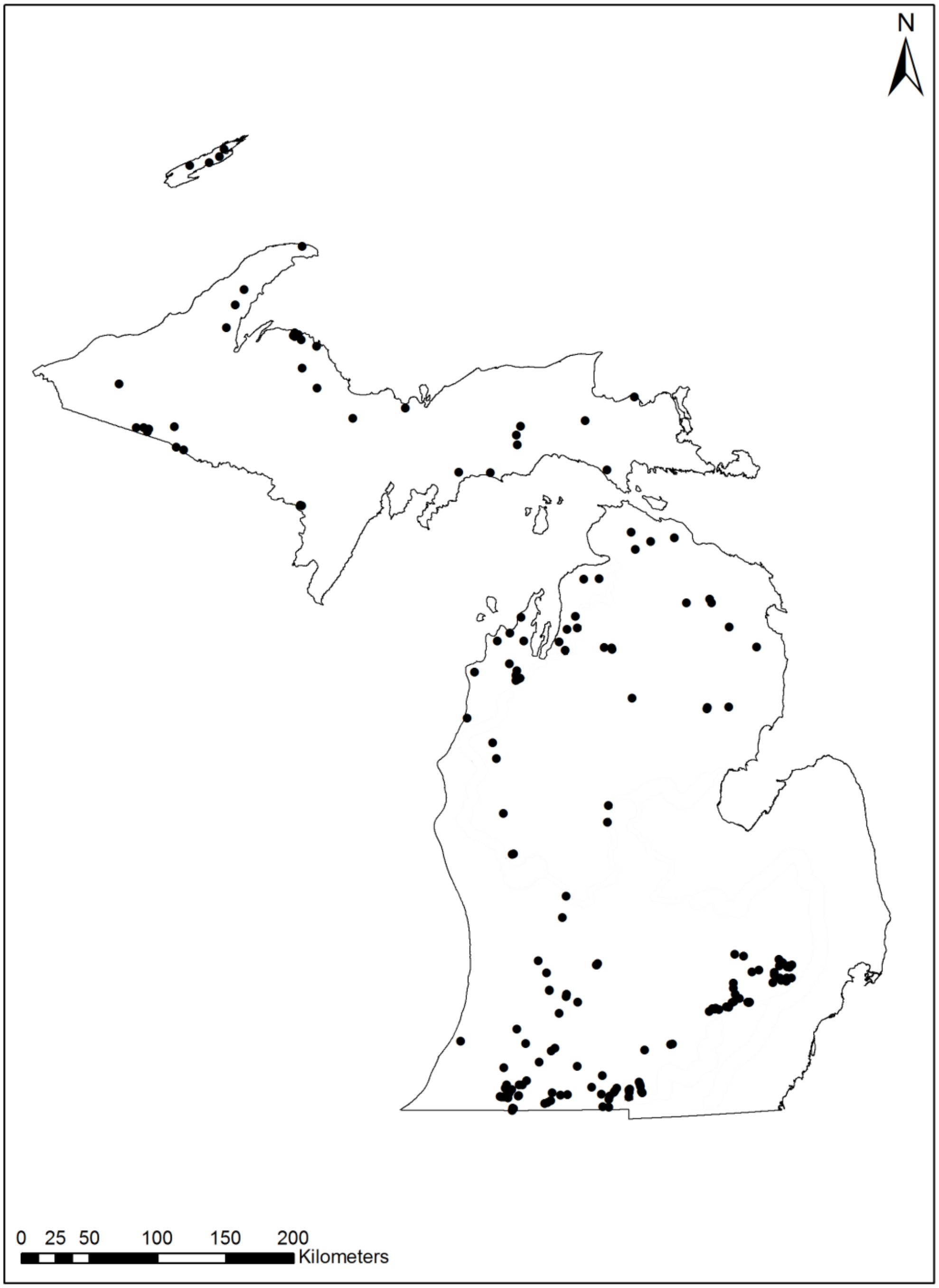
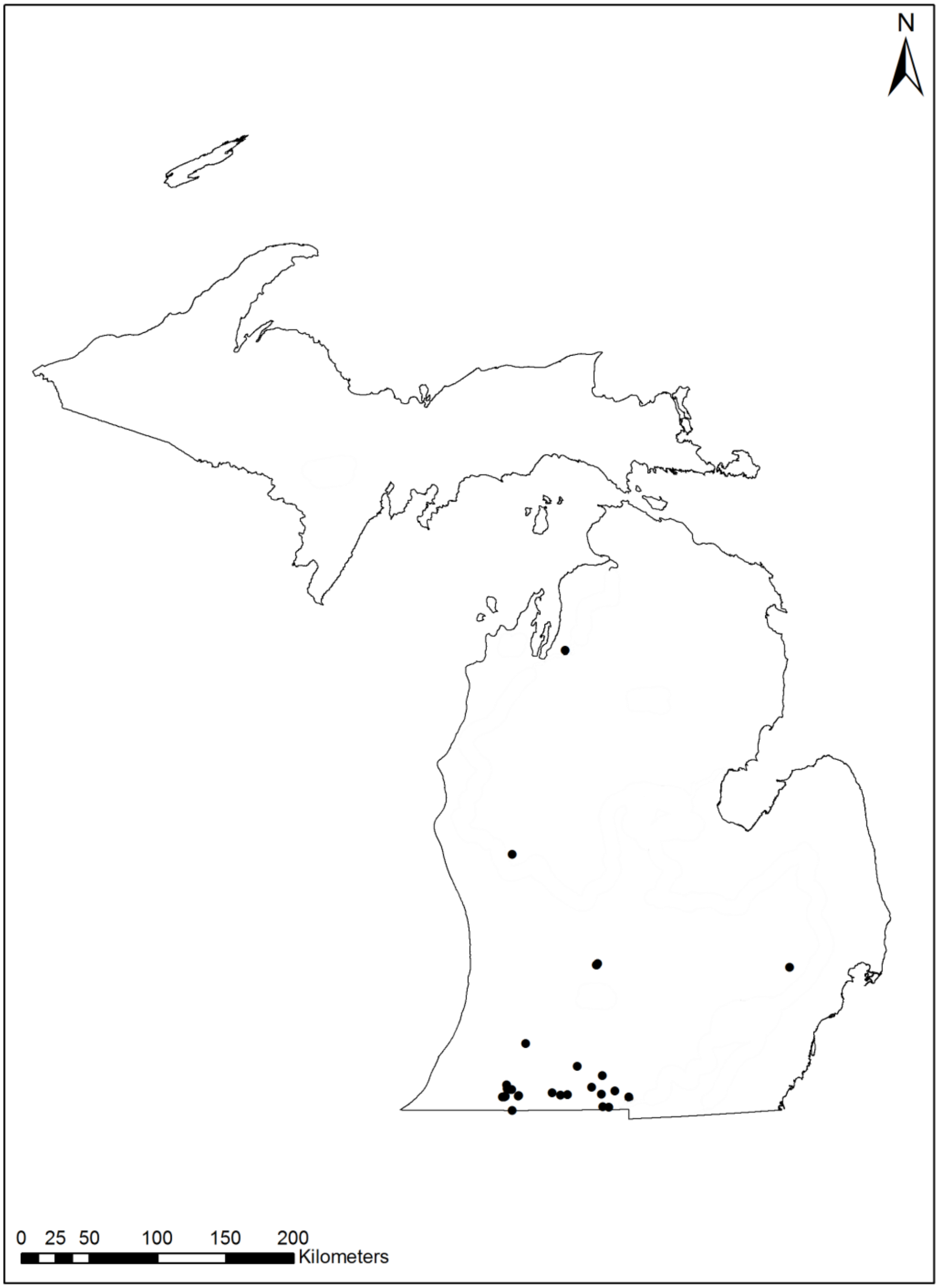


Figure 14. Locations of the 182 Cisco lakes in Michigan from Latta (1995) and MDNR data.



1057 Figure 15. Map of the 29 additional Cisco lakes from MDNR catch records.

A more thorough investigation will be required to accurately determine the current status of the lakes, as there is very little catch and assessment data from the inland waters of Michigan. Updates from Latta (1995) include more recent captures; however, the status for each lake has not changed from those assigned by Latta (1995) (Appendix 2). Approximately 44% of the 182 lakes are of unknown status or lack the sufficient amount of data to ascertain the status of the populations. Currently, 80 of the 182 inland lakes in Michigan are classified as stable; however, even the most recently sampled lakes have not been assessed in the past decade (Appendix 2). Additionally, some of the lakes classified as stable have not been assessed since the 1960's. The assessment of these lakes is critical, as extirpation of Cisco in inland lakes in adjacent states, along the same latitudes, have been projected to occur within this century (Sharma et al. 2011, Jacobson et al. 2012, Jacobson et al. 2013, Fang et al. 2012).

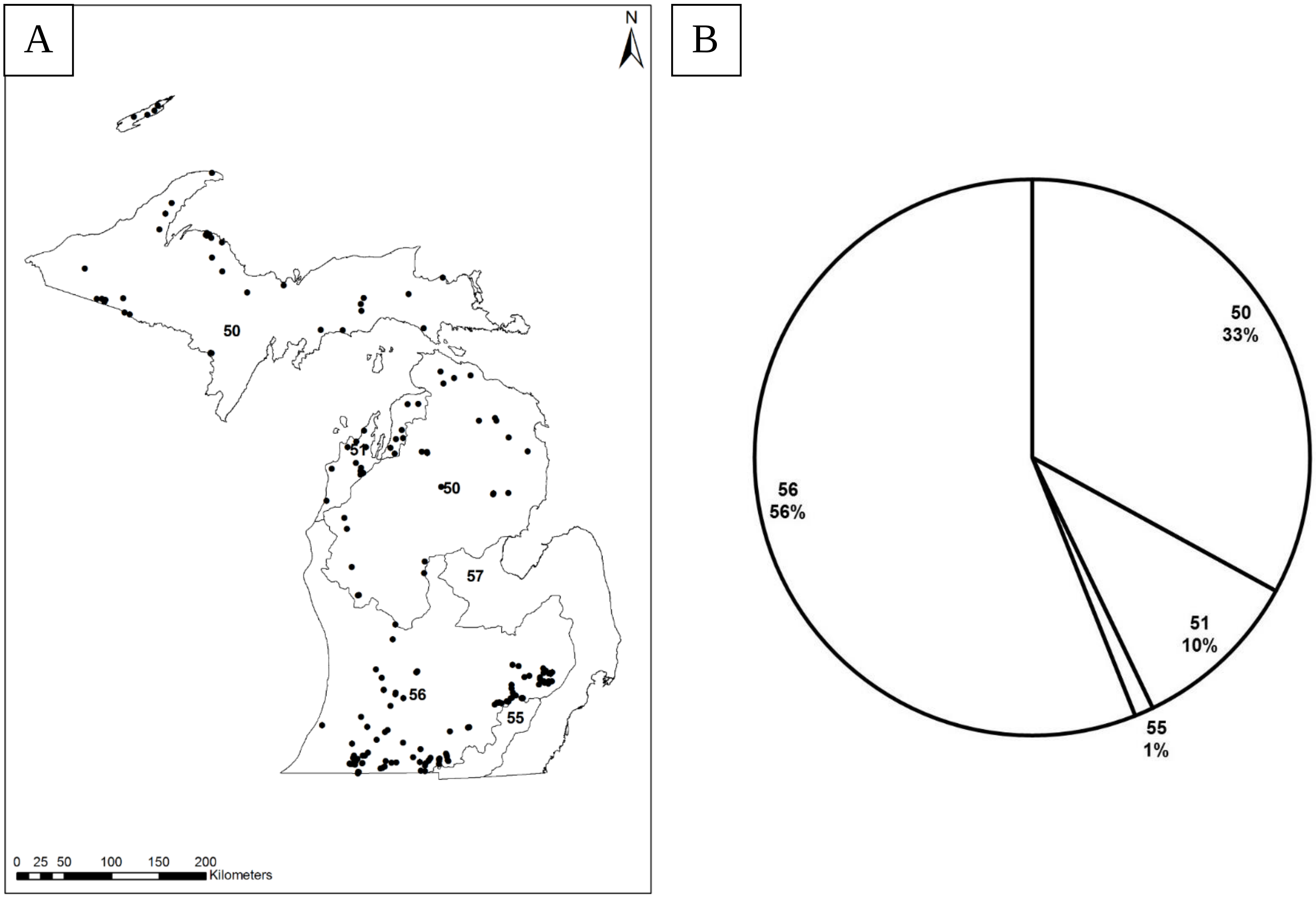


Figure 16. A) Map of the Michigan Cisco lake distribution within EPA Level III ecoregion boundaries. Each ecoregion is labeled with the Level III code (see Table 1). B) Michigan Cisco lake distribution (%) within the EPA Level III ecoregion boundaries.

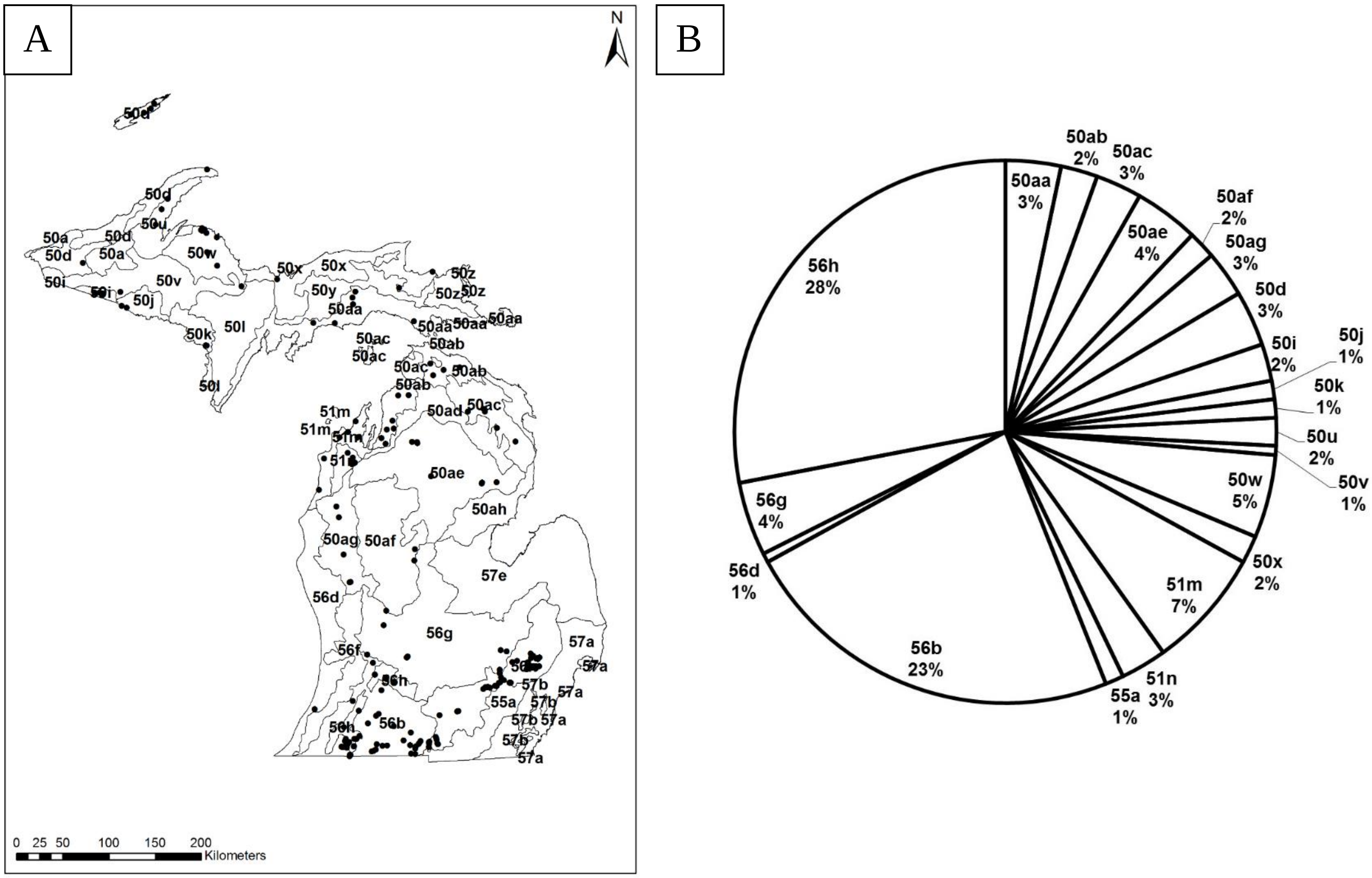


Figure 17. A) Map of the Michigan Cisco lake distribution within EPA Level IV ecoregion boundaries. Each ecoregion is labeled with the Level IV code (see Table 1). B) Michigan Cisco lake distribution (%) within the EPA Level IV ecoregion boundaries.

The primary distribution of Cisco lakes in Michigan lie in the band of kettle lakes in the Southern Michigan / Northern Indiana Drift Plains Ecoregion (Ecoregion Level III, code 56) (Figure 4a, 4b). Fifty-one percent of the Cisco lakes in Michigan lie within two smaller sub-sections (Level IV Ecoregions) of the Southern Michigan / Northern Indiana Drift Plains

Ecoregion: Battle Creek / Elkhart Outwash Plain (56b), and Interlobate Dead Ice Moraines (56h) (Figure 5a, 5b). Ecoregion 56b is 7,381 square kilometers and consists of scattered ice-block kettle lakes, two large streams, and numerous small streams within outwash deposits of sand and gravel mixed with small areas of end and ground moraines (Albert 1995, EPA 2012). Most of the area has been converted to agriculture, and the shorelines of the majority of the kettle lakes have been developed (Albert 1995). Within this region we have identified 42 lakes in which Cisco could potentially inhabit from the Cisco lake dataset. Sixty-four percent of these lakes are of unknown status, 26% were classified as stable, 3% as declining, and 7% as extirpated by Latta 1995 (Figure 6). Ecoregion 56h is 9,033 square kilometers and contains many kettle lakes. Similar to 56b, much of 56h was converted to farmland, and presently the majority of the area consists of residential and metropolitan areas, especially near Detroit (Albert 1995). This has resulted in the eutrophication and altered hydrology of many of the waterbodies in this ecoregion (Albert 1995). Within 56h, we have identified 51 potential Cisco lakes from the Cisco lake dataset. Fifty-three percent of these lakes are of unknown status. Latta (1995) classified 35% of these lakes as stable, 4% and declining, and 8% as extirpated (Figure 7). The lakes in ecoregion 56b and 56h are relatively small in area (Figure 8), and average 17 – 19m in depth (Figure 9). The lakes in 56b and 56h also have relatively high alkalinities when compared to other inland lakes in Michigan, indicating groundwater is a major source of lake water (Figure 10). Very little fisheries survey data are currently available on the sporadically sampled inland lakes, and many of the lakes have not been sampled or have been under-sampled. Of the 182 potential Cisco lakes in Michigan, 93 lakes (51%) are within ecoregions 56b and 56h. The remaining 89 lakes (49%) are dispersed among 19 different ecoregions throughout the state. Level III ecoregion 57 is the only Level III ecoregion that does not contain any Cisco lakes. Fifty-eight percent of the lakes in the combined ecoregions 56b and 56h are of unknown status (Figure 11). In the remaining 89 lakes from the 19 different ecoregions around Michigan, not including 56b and 56h, 57% were classified as stable (Figure 12). Two relatively small ecoregions (56b and 56h) contain the majority of the Cisco lakes in Michigan; and the majority of the Cisco lakes within these ecoregions are currently unsampled or of unknown status. With these two ecoregions experiencing relatively high agricultural land use and development, it is critical to prioritize sampling and conservation efforts in 56b and 56h.

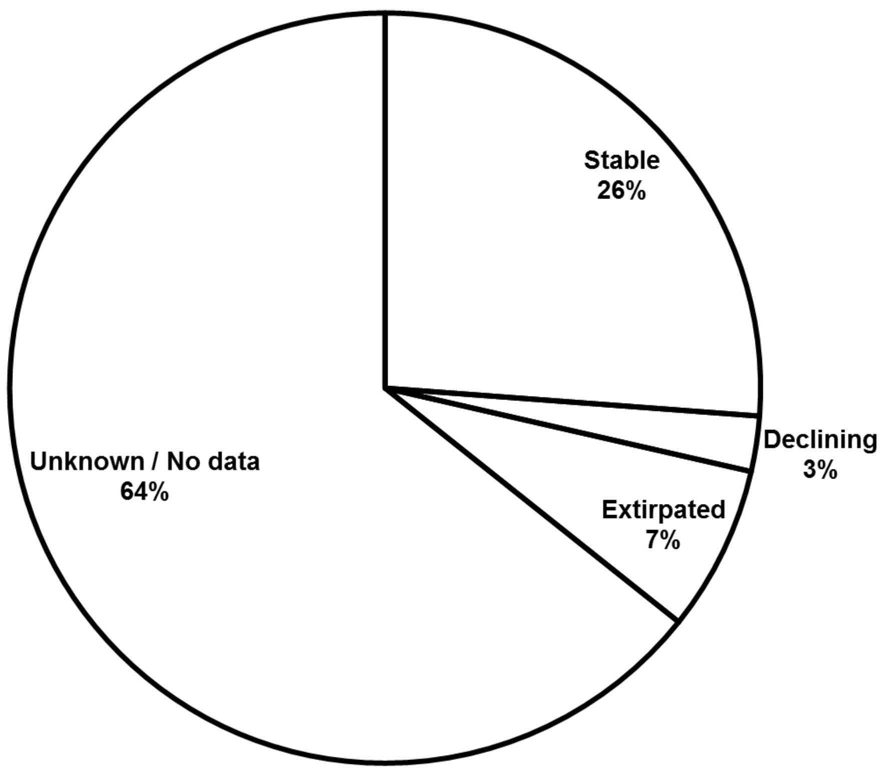


Figure 18. The status of Cisco lakes contained within the Level IV ecoregion 56b.

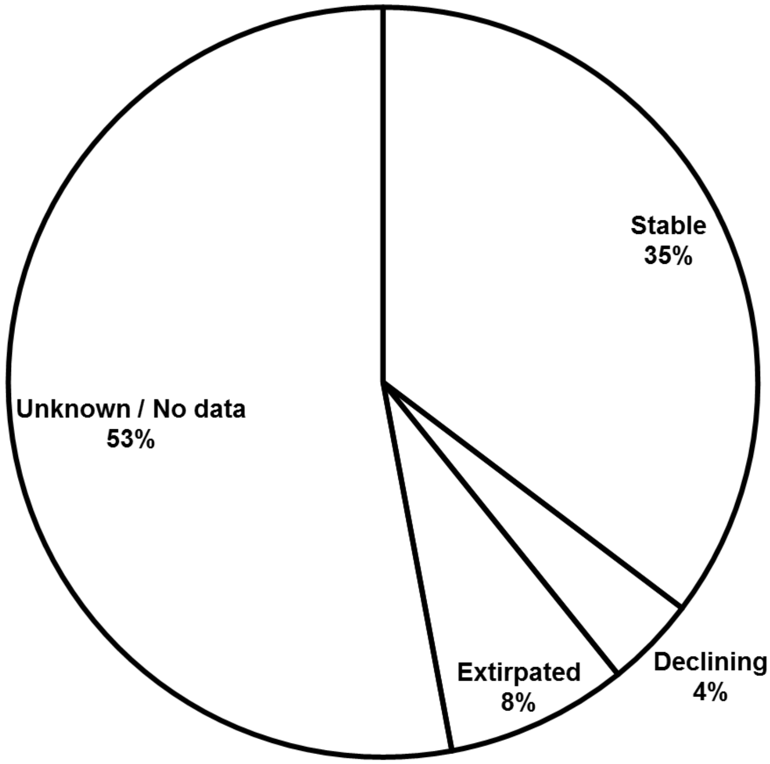


Figure 19. The status of Cisco lakes contained within the Level IV ecoregion 56h.

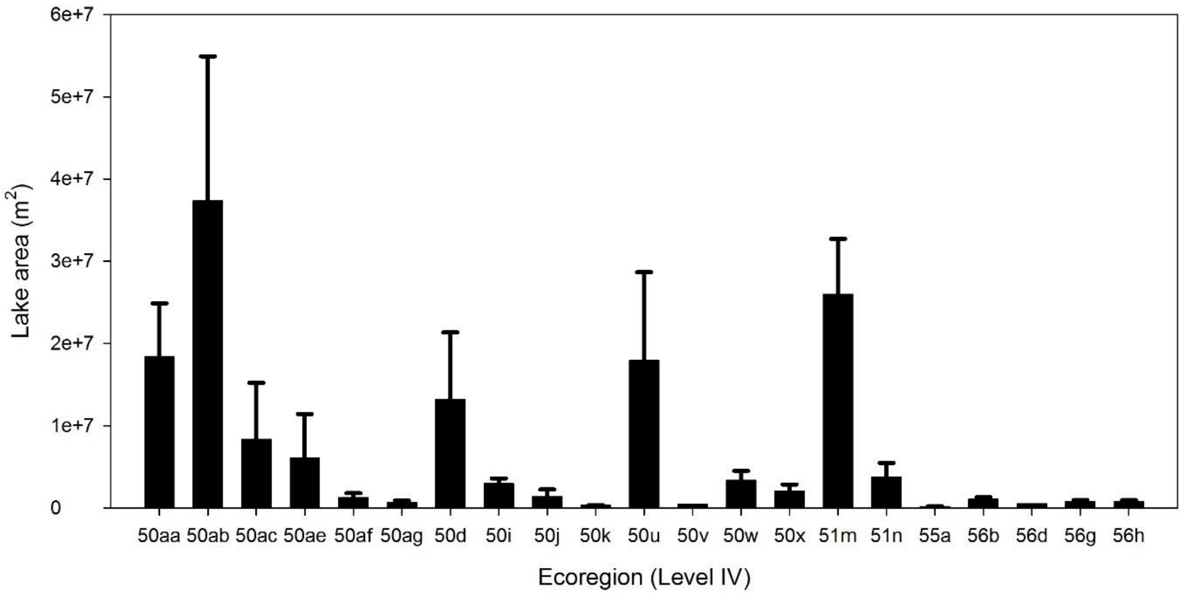


Figure 20. Average area (m^2) (± SE) of the Cisco lakes within each of the Level IV ecoregions in Michigan.

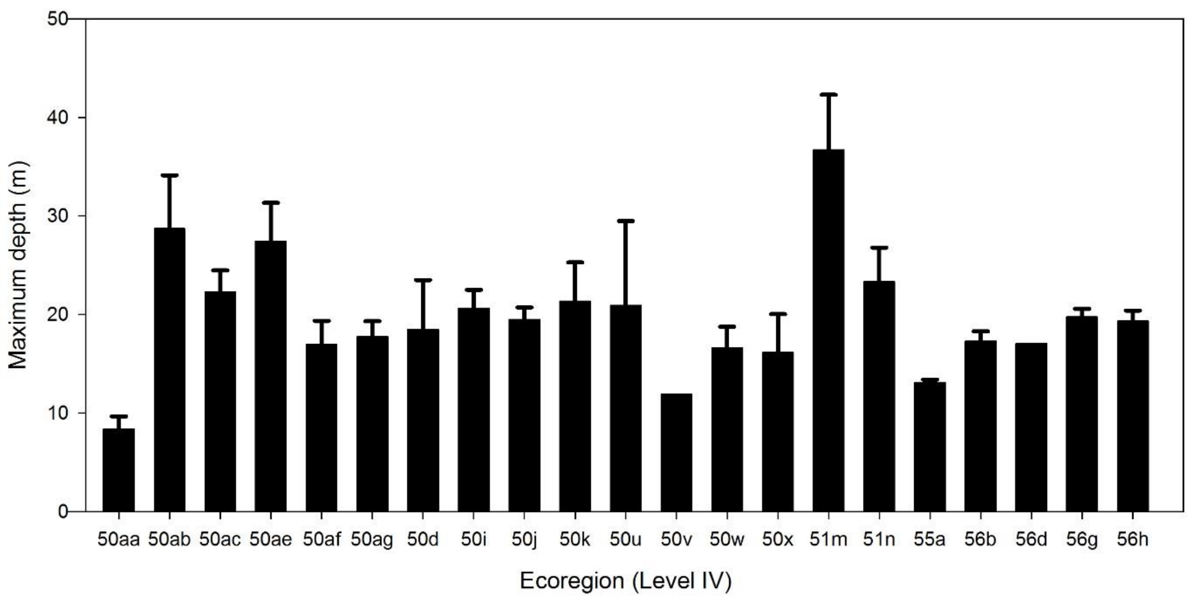


Figure 21. Average maximum depth (m) (± SE) of the Cisco lakes within each of the Level IV ecoregions in Michigan.

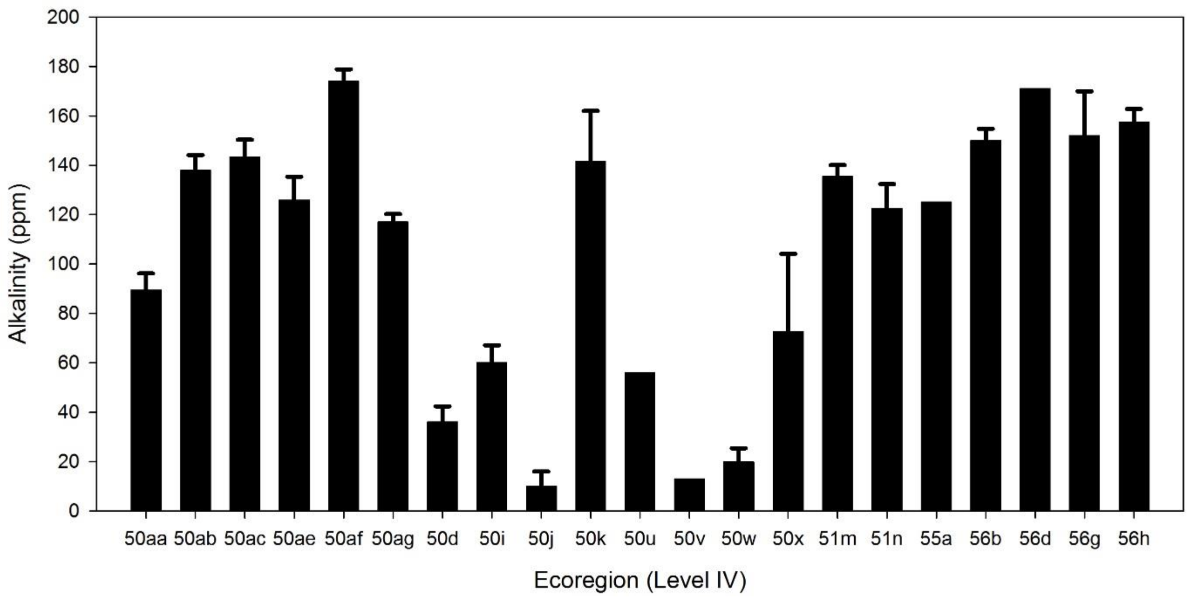


Figure 22. Average alkalinity (ppm) (± SE) of the Cisco lakes within each of the Level IV ecoregions in Michigan.

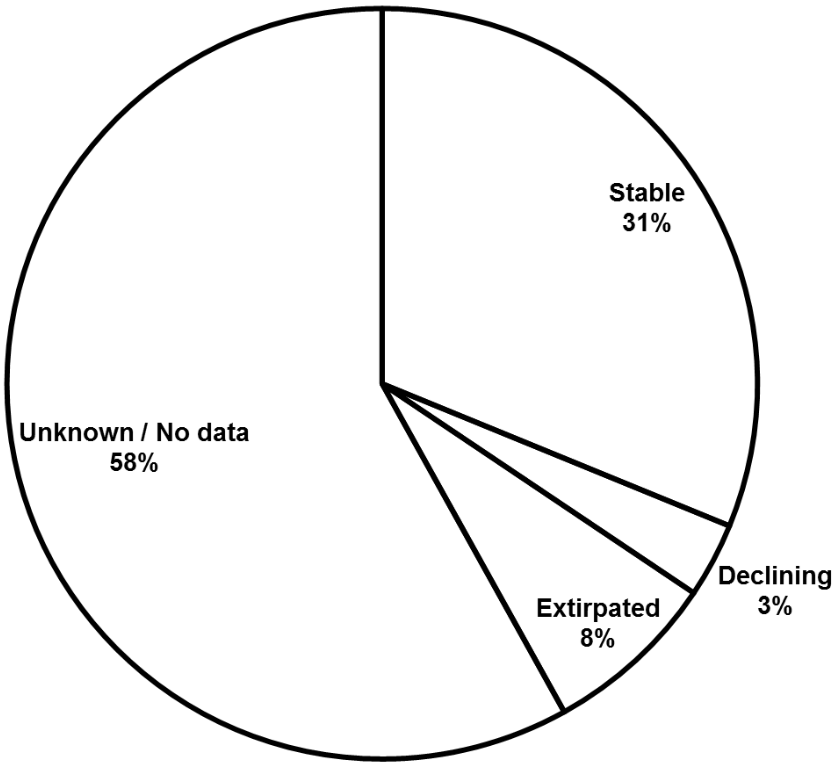


Figure 23. The status of the Cisco lakes within the combined Level IV ecoregions 56h and 56b.

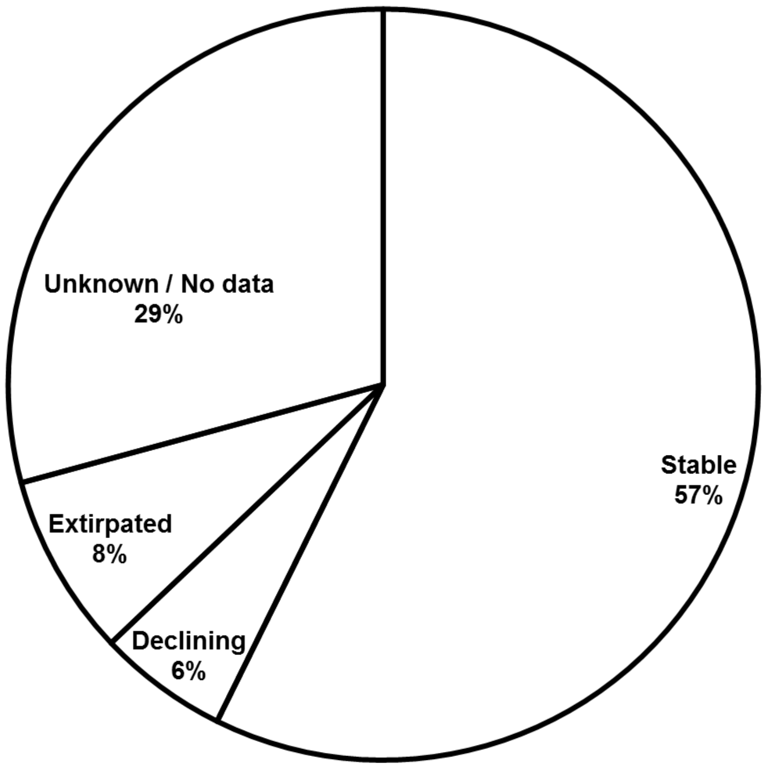


Figure 24. Cisco lake status in all other combined Level IV ecoregions, except 56b and 56h.

There was considerable overlap between the status classifications from Latta (1995) in the NMDS space (Figure 13). The status classifications of Latta (1995) were not separable by the first two NMDS dimensions, and no discernable causes of degradation and extinction can be drawn from this dataset. However, many larger lakes were of "stable" status, whereas all of the "declining" and "extirpated" lakes were smaller in area (P=0.01) (Figure 14). Areas of lakes with "declining" status were not different from lakes with "extirpated" status based upon post-hoc examination (P=0.99). However areas of lakes with "stable" status were different from areas of lakes with "declining" status (P=0.04) and "extirpated" status (P=0.01). Updated data and additional sampling will be required to examine the relationship between abiotic drivers of Cisco status.

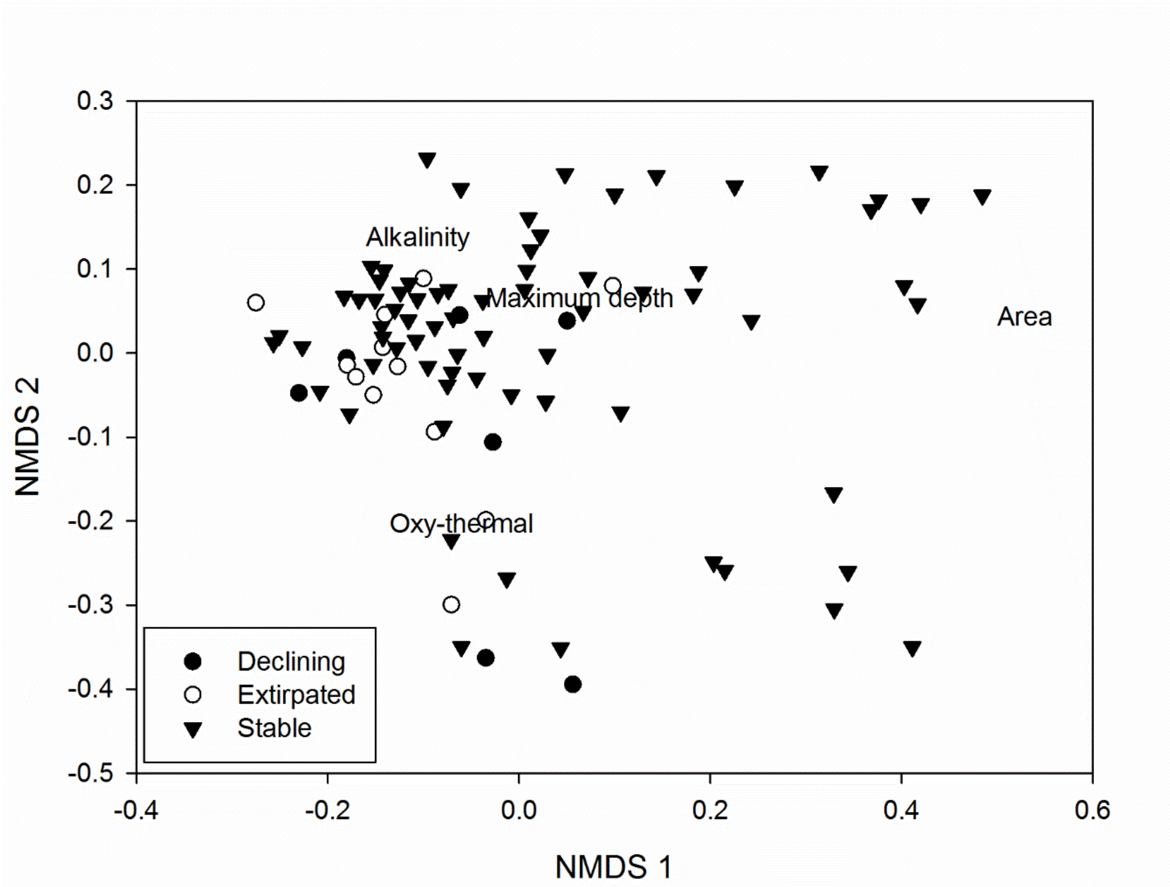


Figure 25. Ordination plot of the 90 examined Michigan Cisco lakes showing the distribution of each lake by status. “Declining”, “extirpated”, and “stable” classifications were given by Latta (1995) based upon catch-per-unit-effort of at least two samples from each lake.

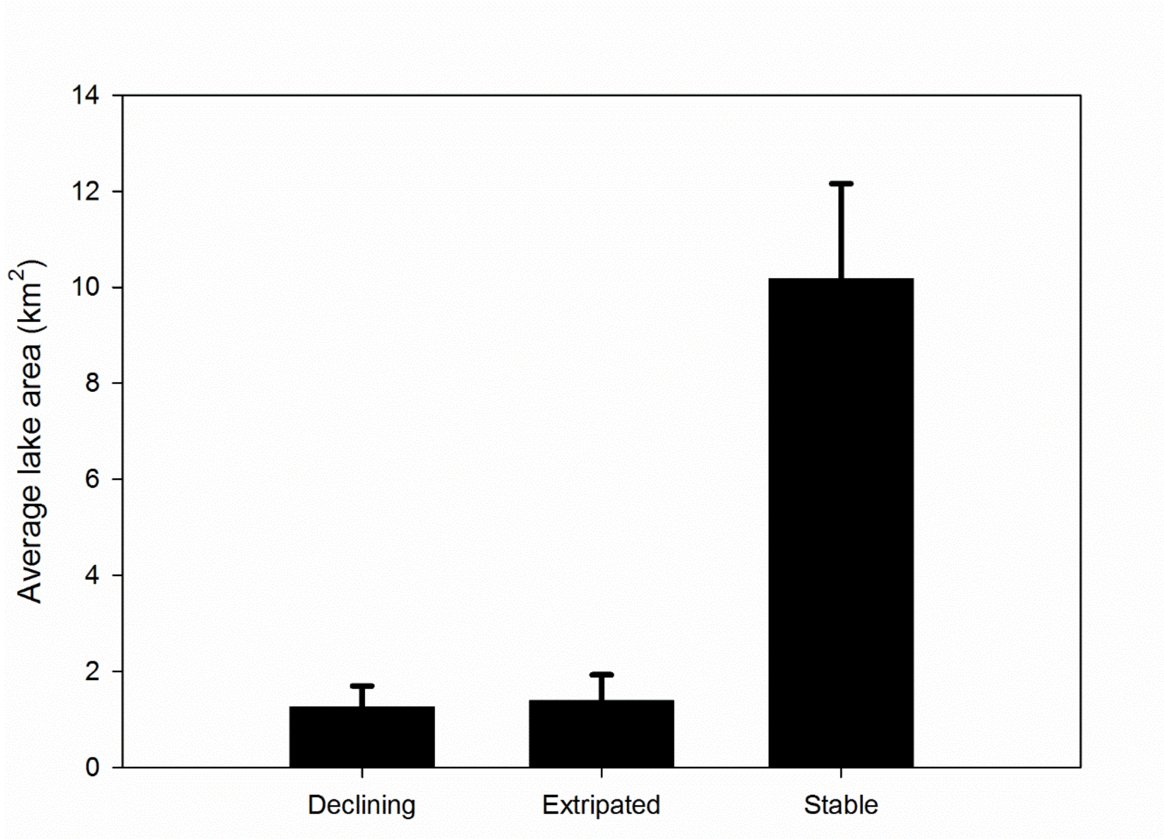


Figure 26. Average lake area (km^2) (± SE) of the 182 Cisco lakes of “declining”, “extirpated”, and “stable” status.

While evidence suggests that contemporary Michigan Cisco populations inhabit only lakes, there is substantial information indicating that Cisco historically frequented rivers across the state for spawning or as migration corridors (Smith 1972). The Manistee (Rozich 1998), the Au Sable (Zorn and Sendek 2001), Muskegon (O’Neal 1997), Flint (Leonardi and Gruhn 2001), and Manistique (Madison and Lockwood 2004) rivers and watersheds have all historically contained river spawning Cisco populations. River-spawning Cisco migrations still occur in less-altered Canadian systems (Lambert and Dodson 1990). These spawning runs were likely decimated by historic riverine habitat degradation, blocked migrations due to dams, and collapse of Great Lakes populations. If spawning runs are still occurring in Michigan rivers, they may go undetected as sampling effort has been limited.

As average air temperatures increase due to climate change, warmer epilimnetic water temperatures, a shallower thermocline and warmed hypolimnetic temperatures are expected (Sharma et al. 2011). Additionally, the increase in temperature and stratification period will cause reduced dissolved oxygen concentrations in the hypolimnion (Sharma et al. 2011, Fang et al. 2012). Because Cisco are extremely sensitive to changes in water temperature and dissolved oxygen, climate change could result in declines of the species throughout Michigan. Latta (1995) reported the most common cause of extirpation of Cisco is the loss of the "cisco layer" – the layer where the temperature is less than 20^{o}C and dissolved oxygen greater than or equal to 3.0mg/L (Colby and Brooke 1969). Reductions in habitat will be accompanied by reductions in range, especially in relatively shallower lakes in the lower latitudes (Fang et al. 2004, Sharma et al. 2011, Jacobson et al. 2013). Jacobson et al. (2013) and Fang et al. (2012) used climate/water quality models to examine inland lakes that would provide suitable thermal habitat for Cisco after climate warming in Minnesota and Wisconsin respectively. Models predict that 67% of current Cisco lakes in Wisconsin could become non-refuge (Fang et al. 2012), and over 70% of Cisco could be extirpated by 2100 (Sharma et al. 2011). Lakes in the most southern regions of the Cisco range are most vulnerable to climate change, exacerbated by agricultural land practices in these areas (Jacobson et al. 2013, Sharma et al. 2011). Cisco lakes in Michigan are found in similar latitudes in high agricultural land use areas and will be susceptible to these same threats. The assessment of Michigan's inland waters are critical, as extirpation of Cisco in neighboring states along the same latitudes have been projected to occur within this century (Sharma et al. 2011, Jacobson et al. 2012, Jacobson et al. 2013, Fang et al. 2012). If Michigan lakes follow the same trend, urbanization and nutrient enrichment resulting in habitat degradation could lead to the potential extirpation of Cisco in the majority of the inland lakes in Michigan. Ecoregions 56b and 56h contain many sources of groundwater, and subsequently higher alkalinities (Figure 10), which may have contributed to these ecoregions historically being more suitable to Cisco (Sampath et al. 2015). The lakes in ecoregions 56b and 56h are relatively small when compared to the Cisco lakes in other ecoregions, yet these two ecoregions are the dominant areas where Cisco lakes are concentrated in Michigan. The influence of groundwater on the lakes in 56b and 56h could be important to the survival and persistence of Cisco in these ecoregions. The groundwater sources of these lakes may provide some resiliency to climate change, in addition to agricultural and land use inputs, which can degrade Cisco habitat; which is why the protection and conservation of ground water is important to Cisco survival in these ecoregions.

Eutrophication has long been recognized as one of the greatest threats to Cisco (Becker et al. 1983, Latta 1995). This relationship has been recognized for almost 60 years. Edwin Cooper, Chief Fishery Biologist, Wisconsin Department of Natural Resources, commented in a 1956 issue of the Wisconsin Conservation Bulletin saying, "Declines of cisco populations have been noted over the past 20 years in lakes which have become increasingly fertile through the actions of man. The generous use of fertilizers in agriculture and the leaching of them into lakes, effluents of sewage disposal systems, even when completely treated, and the widescale erosion of fertile top-soil into lake drainage basins have all resulted in enriching many of the deep lakes in Wisconsin." One result of this eutrophication is precipitous declines in summer dissolved oxygen concentrations, which we have already discussed as a critical element of Cisco survival. The combined effects of eutrophication and climate change could be disastrous for Cisco. Under climate change scenarios, there will be a longer growing season, leading to higher pelagic primary productivity, ultimately resulting in increased depletion rates of DO (Sharma et al. 2011). Furthermore, increased projected runoff from extreme precipitation events under climate change scenarios would also intensify eutrophication. However since 1999, there appears to be an increasing number of lakes with an "oligotrophic" classification and a decreasing number of lakes with an "eutrophic" classification in the Southern Michigan / Northern Indiana Drift Plains ecoregion (Level III, 56) (Fuller and Jodoin 2016). However, this ecoregion did have a higher number of eutrophic lakes and a lower number of oligotrophic lakes when compared to the other Level III ecoregions (Fuller and Jodoin 2016).

A negative response of Cisco to Rainbow Smelt has been observed in both the Great Lakes and inland systems (Fitzsimons & O'Gorman 2006, Sharma et al. 2011, Latta 1995, Krueger & Hrabik 2005, Stockwell et al. 2009, Tsehaye et al. 2014, Ebener et al. 2008). Predation by Rainbow Smelt upon larval Ciscoes in Lake Superior was a driving factor in the lack of recruitment of Cisco (Stockwell et al. 2009). Additionally, Rainbow Smelt predation has been recognized as an impediment to the recovery of Cisco in Lake Michigan (Madenjian et al. 2002, Fitzsimons & O'Gorman 2006). In northern, temperate, inland lakes, rapid declines in coregonid populations have been observed following the establishment of Rainbow Smelt (Krueger & Hrabik 2005). Rainbow Smelt invasions have been directly associated with changes in the zooplankton community and the extirpation of Cisco through predation and competition (Hrabik et al. 1998, Sharma et al. 2011). Krueger & Hrabik (2005) found that Walleye *Sander vitreus* reduced the density, size and consumption of Rainbow Smelt which decreased Cisco mortality in northern Wisconsin lakes. With the future status of Cisco populations intertwined with the success of rainbow smelt, controlling this invasive species is of the highest importance.

Habitat degradation and the lack of knowledge of the Cisco stocks in Michigan are the greatest impediments to their conservation and recovery. The protection of water quality that allows for high hypolimnetic oxygen concentrations is crucial (Jacobson et al. 2013). Systematically prioritizing areas and focusing conservation efforts at the catchment scale is of the highest importance in the conservation of Cisco (Jacobson et al. 2013). We suggest that prioritizing the sampling and monitoring of Level IV ecoregions 56b and 56h, as these two ecoregions contain the majority of Cisco lakes in Michigan, and are also under the greatest threats of habitat degradation. Further, we believe evaluating the status of rivers is required to fully examine potential spawning activity and how Cisco are using rivers as corridors, particularly if Great Lakes populations are increasing. It is critical to determine the stock structure, sources of mortality, genetic diversity, and habitats in each inland lake in these ecoregions. Management considerations for the conservation and future reestablishment of Great Lakes and inland Cisco have been suggested in the Inland Cisco Lakes Wildlife Action Plan (Derosier et al. 2015). An ecoregion-based management approach that incorporates climate and water quality will be required to restore Cisco populations in the inland lakes of Michigan (Stockwell et al. 2009, Gorman et al. 2012, Yule et al. 2012, Rook et al. 2013, Sharma et al. 2011).

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APPENDICES

Appendix 1. Features of the 153 Cisco lakes described in Latta (1995) and 29 additional Cisco lakes (in bold).

Lake ID	Lake name	Latitude	Longitude	Level IV ecoregion code	Area (km^2)	Maximum depth (m)	Oxygen-thermal	Alkalinity (ppm)
1	Au Train	46.404203	-86.838411	50x	3.36	9.14	5	120
2	Deer	46.529067	-87.688686	50w	0.98	22.86	1	8
3	Green	42.751364	-85.595183	56g	1.25	21.03	2	149
4	Beaver	44.938217	-83.798522	50ac	2.69	23.47	3	150
5	Hubbard	44.801497	-83.550458	50ac	35.81	26.52	3	158
6	Bellaire	44.950547	-85.217122	51m	7.18	28.96	1	160
7	Elk	44.861206	-85.385214	51m	31.28	58.52	1	140
8	Intermediate	45.028242	-85.230933	51m	6.13	24.99	2	157
9	Torch	44.942222	-85.309258	51m	75.96	86.87	1	133
10	Big Cedar	42.511458	-85.348128	56h	0.33	10.67	No Data	188
11	Little Cedar	42.528708	-85.340814	56h	0.15	12.80	2	175
12	Barlow	42.671728	-85.518814	56b	0.76	19.20	2	127
13	**Carr Lake & Mud Lake**	42.720475	-85.073108	56g	0.12	No Data	No Data	No Data
14	Gull	42.401489	-85.411983	56b	8.22	33.53	2	116
15	Long	42.475175	-85.243506	56h	0.24	13.11	2	135
16	Lime	42.558186	-85.494156	56h	0.08	11.58	3	146
17	Fish	42.554933	-85.497456	56h	0.67	17.07	2	165

18	Lake Ann	44.717239	-85.8479	51n	2.13	22.86	2	151
19	Crystal	44.661511	-86.171433	51m	39.30	49.38	1	106
20	**Archer Lake & Middle Lake**	41.883628	-84.922014	56b	0.26	10.97	No Data	No Data
21	Bartholomew	41.876944	-84.929378	56b	0.30	17.07	3	174
22	Coldwater	41.828414	-84.9771	56b	6.52	28.04	3	127
23	**Dorsey**	41.910661	-85.12585	56b	0.02	No Data	No Data	No Data
24	**Huyck**	41.778528	-84.978575	56b	0.75	No Data	No Data	No Data
25	**Kenyon**	42.050375	-85.251128	56b	0.25	No Data	No Data	No Data
26	East Long Lake	41.850447	-84.967775	56b	0.50	13.72	3	162
27	Marble	41.903556	-84.90575	56b	3.16	18.29	3	148
28	**Morrison**	41.988733	-85.0292	56b	1.17	14.02	No Data	No Data
29	**Pleasant**	41.781533	-85.0292	56b	0.30	No Data	No Data	No Data
30	**Little Rose**	41.863197	-85.039372	56b	1.44	23.16	No Data	No Data
31	Baldwin	41.776433	-85.828897	56b	1.08	16.76	3	164

32	Birch	41.878739	-85.856406	56b	1.19	28.96	1	122
33	Bunker	42.043897	-85.904503	56h	0.43	17.68	3	No Data
34	**Chain**	41.857306	-85.893514	56b	0.14	12.50	No Data	No Data
35	Curtis	41.853169	-85.940378	56b	0.09	No Data	No Data	No Data
36	Day	41.850803	-85.92765	56b	0.09	7.62	No Data	No Data
37	Donnell	41.907906	-85.890936	56b	1.00	19.20	3	149
38	Harwood	41.928864	-85.768244	56h	0.49	16.76	2	171
39	**Indiana**	41.761542	-85.832811	56b	0.46	No Data	No Data	No Data
40	**Kirk**	41.929331	-85.879722	56b	0.17	7.01	No Data	No Data
41	**Lewis**	41.906881	-85.874833	56b	0.09	8.53	No Data	No Data
42	**Lime**	41.900617	-85.842244	56h	No Data	6.10	No Data	No Data
43	**Long**	41.850014	-85.901133	56b	0.25	No Data	No Data	No Data
44	Long Lake	41.773458	-85.818997	56b	0.98	13.41	3	222
45	**Round**	41.852228	-85.891806	56b	0.03	0.00	No Data	No Data
46	Shavehead	41.843414	-85.866697	56b	1.17	21.34	3	171
47	**Tharp Lake**	41.850069	-85.916014	56b	0.15	No Data	No Data	No Data

48	**Weatherbee**	41.8995	-85.836592	56h	No Data	No Data	No Data	No Data
49	**Wood**	41.856725	-85.778903	56h	0.21	No Data	No Data	No Data
50	**Little Wood**	41.858217	-85.772206	56h	0.04	No Data	No Data	No Data
51	Charlevoix	45.273931	-85.151556	51m	69.85	31.09	2	133
52	Walloon	45.276444	-85.007767	51m	17.48	30.48	2	112
53	Burt	45.464647	-84.662306	50ab	67.58	22.25	2	133
54	Douglas	45.580567	-84.698017	50ab	13.74	25.60	2	126
55	Mullett	45.516736	-84.516894	50ab	67.30	44.81	1	138
56	Twin Lakes 2,3,4	45.539244	-84.292819	50ab	0.81	22.25	2	155
57	Hulbert	46.323331	-85.121619	50x	2.25	22.56	1	85
58	Monacle	46.474489	-84.645969	50x	0.59	16.76	2	13
59	Mary	45.75145	-87.820658	50k	0.35	25.30	2	162
60	Louise	45.749567	-87.810594	50k	0.32	17.37	2	121
61	**Saubee**	42.729028	-85.060597	56g	No Data	No Data	No Data	No Data

62	**Mud**	42.719378	-85.07085	56g	0.06	No Data	No Data	No Data
63	Clark	46.229928	-89.329164	50i	3.60	22.86	2	No Data
64	Crooked	46.224572	-89.283986	50i	2.29	18.29	3	70
65	Gogebic	46.510844	-89.585703	50d	51.80	11.28	5	25
66	Loon	46.204256	-89.296542	50i	1.52	16.76	No Data	0
67	Norwood	46.111192	-89.016644	50j	0.49	18.29	1	4
68	Taylor	46.245836	-89.040836	50v	0.45	11.89	2	13
69	Thousand Island	46.227447	-89.404289	50i	4.37	24.69	2	50
70	Bridge	44.638958	-85.786525	51n	0.13	11.89	3	110
71	Cedar Hedge	44.672081	-85.781325	51n	0.63	21.03	3	92
72	Duck	44.623097	-85.747706	51n	7.81	29.87	1	126
73	Green	44.607453	-85.78635	51n	8.04	31.09	2	133
74	Bear	41.869008	-84.68035	56h	0.47	16.15	4	150
75	Carpenter	41.888153	-84.796581	56b	0.14	21.34	3	132
76	**Denton Chain**	41.845017	-84.798922	56b	0.25	11.28	No Data	No Data

77	Hemlock	41.895883	-84.790961	56b	0.59	19.81	2	137
78	Long	41.874956	-84.794336	56b	0.86	13.72	3	171
79	Middle Sand	41.925953	-84.699386	56h	0.26	11.58	No Data	No Data
80	North Sand	41.941947	-84.706019	56h	0.26	12.19	No Data	No Data
81	South Sand	41.913831	-84.694469	56h	0.32	9.75	No Data	No Data
82	Wilson	41.879125	-84.684722	56h	0.37	18.29	2	139
83	Otter	46.913269	-88.573906	50u	3.60	8.84	1	56
84	Portage	47.063419	-88.497897	50u	39.02	16.46	No Data	No Data
85	Torch	47.167622	-88.413803	50u	11.13	37.49	1	No Data
86	Loon	44.410286	-83.822731	50ae	1.69	39.01	2	109
87	Smoky	46.095006	-88.941636	50j	2.26	20.73	3	16
88	Coldwater	43.663261	-84.958378	50af	1.19	19.81	3	180
89	Littlefield	43.773356	-84.944272	50af	2.23	18.90	2	168
90	Brown	42.187872	-84.419108	56h	0.85	10.67	3	161
91	Swains	42.152017	-84.650439	56h	0.28	19.51	2	125
92	Vandercook	42.190253	-84.403403	56h	0.58	12.80	3	239

93	**Crooked**	42.204922	-85.708864	56h	0.66	15.24	No Data	No Data
94	Howard	42.080067	-85.589933	56b	0.44	14.02	No Data	134
95	Indian	42.152933	-85.484325	56b	3.07	21.03	3	129
96	Little Paw Paw	42.219364	-86.289581	56d	0.51	17.07	3	171
97	Sagamaw	42.172392	-85.448706	56b	0.13	No Data	No Data	No Data
98	Blue	44.808031	-84.894958	50ae	0.46	26.21	2	137
99	Twin	44.821592	-84.964636	50ae	0.87	27.43	2	95
100	North Blue	44.817478	-84.896325	50ae	0.22	23.77	2	147
101	**Skegemog**	44.806033	-85.327678	51m	No Data	No Data	No Data	No Data
102	Murray	43.034719	-85.374247	56g	1.29	21.95	3	139
103	Ziegenfuss	43.176931	-85.338331	50af	0.32	12.19	2	No Data
104	Desor	47.975728	-88.990019	50d	4.25	16.76	No Data	No Data
105	Fanny Hooe	47.464094	-87.862283	50d	0.95	14.63	3	47
106	Richie	48.041714	-88.698981	50d	2.10	11.28	No Data	No Data
107	Sargent	48.092442	-88.657256	50d	1.49	13.72	No Data	No Data

108	Siskiwit	47.999119	-88.799119	50d	18.45	43.28	No Data	No Data
109	Little Bass	44.091028	-85.968036	50ag	0.22	13.72	3	120
110	Glen	44.868722	-85.960453	51m	19.69	39.62	1	135
111	Little Traverse	44.921311	-85.841706	51m	2.59	16.46	2	136
112	North Lake Leelanau	45.025292	-85.740197	51m	11.94	36.88	1	145
113	South Lake Leelanau	44.869089	-85.715631	51m	21.73	18.90	2	149
114	Appleton	42.510197	-83.834161	56h	0.22	11.58	3	155
115	Bass	42.454042	-83.862014	56h	0.74	22.25	2	179
116	Bennett	42.772278	-83.829442	56g	0.54	17.68	4	239
117	Chemung	42.582325	-83.848594	56g	1.27	21.34	4	123
118	Crooked	42.548283	-83.846756	56h	0.15	16.15	3	227
119	Fish	42.454808	-83.7232	55a	0.13	13.41	2	125
120	Limekiln	42.453264	-83.706125	56h	0.11	10.67	No Data	No Data
121	Ore	42.479825	-83.796072	56h	0.95	24.69	4	No Data

122	Portage	42.426394	-83.913444	56h	2.61	25.60	4	168
123	Runyan	42.75855	-83.750017	56g	0.81	16.76	2	110
124	Sandy Bottom	42.451944	-83.714819	55a	0.21	12.80	No Data	No Data
125	Zukey	42.459183	-83.846058	56h	0.63	13.41	4	222
126	N. Manistique	46.287517	-85.735842	50aa	0.22	13.72	3	120
127	Brevoort	45.994331	-84.919775	50aa	17.12	9.14	5	78
128	Manistique	46.231372	-85.775658	50aa	40.99	6.10	5	87
129	South Manistique	46.166692	-85.770914	50aa	16.19	8.84	5	85
130	Pine	44.193653	-86.004381	50ag	0.64	17.68	2	107
131	Portage	44.358367	-86.239519	51m	8.54	18.29	2	120
132	(First) Pine	46.879136	-87.876992	50w	No Data	No Data	No Data	No Data
133	Independence	46.805942	-87.698744	50w	8.04	9.75	5	44
134	Ives	46.847061	-87.849864	50w	No Data	No Data	No Data	No Data

135	Lake Ann	46.871733	-87.928103	50w	No Data	No Data	No Data	No Data
136	Mountain	46.865075	-87.911753	50w	No Data	No Data	No Data	No Data
137	Rush	46.889878	-87.915181	50w	No Data	No Data	No Data	No Data
138	Silver Lake Basin	46.658219	-87.836044	50w	4.05	21.34	1	8
139	Sporley	46.333164	-87.339764	50w	0.31	12.50	2	19
140	Avalon Lake	45.103339	-83.955933	50ac	1.51	22.56	2	118
141	Long	45.127239	-83.973267	50ac	1.20	24.99	2	140
142	Muskellunge	45.105422	-84.19195	50ac	0.46	14.02	3	151
143	**Kimball**	43.455133	-85.826767	50ag	No Data	No Data	No Data	No Data
144	Nichols	43.726875	-85.906231	50ag	0.64	17.37	2	124
145	Pickerel	43.457575	-85.812086	50ag	1.29	22.25	2	117
146	Lake Angelus	42.690853	-83.318953	56h	1.67	28.04	2	120
147	Cass	42.605619	-83.365289	56h	5.22	36.58	2	170

148	Cedar Island	42.629769	-83.4805	56h	0.58	21.95	2	188
149	North/South Commerce	42.580064	-83.493372	56h	1.17	20.12	3	151
150	Deer	42.732792	-83.433464	56h	0.55	19.20	2	154
151	Dunham	42.652792	-83.678422	56h	0.45	38.10	2	No Data
152	Green	42.592342	-83.417392	56h	0.67	21.95	2	109
153	Hammond	42.606603	-83.325028	56h	0.30	34.14	2	82
154	Loon	42.680661	-83.358044	56h	0.98	22.25	3	188
155	Maceday	42.688228	-83.430864	56h	0.89	33.53	1	158
156	Orchard	42.585697	-83.370625	56h	3.19	33.83	3	93
157	Oxbow	42.645783	-83.479125	56h	1.09	21.95	3	No Data
158	Schoolhouse	42.685372	-83.348514	56h	0.15	14.94	2	137
159	**Silver**	42.677756	-83.340461	56h	No Data	No Data	No Data	No Data
160	Townsend	42.707936	-83.400408	56h	0.11	16.76	2	191
161	Union	42.607144	-83.431264	56h	1.88	33.53	3	102

162	Upper Pettibone	42.665864	-83.612664	56h	0.17	16.76	2	161
163	Devoe	44.402369	-84.024647	50ae	0.53	16.15	3	164
164	Grousehaven	44.411475	-84.019892	50ae	0.36	16.76	3	127
165	Higgins	44.480933	-84.714122	50ae	38.20	42.98	1	102
166	Gulliver	45.982706	-86.02725	50aa	3.38	7.92	5	93
167	Indian	45.984806	-86.327214	50aa	32.37	4.57	5	74
168	Corey	41.930225	-85.740933	56b	2.55	24.38	4	105
169	**Fish**	41.877039	-85.478683	56b	1.11	22.86	No Data	No Data
170	Klinger	41.805283	-85.543728	56b	3.36	21.95	1	110
171	Tamarack	41.811386	-85.518386	56b	0.30	14.63	2	179
172	**Pepper**	41.86215	-85.341792	56b	0.08	No Data	No Data	No Data
173	Pleasant	41.958164	-85.702444	56h	1.06	12.19	3	127
174	**Prairie River**	41.859387	-85.401716	56b	No Data	No Data	No Data	No Data
175	Thompson	41.825764	-85.489794	56b	0.62	9.14	3	185
176	Wolf	42.298747	-85.787225	56b	0.11	11.28	2	188

177	Baseline	42.4237	-83.894197	56h	1.03	19.51	4	205
178	Blind	42.415206	-84.019719	56h	0.28	24.38	2	171
179	Bruin	42.418086	-84.039539	56h	0.50	14.63	3	105
180	Halfmoon	42.419108	-84.011858	56h	0.96	24.99	4	No Data
181	Pickerel	42.410139	-83.982661	56h	0.10	17.07	3	No Data
182	South	42.398225	-84.068175	56h	3.30	25.30	2	No Data

Appendix 2. Updated capture data (in bold under "Last Capture") of the 153 Cisco lakes from Latta (1995) and the 29 additional Cisco lakes (in bold under "Lake Name").

Lake ID	Lake Name	Level 4 Ecoregion Code	First Capture	Last Capture	Status
1	Au Train	50x	1951	**2002**	Stable
2	Deer	50w	1953	**2004**	Declining
3	Green	56g	1952	**2003**	Stable
4	Beaver	50ac	1925	1987	Stable
5	Hubbard	50ac	1925	1986	Stable
6	Bellaire	51m	1931	1987	Stable
7	Elk	51m	1888	1990	Stable

8	Intermediate	51m	1931	**1999**	Stable
9	Torch	51m	1888	**2002**	Stable
10	Big Cedar	56h	1890	**2003**	Stable
11	Little Cedar	56h	1962	No Data	Unknown
12	Barlow	56b	1951	1977	Stable
13	**Carr Lake & Mud Lake**	56g	No Data	No Data	No Data
14	Gull	56b	1886	1954	Extripated
15	Long	56h	1988	**2003**	Unknown
16	Lime	56h	1946	No Data	Unknown
17	Fish	56h	1946	1994	Stable
18	Lake Ann	51n	1950	1992	Stable
19	Crystal	51m	1940	**2003**	Stable
20	**Archer Lake & Middle Lake**	56b	No Data	No Data	No Data
21	Bartholomew	56b	1948	No Data	Unknown
22	Coldwater	56b	1886	1967	Stable
23	**Dorsey**	56b	No Data	No Data	No Data
24	**Huyck**	56b	No Data	No Data	No Data

25	**Kenyon**	56b	No Data	1992	No Data
26	East Long Lake	56b	1886	1941	No Data
27	Marble	56b	1941	1986	Stable
28	**Morrison**	56b	No Data	No Data	No Data
29	**Pleasant**	56b	No Data	No Data	No Data
30	**Little Rose**	56b	No Data	No Data	No Data
31	Baldwin	56b	1887	1990	Stable
32	Birch	56b	1887	1990	Stable
33	Bunker	56h	1949	No Data	Unknown
34	**Chain**	56b	No Data	No Data	No Data
35	Curtis	56b	1948	No Data	Unknown
36	Day	56b	1948	No Data	Unknown
37	Donnell	56b	1887	1947	Extripated
38	Harwood	56h	1953	1990	Stable
39	**Indiana**	56b	No Data	**2001**	No Data
40	**Kirk**	56b	No Data	No Data	No Data
41	**Lewis**	56b	No Data	No Data	No Data

42	**Lime**	56h	No Data	No Data	No Data
43	**Long**	56b	No Data	No Data	No Data
44	Long Lake	56b	1887	1980	Stable
45	**Round**	56b	No Data	No Data	No Data
46	Shavehead	56b	1887	**2000**	Stable
47	**Tharp Lake**	56b	No Data	No Data	No Data
48	**Weatherbee**	56h	No Data	No Data	No Data
49	**Wood**	56h	No Data	No Data	No Data
50	**Little Wood**	56h	No Data	No Data	No Data
51	Charlevoix	51m	1926	1990	Stable
52	Walloon	51m	1890	1986	Stable
53	Burt	50ab	1887	**2001**	Stable
54	Douglas	50ab	1959	**2000**	Stable
55	Mullett	50ab	1887	**1998**	Stable
56	Twin Lakes 2,3,4	50ab	1968	**2000**	Stable
57	Hulbert	50x	1940	1953	Stable
58	Monacle	50x	1976	**1998**	Declining

59	Mary	50k	1945	1986	Stable
60	Louise	50k	1950	1956	Stable
61	**Saubee**	56g	1987	1987	No Data
62	**Mud**	56g	1980	No Data	No Data
63	Clark	50i	1966	**2000**	Stable
64	Crooked	50i	1938	1969	Unknown
65	Gogebic	50d	1938	1992	Stable
66	Loon	50i	1966	1983	Stable
67	Norwood	50j	1961	No Data	Unknown
68	Taylor	50v	1960	1972	Stable
69	Thousand Island	50i	1969	1975	Stable
70	Bridge	51n	1950	No Data	Unknown
71	Cedar Hedge	51n	1967	1977	Stable
72	Duck	51n	1950	**1997**	Stable
73	Green	51n	1947	**2003**	Stable
74	Bear	56h	1945	No Data	Unknown
75	Carpenter	56b	1886	**2004**	Unknown

76	**Denton Chain**	56b	1995	**1995**	No Data
77	Hemlock	56b	1886	**2004**	Stable
78	Long	56b	1886	1976	Unknown
79	Middle Sand	56h	1886	**2004**	Unknown
80	North Sand	56h	1992	**2004**	Unknown
81	South Sand	56h	1886	**2004**	Unknown
82	Wilson	56h	1963	No Data	Unknown
83	Otter	50u	1925	1970	Declining
84	Portage	50u	1930	1988	Unknown
85	Torch	50u	1971	1988	Stable
86	Loon	50ae	1931	1981	Stable
87	Smoky	50j	1938	**2001**	Stable
88	Coldwater	50af	1952	1966	Extripated
89	Littlefield	50af	1950	1960	Extripated
90	Brown	56h	1889	1988	Stable
91	Swains	56h	1889	1940	Extripated
92	Vandercook	56h	1889	1988	Stable

93	**Crooked**	56h	No Data	No Data	No Data
94	Howard	56b	1962	1991	Stable
95	Indian	56b	1888	1965	Stable
96	Little Paw Paw	56d	1943	1969	Declining
97	Sagamaw	56b	No Data	1980	No Data
98	Blue	50ae	1930	**1998**	Stable
99	Twin	50ae	1930	**1999**	Stable
100	North Blue	50ae	1930	**2003**	Stable
101	**Skegemog**	51m	1996	No Data	No Data
102	Murray	56g	1927	1990	Stable
103	Ziegenfuss	50af	1891	1971	Stable
104	Desor	50d	1929	No Data	Unknown
105	Fanny Hooe	50d	1926	1952	Extripated
106	Richie	50d	1929	No Data	Unknown
107	Sargent	50d	1929	No Data	Unknown
108	Siskiwit	50d	1929	No Data	Unknown
109	Little Bass	50ag	1953	No Data	Unknown

110	Glen	51m	1949	**1997**	Stable
111	Little Traverse	51m	1970	No Data	Unknown
112	North Lake Leelanau	51m	1949	**2002**	Stable
113	South Lake Leelanau	51m	1967	1994	Stable
114	Appleton	56h	1956	1991	Declining
115	Bass	56h	1952	1977	Unknown
116	Bennett	56g	1968	1979	Unknown
117	Chemung	56g	1942	1956	Extripated
118	Crooked	56h	1943	1970	Stable
119	Fish	55a	1972	No Data	Unknown
120	Limekiln	56h	1970	No Data	Unknown
121	Ore	56h	1890	1953	Extripated
122	Portage	56h	1880	1967	Stable
123	Runyan	56g	1979	No Data	Unknown
124	Sandy Bottom	55a	1970	No Data	Unknown
125	Zukey	56h	1985	No Data	Unknown
126	N. Manistique	50aa	1926	**2003**	Stable

127	Brevoort	50aa	1979	**1997**	Stable
128	Manistique	50aa	1936	**2003**	Stable
129	South Manistique	50aa	1926	**2003**	Stable
130	Pine	50ag	1932	1994	Stable
131	Portage	51m	1948	1976	Stable
132	(First) Pine	50w	1927	No Data	Unknown
133	Independence	50w	1953	1994	Stable
134	Ives	50w	1927	No Data	Unknown
135	Lake Ann	50w	1927	No Data	Unknown
136	Mountain	50w	1927	No Data	Unknown
137	Rush	50w	1927	No Data	Unknown
138	Silver Lake Basin	50w	1954	**1999**	Stable
139	Sporley	50w	1941	1955	Extripated
140	Avalon Lake	50ac	1939	1990	Declining
141	Long	50ac	1955	**2001**	Stable
142	Muskellunge	50ac	1952	No Data	Unknown
143	**Kimball**	50ag	1984	1984	Unknown

144	Nichols	50ag	1926	1937	Unknown
145	Pickerel	50ag	1952	1984	Stable
146	Lake Angelus	56h	1890	1952	Stable
147	Cass	56h	1890	**2001**	Stable
148	Cedar Island	56h	1971	1994	Unknown
149	North/South Commerce	56h	1890	1968	Unknown
150	Deer	56h	1890	1989	Stable
151	Dunham	56h	1890	1976	Stable
152	Green	56h	1961	1970	Stable
153	Hammond	56h	1957	No Data	Unknown
154	Loon	56h	1944	1972	Stable
155	Maceday	56h	1890	**1996**	Stable
156	Orchard	56h	1890	1976	Stable
157	Oxbow	56h	1970	No Data	Unknown
158	Schoolhouse	56h	1950	No Data	Unknown
159	**Silver**	56h	No Data	**1998**	No Data
160	Townsend	56h	1951	No Data	Unknown

161	Union	56h	1930	**2002**	Stable
162	Upper Pettibone	56h	1945	No Data	Unknown
163	Devoe	50ae	1931	1946	Extripated
164	Grousehaven	50ae	1931	1946	Extripated
165	Higgins	50ae	1887	**1997**	Stable
166	Gulliver	50aa	1940	1983	Stable
167	Indian	50aa	1937	**2001**	Stable
168	Corey	56b	1887	1966	Declining
169	**Fish**	56b	No Data	No Data	No Data
170	Klinger	56b	1887	**1996**	Stable
171	Tamarack	56b	1957	No Data	Unknown
172	**Pepper**	56b	No Data	No Data	No Data
173	Pleasant	56h	1887	1985	Unknown
174	**Prairie River**	56b	No Data	No Data	No Data
175	Thompson	56b	1887	No Data	Unknown
176	Wolf	56b	1927	1945	Extripated
177	Baseline	56h	1890	1943	Extripated

178	Blind	56h	1946	1985	Stable
179	Bruin	56h	1954	1971	Unknown
180	Halfmoon	56h	1942	**2002**	Extripated
181	Pickerel	56h	1948	1982	Declining
182	South	56h	1973	**1998**	Stable