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Drivers And Impacts Of The Invasive Round Goby (neogobius Melanostomus) In Michigan Tributaries To The Great Lakes

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DRIVERS AND IMPACTS OF THE INVASIVE ROUND GOBY (*NEOGOBIUS MELANOSTOMUS*) IN MICHIGAN TRIBUTARIES TO THE GREAT LAKES

by

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DISSERTATION

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INTRODUCTION

Invasive species can impact all levels of biological organization from genes to ecosystems. Invaders can cause declines in native biodiversity (e.g., Molnar et al. 2008) and disrupt community composition (Schultz and Dibble 2012) and ecological processes (Ashton et al. 2005; Vilà et al. 2011). At the population level, invasive species can instigate species declines (e.g., Savidge 1987) and serve as vectors for introduced pathogens and parasites (Telfer and Bown 2012), causing long-term persistence problems for locally adapted species. Nonnative individuals may also directly impose stress on native species via competition and/or predation, thus reducing local fecundity and recruitment (e.g., Gould and Gorchov 2000). Similarly, there can be evolutionary consequences for native species interacting with invasives via hybridization, introgression, and disruption of local adaptation (Mooney and Cleland 2001). Invasive species can also exert control over abiotic aspects of ecosystems (e.g., ecosystem engineering) which can induce trickle-down consequences for native communities (Crooks 2002). Because of the large potential for non-native species to alter ecosystems, investment in invasive species research is among the foremost priorities for biological conservation. Thoroughly documenting the role of an invader is an important step in designing effective prevention and mitigation strategies for invasive species. This is especially important for widespread, rapidly invading species where a full understanding of their ecological niche can help to prioritize conservation resources (Byers et al. 2002).

In the Great Lakes region, nonnative introductions are a significant concern. The Great Lakes are home to a large human population and are widely used for recreation, travel, commercial, and industrial endeavors. With increased human activity, the incidence of nonnative introduction increases (Davidson et al. 2017). With over 180 currently established invaders, coping with nonnative and invasive species is a focus for management in the area (NOAA 2012). One of the more prolific invaders in recent years has been the round goby (*Neogobius melanostomus*). In the last thirty years, round goby has spread across the basin, occupying all five Great Lakes, and is currently undergoing secondary invasion to tributaries and inland lakes (Campbell and Tiegs 2012). While understanding the nature of this invasion has been a research priority in the Great Lakes, most of the current knowledge about round goby in North America was gained directly from lake populations. While this research provides a valuable background upon which to base predictions and potential management strategies, application of knowledge gained from one system to another is not always straightforward. Understanding the context of the current secondary spread of round goby across the Great Lakes basin would provide the best opportunity to curtail consequences on native species and ecosystems. Further, the ongoing invasion of round goby provides opportunity to test hypotheses about invasion dynamics in general.

In this body of work, I investigate the nature of round goby secondary spread and answer some key questions about the impact of this invasion in Great Lakes tributaries. In Chapter 1, I identify the specific consequences incurred by a native competitor along the round goby invasion front. I also address how stream quality and environmental context impact this relationship. In Chapter 2, I propose a means for increasing monitoring efforts for degradation of site quality, and potential methods for early identification of invaders by incorporating citizen science into traditional research and monitoring efforts. In Chapter 3, I combine physical, biological, chemical, and land use data to develop a model which identifies the environmental characteristics common among areas that have been invaded by round goby and host large, sustained populations. This work contributes to the growing understanding of round goby invasion in North America and identifies some key relationships between environmental context and invasion success.

CHAPTER 1 IMPACTS OF THE INVASIVE ROUND GOBY (*NEOGOBIUS MELANOSTOMUS*) ON NATIVE MICHIGAN STREAM FISHES

Introduction

The round goby (*Neogobius melanostomus*) has become one of the most rapidly spreading invaders in the Laurentian Great Lakes since its introduction to North America around 1990 (Kornis et al. 2012). The round goby is a benthic perciform fish native to the Ponto-Caspian region of central Europe (Jude et al. 1992). Since its initial introduction via ballast water exchange, it has established populations in all five Great Lakes, which now serve as points of propagule pressure for current invasion fronts to inland waters. Some tributaries and inland lakes serve as suitable habitat, facilitating the current spread of round goby across the basin (e.g., Campbell and Tiegs 2012). For example, round goby has recently experienced range expansion associated with human transport, likely bait bucket transfer (Johansson et al. 2018)

Because round goby is such a prolific invader, there is great concern regarding the impact of round goby on native ecosystems. In its introduced range, the round goby is linked to population declines of native fishes including economically important game species and species of conservation interest including lake trout (*Salvelinus namaycush*), lake sturgeon (*Acipenser fulvescens*), walleye (*Sander vitreus*), and smallmouth bass (*Micropterus dolomieu*), largely due to egg predation (Kornis et al. 2012). The round goby is similarly linked to decreases in native benthic fish which are the likely competitors in invaded streams including a suite of percid species and mottled sculpin (*Cottus bairdii*) (French and Jude 2001; Lauer et al. 2004; Poos et al. 2010; Burkett and Jude 2015). However, in some systems, round goby has established with no apparent negative consequences for native fish abundances or assemblage composition thus far (Riley et al. 2008; Kornis et al. 2013). In fact, the round goby has been credited as a novel, but significant component of lake food webs by supplementing lake trout diets where alewife populations have been in decline (Colborne et al. 2016).

In addition to impacts on abundance and richness, round goby may have species-specific impacts on native fish physiology and behavior. Balshine et al. (2005) found that round goby outcompeted native logperch (*Percina caprodes*) for habitat through aggressive territorial behaviors in laboratory studies. Similar aggressive interactions were observed with mottled sculpin (Dubs and Corkum 1996), demonstrating potential problems for spatial displacement of native competitors. The high densities typical of round goby populations are also problematic for native recruitment due to egg and larval fish predation (Chotkowski and Marsden 1999). In streams specifically, the round goby has induced shifts in diet composition in native benthic species due to competition for resources (Stauffer et al. 2016). General alteration of food web structure may be one of the most significant consequences of round goby invasion because of the variety of new energetic pathways filled by round goby. Round goby serve as both novel competitors and prey items for fishes in the Great Lakes basin (Steinhart et al. 2004; Colborne et al. 2016), in addition to enhancing energetic pathways from low to high trophic levels via consumption of dreissenid mussels (Johnson et al. 2005).

While much is known about round goby invasion in North America, conflicting results about its impact have made it difficult to assess the round goby's role in ecosystem alteration. Given the increasing spread of round goby and the uncertainty regarding its impact, particularly in inland waters, I investigated the role of round goby in stream ecosystems. Specifically, I examined changes in fish assemblages associated with round goby invasion and competition with a native benthic species, the Johnny darter (*Etheostoma nigrum*). The Johnny darter is an ideal candidate for this comparison due to its wide distribution, and prior evidence of negative interactions with round goby in other systems (Lauer et al. 2004; Burkett and Jude 2015). I hypothesized that the round goby occupies a similar ecological niche as native benthic fishes and is thus associated with a decrease in their abundances and diversity. Further, due to its aggressive nest defense and territorial behaviors I also expected a shift of feeding strategy and reproductive timing in native competitors in response to round goby population growth. I investigated this relationship in seven Great Lakes tributaries over the course of three years to provide context for the impact of round goby on native species spanning the period during which invasion occurred.

Methods

Field sampling

Five watersheds were sampled in 2015 (Au Sable, Rifle, Muskegon, Rouge, and Clinton) to address the impact of round goby in Great Lakes tributary systems; in 2016 two additional watersheds (Ocqueoc and Stony Creek) were added for a total of seven sampled in 2016 and 2017 (Figure 1). Sampling occurred during spawning season in the spring of each year. Spawning season corresponds to adequate increase and stability of water temperature for stream fish. As a result, there was some variation in calendar date of sampling events among years. In general, sampling events began when the southern-most river had a steady temperature of 10-15°C to coincide with the beginning of spawning season for Johnny darter (late April/early May; Speare 1965). Watersheds were generally sampled from lower to higher latitudes to minimize the effects of temperature variation between watersheds. Three sites were sampled along the main stem of each river (Appendix A: Table S1). Sites were chosen based on known distribution of round goby such that sites with and without existing round goby populations were included. Sites were on average 21.14±16.20 river km apart and 41.07±28.94 river km upstream from the mouth, with the downstream-most site occurring on average 21.22±12.91 river km upstream from the mouth.

Distances varied according to size of the watershed but were chosen to represent distinct locations along an upstream to downstream gradient.

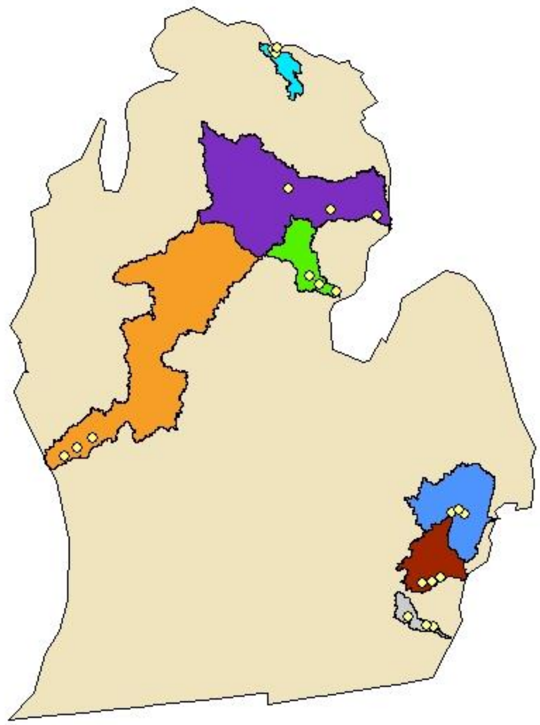


Figure 1. Seven watersheds in the lower peninsula of Michigan sampled from 2015 to 2017. Clockwise from left: Muskegon (orange), Ocqueoc (light blue), Au Sable (purple), Rifle (green), Clinton (blue), Rouge (red), and Stony Creek (grey). Sample sites are indicated by yellow points.

The fish assemblage at each site was sampled once per year using 3x1.5m nylon mesh seines (3.18mm mesh) for approximately one hour, which allowed for all available habitat to be adequately sampled. The reach length varied based on the morphometry of the site and was 282.81±99.26 meters on average. All fish were identified and enumerated prior to release. A subset of individuals was kept, serving as voucher specimens to use in further analyses in the lab. These individuals were euthanized in MS222 (tricaine methanesulfonate) and initially fixed in 10% formalin. Fish were kept in formalin for 1-2 weeks depending on body size, and then gradually transferred to 70% ethanol for final storage. Samples of basal carbon resources (e.g., algae, leaves – up to six samples per site) were collected to serve as reference material for the basal resources of the food web. These samples were hand-collected from the stream benthos and riparian area at each site, and frozen upon return to the laboratory.

Sample processing

Three to five round goby and Johnny darter voucher specimens were dissected for gut content and gonad analyses per site (Table 1). A wet weight was taken for each individual by blotting excess fluid with a paper towel and recording initial body weight in grams (±0.1mg). The size range for each species collected was estimated by measuring voucher fish in the laboratory. Abdominal organs (inside the peritoneum) were removed and an eviscerated body weight was recorded.

Table 1. Sample summary of individual fish dissected for gonad and gut content analyses. Number of samples per tissue was evenly distributed across samples sites where possible. Stony Creek and the Ocqueoc River were not sampled in 2015.

		Individuals per year		
River	Species	2015	2016	2017
Ocqueoc	Round goby	NA	0	0
	Johnny darter	NA	6	4
Au Sable	Round goby	0	0	0
	Johnny darter	3	3	3
Rifle	Round goby	0	1	2
	Johnny darter	3	3	2
Muskegon	Round goby	5	7	7
	Johnny darter	9	8	5
Clinton	Round goby	2	11	4
	Johnny darter	10	7	5
Rouge	Round goby	9	13	21
	Johnny darter	4	7	8
Stony Creek	Round goby	NA	3	3
	Johnny darter	NA	3	0

Gut content analysis included the entire gut tract from esophagus to anus. The gut tract was isolated from other organs and opened to reveal contents. The contents were identified to the lowest practical taxonomic unit (largely to family for invertebrates) and enumerated. Contents were also quantified by surface area over a 1mm grid to provide an estimate of proportional composition (Krabbenhoft et al. 2017). Items not appropriate for simple enumeration were quantified only by surface area (e.g., detritus, sand).

Gonads were blotted to remove excess fluid and weighed in grams. The gonadosomatic index (GSI) was calculated for each individual:

$$GSI = \frac{G}{W}$$

where G is the weight of both gonads, and W is the eviscerated body weight. Gonad weight is standardized by eviscerated body weight to account for differences in body size.

Up to five fish per species per site (where available) were analyzed for stable isotope (nitrogen and carbon) composition (Table 2). For all fish, the skin and scales were removed, and muscle tissue was taken from the right caudal peduncle. Basal carbon resources were assessed using up to three instream resource samples (e.g., algae, macrophytes) and three allochthonous samples (e.g., leaves, grass) per site (Table 2). Invertebrates were manually removed from frozen plant and algal tissues under a microscope and remaining sample tissue was immediately transferred to an oven. All tissues were dried at 60°C for 36-48 hours. No corrections were made for lipid content of any tissue. An average of 1.12±0.31 mg of dry fish tissue was ground to a fine powder and packed in 3.5 x 5mm tin (Sn) capsules. Plant tissues were similarly processed, packed in 5 x 7 mm tin capsules, and weighed to 3.39±0.42 mg.

Stable isotope analysis was conducted at the University of California, Davis Stable Isotope Facility. Samples were analyzed for ^{13}C and ^{15}N isotopes using a PDZ Europa ANCA-GSL elemental analyzer interfaced to a PDZ Europa 20-20 isotope ratio mass spectrometer (Sercon Ltd., Cheshire, UK). Elemental carbon and nitrogen composition are reported in parts per thousand (‰) or in delta (δ) notation (Fry 2006) relative to international standards (Vienna PeeDee Belemnite

Table 2. Sample summary for the stable isotope analysis. Number of samples per river is indicated for each type of tissue analyzed for each of the three years of sampling. Number of samples per tissue was evenly distributed across samples sites where possible. Stony Creek and the Ocqueoc River were not sampled in 2015.

		Samples per year		
River	Taxon	2015	2016	2017
Au Sable	Johnny darter	6	8	3
	Round goby	3	3	3
	algae	4	1	0
	grass	1	2	1
	leaf	5	4	5
	macrophyte	2	4	6
Clinton	Johnny darter	13	7	6
	Round goby	6	9	9
	algae	6	8	4
	grass	0	2	1
	leaf	9	5	5
	macrophyte	0	0	2
Muskegon	Johnny darter	15	12	5
	Round goby	7	11	11
	algae	7	4	3
	grass	3	0	3
	leaf	6	6	5
	macrophyte	2	8	6
Ocqueoc	Johnny darter	NA	10	4
	Round goby	NA	0	3
	algae	NA	1	2
	grass	NA	3	2
	leaf	NA	3	4
	macrophyte	NA	5	2
Rifle	Johnny darter	5	4	5
	Round goby	0	1	2
	algae	0	0	2
	grass	1	0	1
	leaf	2	0	2
	macrophyte	3	0	1
Rouge	Johnny darter	8	7	8
	Round goby	11	15	21
	algae	9	9	9
	grass	0	0	1
	leaf	9	9	8
Stony Creek	Johnny darter	NA	7	2
	Round goby	NA	8	8
	algae	NA	1	4
	grass	NA	0	1
	leaf	NA	6	5

for carbon and air for nitrogen). Corrections were applied to each batch of samples based on interspersed replicates of two laboratory standards. Reproducibility was within 0.2‰ for carbon and 0.3‰ for nitrogen.

Data analysis

Fish assemblages were analyzed using nonmetric multidimensional scaling (NMDS) plots based on Bray-Curtis similarity (using R add-on package, ‘vegan’ [Oksanen et al. 2017]) to visualize similarities between proportional abundances of species at sites and rivers over time. Native assemblage diversity was calculated using Chao and Shen’s (2003) adjusted measure of Shannon’s diversity index. This metric corrects traditional Shannon diversity by using maximum likelihood to assess the probability of discovery for individual species. Diversity was then compared to proportional round goby abundance using a Kendall correlation coefficient (normality of residuals from linear regression was violated, so a non-parametric method was chosen). A principal component analysis (PCA) was used to identify the common factors in dissimilarity of fish assemblages. Round goby influence on overall fish assemblage composition was further analyzed using multinomial logistic regression. For both PCA and multinomial regression, a subset of fish species (where at least one species was represented by only one or two individuals throughout the study, or else were hybrids) were pooled to the genus level to aid in dimensionality reduction for analysis. This resulted in the binning of six species groups, largely to accommodate for unidentifiable juveniles (which were combined with the numerically abundant species in the appropriate taxon group – e.g., a single ‘unidentified lamprey ammocoete’ was combined with American brook lamprey (*Lethenteron appendix*)) (Appendix A: Table S2).

Proportion of Johnny darter relative to round goby in the fish assemblage was assessed using a repeated measures correlation procedure from the R package ‘rmcorr’ (Bakdash and Marusich 2017). The procedure is based on analysis of covariance (ANCOVA) methods altered to accommodate a continuous independent variable, which is influenced by an ordinal factor (in this case, sample year). Bootstrapping was done using 1000 resampling events to make inferences about this relationship on a larger scale. This method incorporates the importance of time since invasion relative to the relationship between species abundance, such that the change in Johnny darter density as round goby became more abundant in a subset of sites can be identified.

Log transformed (log[x+1]) reproductive investment (GSI) was compared between species for each gender using a two-way analysis of variance (ANOVA) with rivers as a blocking factor to identify differences in timing of reproduction. Differences for each species among rivers were identified using one-way ANOVA. Mean Johnny darter log GSI was further compared to round goby density using independent two-group t-tests to determine the impact of potential nest site competition with round goby.

Round goby and Johnny darter gut contents were compared using permutational multivariate analysis of variance (PERMANOVA – Anderson 2001) using the 'vegan' R package (Oksanen et al. 2017) with rivers and sites as blocking factors based on Bray-Curtis dissimilarity matrices and using 999 iterations of the data. Johnny darter gut contents were further compared among sites relative to round goby density. Shannon diversity of darter gut contents was assessed for correlation with round goby proportional abundance using Pearson Correlation. Gut diversity was also compared between species using the Kruskal-Wallis test (due to non-normality of distribution of darter diversity data). The total number of items consumed by Johnny darter was also compared to round goby proportional abundance using Pearson Correlation.

Isotopic signatures of carbon and nitrogen were examined for overlap between species and among sites in bivariate isotope space using the analytical hypothesis tests outlined by Turner et al. (2010a) based on Layman et al. (2007). This method uses a permutation procedure (999 iterations) to determine the overlap of groups in bivariate isotope space. Three metrics were calculated to determine overlap between species and sites: the Euclidean distance between group centroids (MD), the spread of neighboring points (MNN), and the mean distance to the group centroid from its surrounding points (MDC).

Trophic structure was inferred using the 'siar' statistical package in R for analyzing organism isotopes (Parnell and Jackson 2013). Stable isotope data (C and N) from basal resources (algae, leaves, grass, and macrophytes) were used in a Markov Chain Monte Carlo (MCMC) procedure to partition contribution from each source to the diet of the fishes sampled from the same site (Phillips et al. 2005). This analysis allowed for the comparison of goby and darter isotopic values between sites, despite differences in basal resource signatures. Trophic position was calculated for individual fish using a standardization procedure, which accounts for baseline values and trophic enrichment factors (Vander Zanden and Rasmussen 1999; Mercado-Silva et al. 2008):

$$TP_{consumer} = \frac{\delta^{15}N_{consumer} - \delta^{15}N_{baseline}}{3.4} + 2$$

where TP is the trophic position, and the $\delta^{15}N$ for the consumer and baseline samples are standardized by an established trophic enrichment factor of 3.4‰ per trophic level (Post 2002). Trophic position for darters was compared between sites relative to proportional abundance of round goby using two sample t-tests.

All analyses were conducted in the statistical software package, R (R Core Team, 2016). Add-on packages were used as indicated above.

Results

Fish Surveys

Sampling occurred between April 15th and August 11th each year. The sampling periods encompassed a mean 92.33 days but were longer in 2016 and 2017 to accommodate the additional two watersheds sampled. A total of 61 fish species were captured totaling 4,663 individuals encountered over the three-year study period. A mean (±standard error) 83.32±53.5 individual fish were captured at each site per year. The native fish assemblage was largely composed of individuals from the families Cyprinidae, Percidae, Centrarchidae, and Catostomidae with a smaller percentage composed of eight additional families (Appendix A: Figure S1). Average Shannon's diversity of native fish assemblages decreased from 1.79 in 2015 to 1.67 in 2017, with a mean decrease of 0.06 per site. However, when round goby was included in this assessment (to address the impact of invasion on diversity), the decrease in diversity more than doubled to 0.15 per site on average, suggesting invasion contributed to a decline in assemblage diversity, but was not the only factor. Overall, there was a declining trend in Shannon's diversity of native species at each site but was not significantly correlated with round goby proportional abundance ($P = 0.051$; $T_b = -0.183$).

Round goby ranged in abundance from 0 to 93 individuals per site and comprised an average of 15% of the fish assemblage at sites where it was present (11% average over all sites). There was a general increase in both abundance of round goby and the number of sites at which it was present over time. In 2015, round goby was found at 8 of 15 sites. In 2016 and 2017, they were found at 16 of 21 sites. The addition of round goby to a site's assemblage from an initial abundance of zero was interpreted as the initial invasion of the species to this site (the invasion front); this occurred in five of seven rivers (Ocqueoc, Rifle, Muskegon, Rouge, and Stony Creek).

Non-metric multidimensional scaling of fish assemblages at each site through time (Figure 2) showed variation in fish species abundances and differences between sites (within and between rivers). Variation of fish assemblage within river was also highly variable. Sites within the Muskegon River, Rouge River, and Stony Creek largely clustered in multidimensional space suggesting relatively minimal differences along an upstream to downstream gradient. The Ocqueoc, Au Sable, Clinton, and Rifle had much greater variation among sites within each watershed. The Au Sable and Rifle Rivers each had one site that was highly variable over time (the upstream site in the Au Sable, the midstream site in the Rifle). Round goby proportional abundance had a large contribution to the first principal component in a PCA of fish assemblages (Figure 2). Similarly, the multinomial logistic regression identified significant differences in fish assemblage relative to round goby abundance. Fifteen species had a significant response to the abundance of round goby across all sampling sites and times, including western blacknose dace (*Rhinichthys obtusus*; $P < 0.001$), logperch (*Percina caprodes*; $P < 0.001$), blacknose shiner (*Notropis*

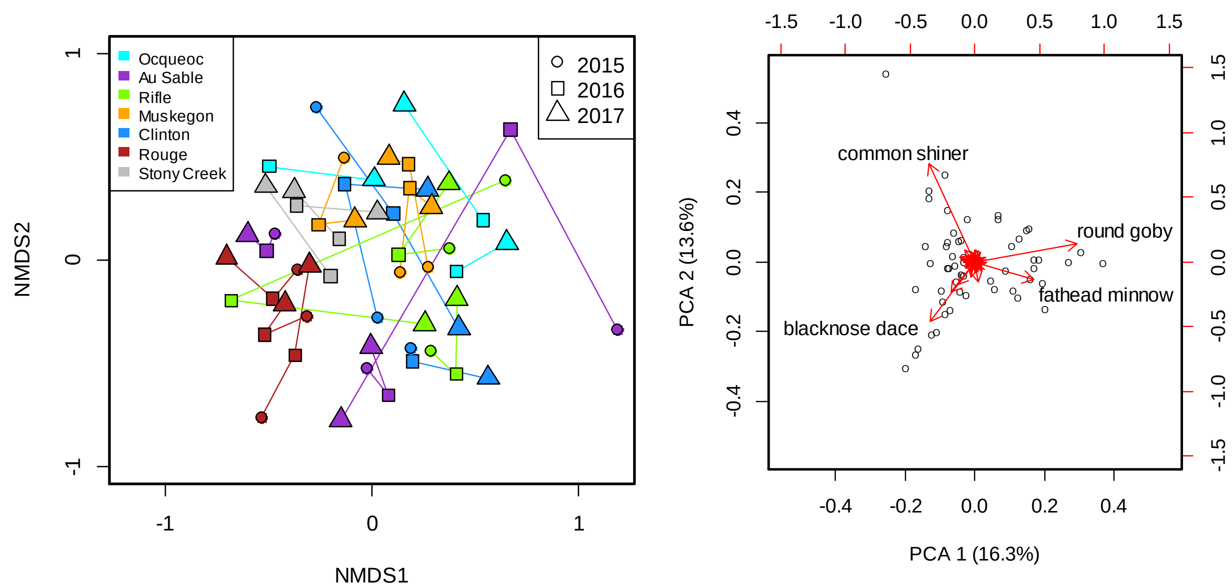


Figure 2. (Left) Nonmetric multidimensional scaling (NMDS; stress= 0.274) of fish assemblage data (proportional abundance). Trajectory lines demonstrate changes in fish assemblage at a single site through time (as points become larger). (Right) Principal components analysis (PCA) of fish assemblages (all years combined). Standard deviation of principal components was 0.18 for the first component and 0.16 for the second component (total variation explained = 30%).

heterolepis; $P < 0.001$), and fathead minnow (*Pimephales promelas*; $P < 0.001$). Johnny darter proportional abundace did not correspond to the proportion of round goby in the fish assemblage ($P = 0.214$).

Reproductive Investment

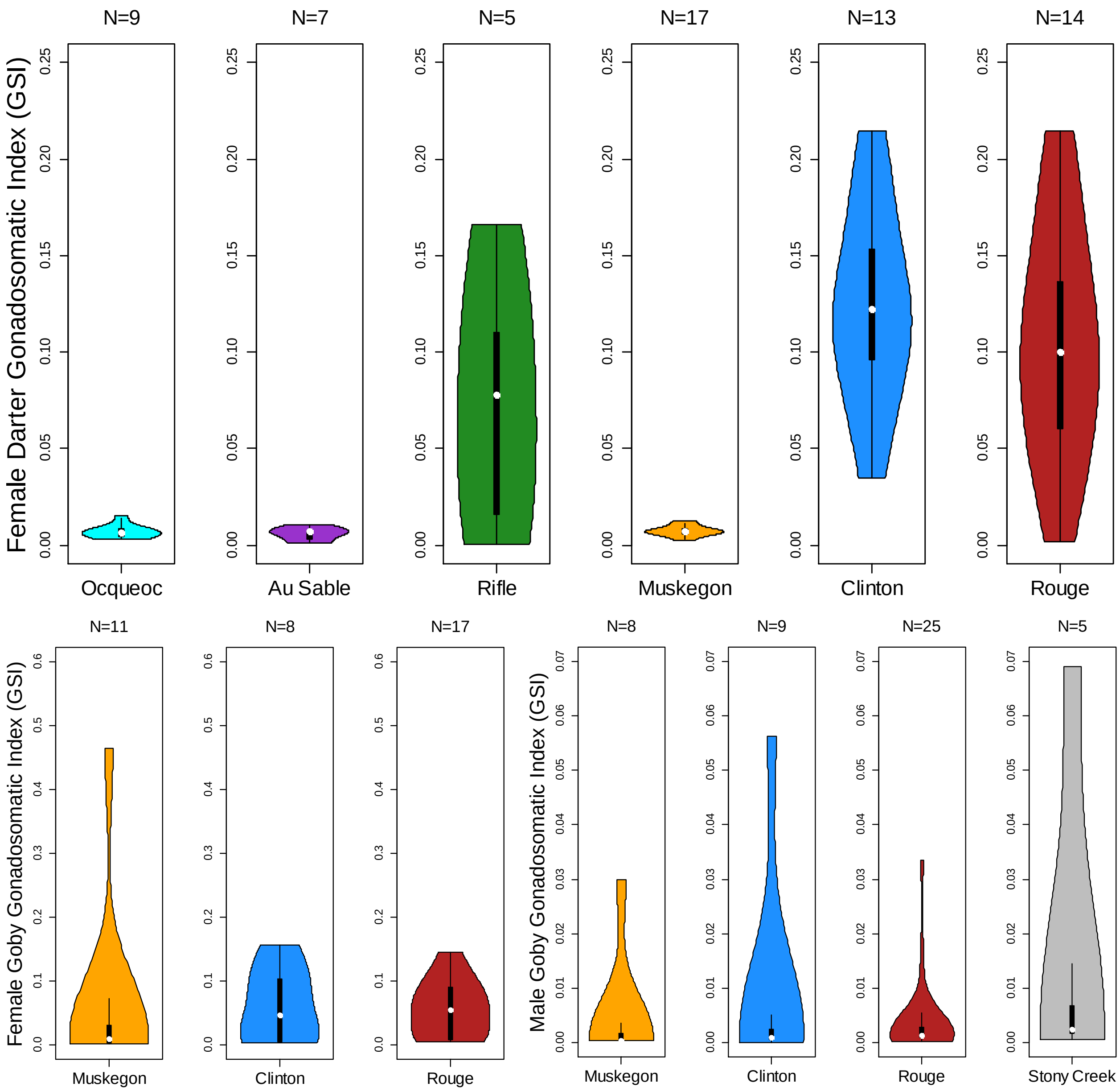


Figure 3. Female GSI for Johnny darter (top) and round goby (bottom – females left, males right). Violin plots represent the median GSI (white dot), the interquartile range (thick black line), the 95% confidence interval (thin black line), and the distribution of the data (width of the shape). All GSI values were transformed via log(x+1) transformations. Rivers with fewer than three individuals per sex per species were excluded (thus there are no plots for male darters).

Gonadosomatic index did not differ between round goby and Johnny darter (F = 0.631; *P* = 0.841 for females; F = 0.143; *P* = 0.906 for males), but there was an effect relative to river for females (F = 5.39, *P* < 0.001; F = 0.890, *P* = 0.508 for males). Johnny darter GSI was found to be highest in the Clinton, Rouge, and Rifle river watersheds for females (Figure 3; F = 17.55, *P* < 0.001 for females; F = 6.03, *P* = 0 .002 for males). This differed from round goby GSI patterns which were relatively consistent across watersheds (F = 0.259, *P* = 0.902 for females; F = 0.997, *P* = 0.420 for males). For female darters, GSI at sites where round goby were up to 10% of the fish assemblage was significantly greater than at sites where round goby were absent (t = 0.347; *P* = 0.035; Figure 4). There were four individuals with anomalously high GSI values in this group (outliers), all of which were sampled from the upstream-most site in the Clinton. For males, GSI values were highest at sites where round goby composed between 0 and 10% of the fish assemblage (t = -3.401; *P* = 0.004 for comparison with goby-absent sites; t = 2.538; *P* = 0.026 for comparison with goby >10% sites).

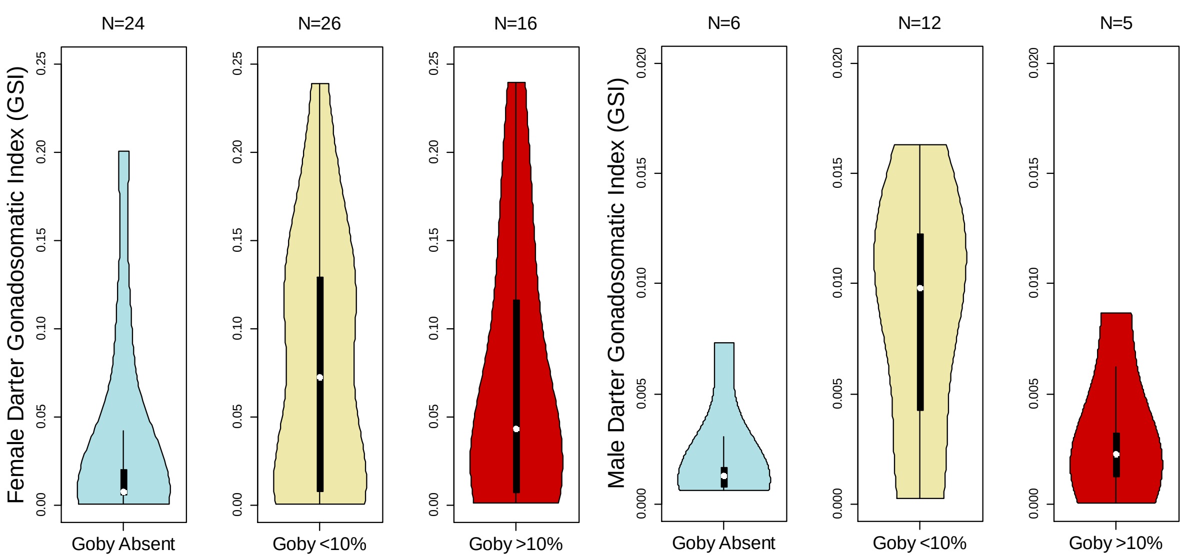


Figure 4. Female (left) and male (right) darter GSI relative to round goby percent abundance at sites where individuals were sampled.

Gut Contents

Round goby and Johnny darter gut contents differed ($P = 0.001$, species $R^2 = 10.2\%$; Figure 5), and the relationship between species varied relative to sampling year ($P = 0.001$, species:year $R^2 = 9.0\%$). Johnny darter gut contents varied relative to the proportional abundance of round goby

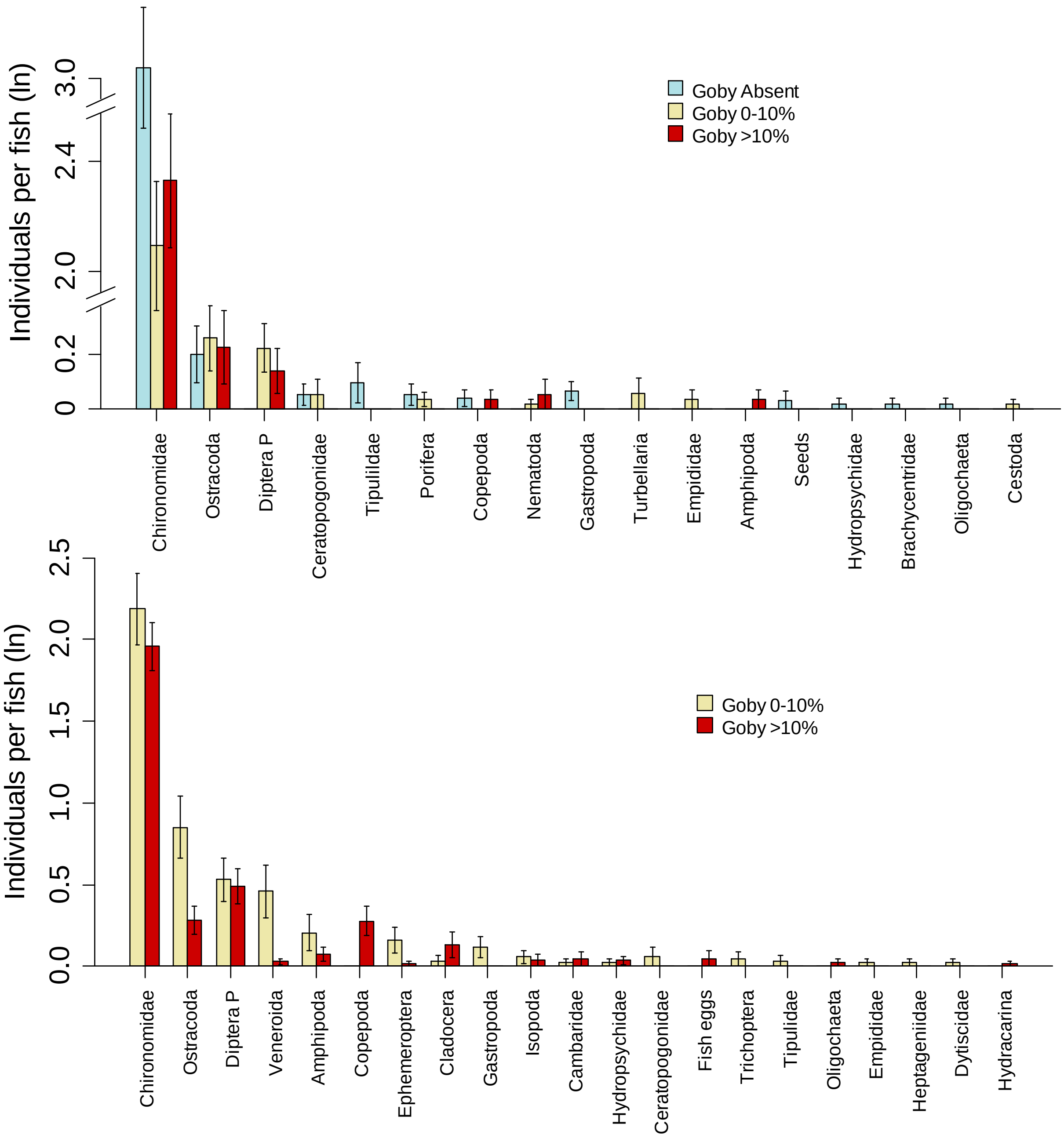


Figure 5. Gut contents of Johnny darter (top) and round goby (bottom) (natural log of item abundance). Bar colors correspond to round goby proportional abundance of the fish assemblage at the site where individual fish were collected. Insect diet items are larvae or nymphs unless otherwise specified (P = pupae). Error bars indicate standard error.

at a particular site (P = 0.001, goby abundance R^2 = 37.0%; Figure 5). Larval midges were the most abundant diet item regardless of round goby presence or abundance. However, as round goby increased in abundance, the diversity of the items in darter diets decreased (PCC: -0.262), largely shifting to a higher proportion of larval midges (Chironomidae). Further, the total number of items per individual (and gut fullness) decreased as round goby became more abundant (PCC: -0.189). Round goby diet reflected a similar importance of larval midges in the diet but contained a greater diversity of diet items overall than did Johnny darter (χ^2 = 67.23, P < 0.001; Figure 5). There was a similar, but smaller, decrease in diet diversity for round goby as they increased in density (PCC: -0.177).

Stable Isotopes

Carbon and nitrogen signatures of both Johnny darter and round goby varied between rivers along a gradient roughly corresponding to urban population density (Figure 6; see also watershed land use data in Chapter 3 for quantification of 'urban' and other land uses). Urban watersheds like the Rouge and Clinton (which encompass the Detroit metro area) had higher signatures in carbon and nitrogen than did relatively low population density, rural watersheds in northern Michigan (e.g., the Ocqueoc and Au Sable). The Muskegon river and Stony Creek were intermediate in carbon signatures but had the highest nitrogen signatures, likely due to the higher proportion of agricultural land use (for more details see Chapter 3) and associated nitrogen-rich runoff from manure and fertilizers (Derse et al. 2007).

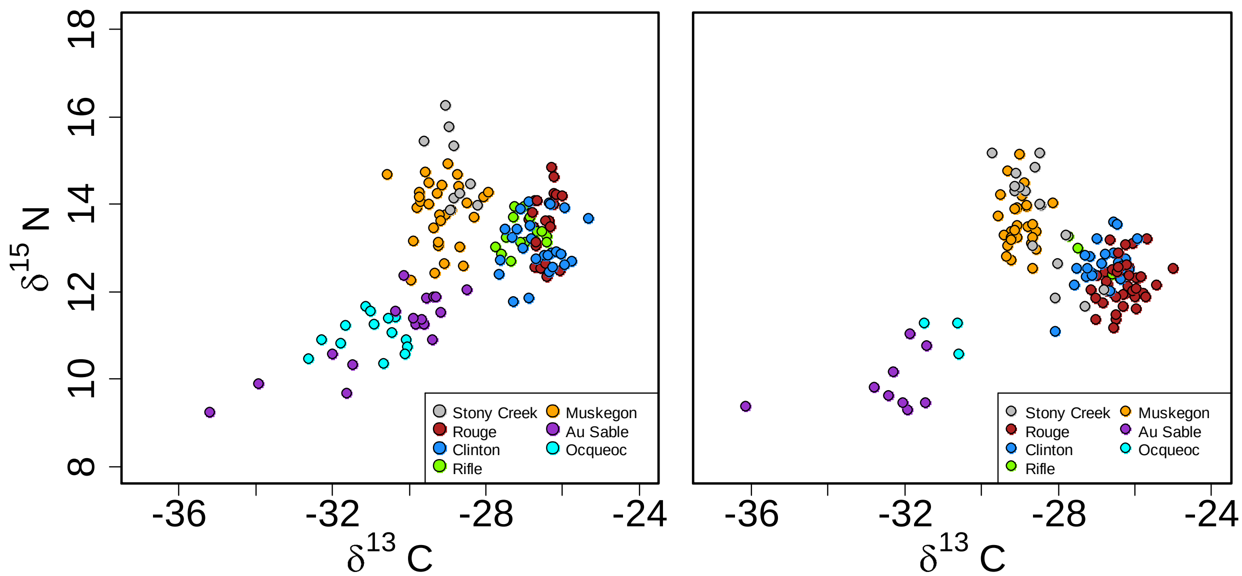


Figure 6. Stable isotope (carbon and nitrogen- uncorrected) values for all Johnny darters (left) and Round goby (right). Coloration corresponds to watershed.

Differences between round goby and Johnny darter isotopic signatures were evident in four of seven rivers (MD - Table 3). There was also less overlap among individuals for Johnny darters than round goby in the Muskegon river (MDC – Table 3). Johnny darters in the Rouge river also had a significantly higher eccentricity (ECC – Table 3), occupying a greater range of nitrogen values than round goby. This finding was similar to results from the Trophic Position estimation where differences in basal resources were taken into account (Appendix A: Figure S2). Johnny darters increased in trophic position as round goby became more abundant in the fish assemblage

Table 3. Statistical summary of differences between Johnny darter and round goby isotopic signatures for each river (sites and years combined). All four isotope metrics are reported: mean distance between groups (MD), mean distance to group centroid from surrounding points (MDC), mean distance to nearest neighboring point within group (MNN), and eccentricity of group points (ECC). Significant differences appear in bold.

River	MD	MDC	MNN	ECC
Ocqueoc	0.984	0.423	0.546	0.926
Au Sable	**0.001**	0.494	0.488	0.238
Rifle	0.178	0.821	0.3005	0.386
Muskegon	0.285	**0.028**	0.128	0.109
Clinton	**0.012**	0.078	0.961	0.168
Rouge	**0.001**	0.151	0.921	**0.018**
Stony Creek	**0.023**	0.261	0.997	0.402

(Figure 7). When round goby composed greater than 10% of the community, trophic position was significantly greater than when goby was absent (t= -3.05; P = 0.003) or when they were in relatively low abundance (t = -2.50; P = 0.016).

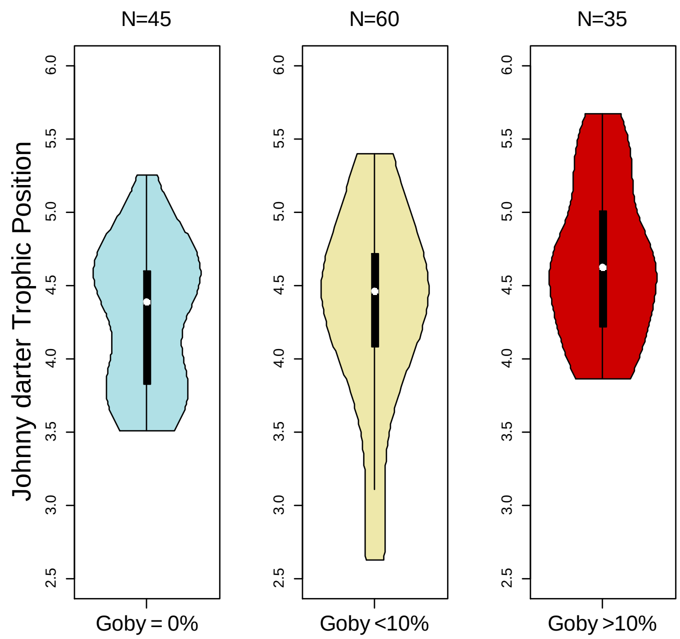


Figure 7. Estimated trophic position of Johnny darters relative to round goby percent abundance in the fish assemblage.

Discussion

The impact of an invasive species on native species and ecosystems can be difficult to identify because system-specific attributes may make generalizations difficult. While drawing from a large body of work on the round goby invasion in the Laurentian Great Lakes is beneficial, the role of round goby in lotic systems remains unclear because of conflicting data from multiple studies. Here I executed a three-year study on round goby populations and the corresponding response by the native Johnny darter to determine the specific consequences across a broad gradient of watershed types. Specifically, I addressed the direct and indirect impacts observed on the native fish assembly and the reproductive and trophic changes for Johnny darter in response to round goby invasion.

Fish Assembly

Round goby was associated with a marginal decrease in fish diversity over time. Such a common phenomenon (xenodiversity; e.g., Orlova 2006) that underscores the importance of assessing nonnative contribution to community composition and traditional metrics, like diversity, when assessing the state of a community. Although an increase in goby abundance was observed over time, Johnny darter abundance was not affected by the proportion of round goby in the assemblage in all rivers. This may suggest a lag time between initial invasion and observable consequences in the fish assemblage (Crooks et al. 1999). For many of the sites, invasion only began during this three-year study. Ongoing, consistent monitoring may detect consequences as round goby undergoes several seasons of reproduction and becomes fully established in the community.

Round goby abundance was one of the important factors driving differences in the fish assemblage. This suggests that round goby may induce a response in the assembly of the native community after initial invasion, or else that there are common factors among native assemblages, which contribute to the likelihood of round goby invasion. For example, blacknose dace, blacknose shiner, and logperch are all typically found in habitats with cool, clear waters and gravel substrate (Smith et al. 2010). An inverse relationship between abundance of these species and round goby may indicate a site was less suitable for round goby invasion due to habitat constraints. However, there could have also been a decline in these species in response to round goby invasion due to competition for benthic habitat space, or some other factor. Conversely, fathead minnow were positively correlated with round goby abundance and are known to be broadly tolerant of environmental conditions (Cross 1950; Smale and Rabeni 1995). If there are abiotic factors (e.g., high turbidity and salinity, low flow and dissolved oxygen, etc.) which are similarly common to round goby invasion (Raab et al. 2018) and tolerant species like fathead minnow, we would expect a higher abundance of those species where such environmental conditions are found. Abiotic factors common to areas with high round goby density may ultimately inform the environmental context associated with invasion (see Chapter 3).

Reproductive Investment

While GSI of Johnny darter and round goby did not differ, variation was higher for darters. This response was in part due to increasing round goby abundance, particularly for females. The higher mean and variance of darter GSI in areas where round goby were abundant suggests that timing of reproduction may have shifted in response to round goby presence. This was particularly evident in the urban watersheds, the Clinton and the Rouge, potentially suggesting some interaction between stressors inherent to urban watersheds and the addition of a nonnative competitor (McKinney 2002; see Chapter 3). If, for example, at the time at which these samples were taken, Johnny darter would normally have a lower GSI (as indicated by goby-absent sites), Johnny darter may be shifting reproductive timing to earlier in the year in response to competitive interactions with round goby (Rathcke and Lacy 1985). This shift could have negative consequences for Johnny darter reproduction due to a mismatch of resource availability and early ontogeny, or else greater intraspecific competition for resources among young-of-year (Turner et al. 2010b; Krabbenhoft et al. 2014). Darters also run the risk of hitting a physiological threshold past which they are no longer able to shift reproductive efforts (due to environmental conditions). This being the case, the reproductive season could be shortened, ultimately coming at a net cost to recruitment (Krabbenhoft et al. 2014).

Of note in this study is that round goby GSI values were much lower than those found for round goby in lake systems, where spawning activity did not occur until GSI was 0.8 for females and 0.1 for males (Zeyl et al. 2013). This suggests that the physiology of reproductive investment in streams fundamentally differs from that in lakes. Similar patterns have been observed for body size and shape in streams vs. lakes for many species (e.g., Berner et al. 2008). As such, there may be consequences of applying conclusions based solely on lake populations to those in streams (Mansfield 1984). This underscores the importance of investigating stream ecosystems to address secondary invasion concerns and better understand the process of establishment.

Trophic structure

Diet and stable isotope data suggest a shift in resource acquisition in Johnny darters where round goby were more abundant. Darter diet diversity and fullness also decreased as round goby increased in abundance. This may indicate that in areas where goby remain relatively rare (typically at the front of the invasion), their limited numbers have no substantial impact on resource limitation (Brandner et al. 2013). In contrast, where goby have become highly abundant in the population following initial invasion (up to 60% of the fish assemblage in this study), darters may have shifted their foraging strategy to negate increased competition for resources. The evidence for resource limitation as round goby increase in abundance was supported by the decrease in round goby diet diversity, suggesting this increased pressure on resources could impact the persistence of the invasive populations over the long term (e.g., a ‘boom and bust’ cycle – Simberloff and Gibbons 2004).

Differences in Johnny darter diet were also reflected by an increase in trophic position where round goby were more abundant. This corresponded to the shift in gut contents to a diet favoring larval midges as the primary diet source. The midge family is composed of species from a variety of functional feeding groups, many of which are predators (Merritt et al. 2008). A loss of primary consumer invertebrates in favor of predatory midges in the diet would result in an increase in nitrogen signatures of Johnny darter. This may also correspond to a shift in habitat use among darters as midges as a group are considered a tolerant invertebrate taxon and occupy a wide range of habitats, including those with higher contaminants (Haas et al. 2005), lower oxygen, and higher temperatures (Walshe 1948). While an increase in trophic position of Johnny darters may be a counterintuitive response to increased competition for resources, this response may simply reflect an increase in diet specialization in response to round goby invasion, as seen in other studies where species diversity was increased (Mason et al. 2008). Ultimately, a decrease in feeding generalization could be associated with increased intraspecific competition (Amundsen 1995), energetic consequences (Britt et al. 2006), or a shift in habitat use (Holbrook and Schmitt 1992) but would warrant further study to fully characterize for round goby.

Conclusion

The specifics of a native community response to an invader can vary spatially and temporally and may take entirely different forms based on the temporal, spatial, and anthropogenic factors of a particular system. While many hypotheses exist regarding the factors, which influence this range of responses, it is important to understand the impact nonnative introductions can have in a variety of ecosystems and native communities. Here I have investigated the invasion of round goby to tributaries of the Great Lakes and how the establishment of this invader has affected native competitors. Previous literature on round goby invasion has largely focused on lacustrine environments and has produced evidence of potentially positive (as a predator of invasive dreissenid mussels [Ray and Corkum 1997], and prey of native lake trout [Dietrich et al. 2006]) and negative (via reduction of native benthic competitors [Burkett and Jude 2015]) impacts of round goby on native systems and species. This study provides further evidence of both negative and neutral interactions of round goby with a native competitor as well as behavioral changes in the native as a response to growth in round goby populations. This research thus addresses gaps in knowledge regarding the role of round goby in contributing to community assembly, reproductive strategies of native competitors, and the overall trophic structure in invaded streams.

CHAPTER 2 ASSESSING STREAM QUALITY THROUGH CITIZEN SCIENCE: QUALITATIVE AND QUANTITATIVE DATA REACH SIMILAR CONCLUSIONS

Introduction

Conservation and management benefit from volunteer programs, or citizen science, to monitor a variety of ecosystem types. These organizations take many forms, including enlisting volunteers to assist traditional research, hands-on data collection by volunteers (e.g., Biggs et al. 2015), soliciting information or photos from citizens participating in recreational activities (e.g., Hurlbert and Liang 2012), or data collection where citizens take initiative to contribute data or information (Swanson et al. 2016). To reinforce understanding of the local ecology among citizens, and encourage continued citizen participation, volunteers are typically provided information about the ecosystem of interest and the research goal. As such, these programs have increased in popularity as a way to produce ecological data while simultaneously investing in a more informed public (Dickinson et al. 2012). The most effective programs have small infrastructure costs but maintain active and regular sampling through motivating volunteers, often those living in or near the ecosystem of interest (Cooper et al. 2007).

The benefits of citizen science programs include engaging individuals directly benefitting from ecosystem services provided by the system and providing environmental data that either was previously lacking from that system or enhances existing data sets collected by state and federal agencies, companies, and academics. Long-standing citizen science programs can provide valuable additions to more traditional monitoring and research efforts. Costs associated with these programs are comparatively low when compared to agency or academic research efforts (Bonney et al. 2009; Aceves-Bueno et al. 2015) and have provided an estimated $2.5 billion worth of person-hours to global management and conservation efforts annually (Theobald et al. 2015). While mobilizing a sufficiently large citizen base can be difficult, once established, these programs can address monitoring and management issues at a scale far greater than is possible with local agency or academic endeavors (Cooper et al. 2007). Further, sustained monitoring activity can provide a more robust system for identifying environmental problems including providing early warning systems for nonnative species (Simpson et al. 2009). Soliciting help from the public also represents a positive feedback loop in maintaining programs over the long term such that public interest can support these programs and their conservation goals through legislation (i.e., public ecology [Robertson and Hull 2001]), support of public and private funding, and education of the public in ecological concerns (Bonney et al. 2009).

Despite numerous citizen science efforts and their utility for management decisions, volunteer data remain under-utilized in traditional research. In part, this is due to a lack of trust among researchers about the quality and reliability of data produced by amateur volunteers (Catlin-Groves 2012). One uncertainty associated with these programs is how accurate volunteer-obtained data are, particularly when they may be produced using different sampling methods. Further, the ability to readily incorporate information into databases used by managers and researchers can be problematic (Conrad and Hilchey 2011). Although several case studies demonstrate that citizen science data can achieve similar accuracy and precision as that produced by traditional research (i.e., standardized, quantitative methods typically employed by trained ecologists) (e.g., Darwall and Dulvy 1996), there is still limited use of the data due to lingering stigma. Despite reservations among researchers about the quality of volunteer data, monitoring through citizen science presents an opportunity to fill data gaps where funding or agency resources are limited (Canfield et al. 2002; Delaney et al. 2007).

The value of citizen science has been demonstrated by several studies in which data produced by citizen science was directly compared to that of trained professionals to determine the quality and utility of these programs. Delaney et al. (2007) found that school-aged children could identify crab species with great accuracy (80-95% agreement with professional assessment), producing data that was used to develop a distributional database for additional research. Goffredo et al. (2010) found slightly lower accuracy of citizen data (50-80%) but concluded that the greatest barrier for utilization of this data was not accuracy, but inadequate spatial coverage, suggesting larger investment in citizen science programs would provide an overall benefit to the landscape of research. Further, the quality of citizen science data is greatly improved with increased regularity of participation and supplemental training by professionals (Darwall and Dulvy 1996).

While explicit comparisons of traditional and citizen-produced data are valuable in demonstrating the utility of citizen science, such comparisons are often ecosystem-specific. For example, in the Laurentian Great Lakes, citizen science programs are common, but many evaluations of their quality are of solely terrestrial programs (e.g., butterflies [Matteson et al. 2012]; birds [Vargo et al. 2012]; plants [Crall et al. 2015]). However, in Michigan, state infrastructure supports a thriving citizen science community in aquatic systems through the Michigan Clean Water Corps (MiCorps [Latimore and Steen 2014]). In 2015, MiCorps integrated existing volunteer stream monitoring programs into the statewide Volunteer Stream Monitoring Program. This program allows for standardization of stream monitoring procedures for watershed-specific nonprofit groups across the state. Watershed groups are particularly active in southeastern Michigan (due to high urban population density) where several Great Lakes tributaries hold immense ecological value as nurseries for young-of-year fish and for recreation and economic opportunities. MiCorps citizen monitoring programs are based on sampling macroinvertebrate communities to demonstrate relative stream quality at various sites within a watershed (Nerbonne et al. 2008). Yet, the data collected from these programs are not typically utilized by state agencies or academics for resource management or for evaluating and monitoring ecosystem health.

While rigorous comparative studies that demonstrate the quality and utility of these data for research are becoming more common, comparative data for streams in Michigan are lacking. Southeastern Michigan remains an ideal location for such studies due to the active volunteer stream monitoring programs, the variation in environmental stressors which require monitoring, and the economic benefits from fisheries, municipal water, and recreational activities. Here I provide a site-specific comparison of qualitative invertebrate monitoring data from two volunteer organizations with quantitative data produced by trained stream ecologists in two urban rivers. Through this comparison I determine the reliability of citizen data and identify the similarities with traditionally-produced data. I evaluate the pros and cons of each method as they relate to describing site-specific environmental and water quality conditions and provide suggestions for incorporating these data into larger research frameworks.

Methods

Two sampling methods, one highly quantitative often employed by academic researchers and the second a qualitative method common in volunteer monitoring, were compared among four sites in the Rouge and Clinton watersheds in southeastern Michigan over several years (see Chapter 1 for map of sites). Employing traditional quantitative sampling methods, I visited sites to complement the timing and efforts of qualitative citizen science sampling events for two non-profit groups, the Friends of the Rouge (FOTR) and the Clinton River Watershed Council (CRWC). Both organizations are 501(c)(3) nonprofits that have been serving their communities for 32 and 44 years, respectively. Their missions are to restore, protect, and enhance their respective watersheds by engaging the public and investing in cleanup and conservation efforts within the watersheds. Among their many annual activities are citizen science macroinvertebrate monitoring events. Academic and volunteer sampling typically took place from the last week of April to the first week of May from 2015 to 2017, depending on weather and stream conditions; qualitative sampling occurred within approximately one week of my quantitative sampling (Table 4).

Qualitative Volunteer Assessments

Table 4. Site locations and sample dates for all quantitative academic and qualitative volunteer sampling events. Coordinates according to GCS_WGS_1994. Volunteer sampling for the Rouge and Clinton Rivers is done by the Friends of the Rouge (FOTR) and the Clinton River Watershed Council (CRWC), respectively. 'Academic' refers to my own, quantitative sampling.

River	Site Name	Latitude	Longitude	Academic sampling dates	Volunteer sampling dates
Rouge	Morton Taylor	42°16'58"N	83°27'58"W	26-Apr-15	18-Apr-15
				24-Apr-16	16-Apr-16
	Inkster	42°17'56"N	83°18'24"W	1-May-15	18-Apr-15
				24-Apr-16	16-Apr-16
Clinton	Avon	42°39'53"N	83°09'18"W	14-May-15	2-May-15
				6-May-16	7-May-16
				8-May-17	7-May-17
	Cider Mill	42°40'17"N	83°05'46"W	6-May-16	7-May-16

Subsets of citizen science monitoring data from 2015 to 2017 was obtained from the FOTR and the CRWC. Sampling methods are standardized for all volunteer stream monitoring organizations in Michigan, including these two, through the Michigan Clean Water Corps (MiCorps). Monitoring involves a group of volunteers who are assigned one or more sites with two team leaders per team, at least one of whom has participated as a team leader previously. Team leaders are provided a half day of training with the watershed organization prior to volunteer events. On the day of sampling, team leaders take their team of volunteers to a site identified by the organization. They use a D-frame net (maximum 1mm mesh) to sweep invertebrates from a variety of stream habitat types for at least 30 minutes. Samples are sorted onsite by other volunteers and team leaders utilize keys to identify the taxa present. A subset of invertebrates is preserved in

70% ethanol and returned to the organization to verify identifications. Taxon abundances are recorded as Rare (1-10 individuals sampled) or Common (11+ individuals sampled). A stream quality index (SQI) is calculated for each site based on the abundances of each taxon and their established tolerances of degraded habitats (Latimore 2006). Physiological tolerances are categorized into three groups: Group 1 – Sensitive species; Group 2 – Somewhat-sensitive; and Group 3 – Tolerant (Figure 8). Taxa sensitive to degradation are weighted heavier in the SQI calculation than those of more tolerant groups such that an abundance of 'sensitive' taxa would indicate a relatively high stream quality. The SQI values are further binned to produce a ranking of stream quality (i.e., Poor, Fair, Good, Excellent).

Figure 8. Michigan Clean Water Corps (MiCorps) macroinvertebrate sampling data sheet. Tolerance of each taxon is categorized into three groups and abundance is recorded as either 'Rare' or 'Common'. Abundances of taxa are recorded on the left and scores are tabulated in the box to the right. A sum is produced to provide an overall Stream Quality Index (SQI).

Identification and Enumeration

Use the codes "R" (rare) = 1-10, or "C" (common) = 11 or more when recording the number of individuals in each taxonomic group.

Group 1: Sensitive

____ Caddisfly larvae (Trichoptera) *EXCEPT Net-spinning caddisflies
____ Hellgrammites (Megaloptera)
____ Mayfly nymphs (Ephemeroptera)
____ Gilled (right-handed) snails (Gastropoda)
____ Stonefly nymphs (Plecoptera)
____ Water penny's (Coleoptera)
____ Water snipe fly (Diptera)

Group 2: Somewhat-Sensitive

____ Alderfly larvae (Megaloptera)
____ Beetle adults (Coleoptera)
____ Beetle larvae (Coleoptera)
____ Black fly larvae (Diptera)
____ Clams (Pelecypoda)
____ Crane fly larvae (Diptera)
____ Crayfish
____ Damselfly nymphs (Odonata)
____ Dragonfly nymphs (Odonata)
____ Net-spinning caddisfly larvae (Trichoptera)
____ Scuds (Amphipoda)
____ Sowbugs (Isopoda)

Group 3: Tolerant

____ Aquatic Worms (Oligochaeta)
____ Leeches (Hirudinea)
____ Midge larvae (Chironomidae)
____ Pouch snails (Gastropoda)
____ True bugs (Hemiptera)
____ Other true flies (Diptera)

STREAM QUALITY SCORE
(metric created by MiCorps, www.micorps.net)

Group 1
____ # of R's * 5.0 = ____
____ # of C's * 5.3 = ____
Group 1 Total = ____

Group 2
____ # of R's * 3.0 = ____
____ # of C's * 3.2 = ____
Group 2 Total = ____

Group 3
____ # of R's * 1.1 = ____
____ # of C's * 1.0 = ____
Group 3 Total = ____

Total Stream Quality Score = ______
(Sum of totals for groups 1-3; round to nearest whole number)

Excellent (>48)
Good (34-48)
Fair (19-33)
Poor (<19)

Quantitative Academic Assessments

To compare the qualitative volunteer data with more traditional, research-focused stream ecology methods, we conducted sampling from 2015 to 2017 at sites in the Rouge and Clinton rivers which correspond to volunteer monitoring sites. To produce more quantitative and reproducible samples, invertebrate samples were collected in triplicate at each site with an 860-cm^2 Hess stream bottom sampler (243 μm-mesh). This is a common quantitative sampling technique used in research studies, typically used in riffle habitats because they often harbor high macroinvertebrate diversity (Brooks et al. 2005); targeting the area with the highest diversity ideally provides a conservative estimate of habitat degradation. In the lab, invertebrates preserved in 90% ethanol were removed from substrate, enumerated and identified to the family level (typically order or class for non-insects). Mean abundances for each taxon were calculated from the triplicate samples to represent the assemblage of macroinvertebrates at each site.

Data Analysis

To compare the invertebrate sampling methods, nonparametric estimates of taxon richness were used to determine the actual number of taxa at each site (parametric estimates were deemed inappropriate due to drastic differences in abundance and the high number of rare taxa). Estimates are based on relative abundances of sampled taxa by considering the number of taxa represented by only one or two individuals (Chao1 estimator; Chao et al. 2009). Because data are derived from a mean of three samples, I altered the Chao1 estimator to accommodate non-integers, and extend its range for 'rare' taxa to allow for taxa that had one or two individuals in at least one of the triplicate samples. Thus, the number of taxa in the invertebrate assemblage (S_{est}) was estimated using the following equation from Chao et al. (2009):

$$S_{est} = S_{obs} + f_1^2/(2f_2)$$

where S_{obs} is the total number of taxa observed, f_1 is the number of taxa represented by a single individual in at least one of the triplicate samples (but maintained a mean abundance of less than two), and f_2 is the number of taxa represented by only two individuals in at least one of the replicates (but with a mean abundance of less than three). This method provides an estimate of the actual number of taxa at a site to indicate whether the sampling method was adequate in taxon detection; if S_{est} and S_{obs} are similar, the data represent a thorough sample of the invertebrate assemblage. Estimated richness values were compared to observed richness from both academic and volunteer data using Pearson's chi-squared test.

As an initial look at stream quality, I calculated the Shannon diversity index, taxon richness, and Ephemeroptera, Plecoptera, and Trichoptera (EPT) richness from the academic invertebrate data. Because volunteer invertebrate data are reported only with a 'Rare' or 'Common' designation (not numerical abundances), only richness could be calculated. To compare between data types, I also calculated the MiCorps algorithm for stream quality index (SQI; Fig. 1) for each data type. Quantitative and qualitative data were compared using chi-square or Fisher's exact tests for each site. Pearson's chi-square test was used to compare richness and SQI values between academic and volunteer data, while Fisher's exact test was used to compare the EPT richness (due to the high proportion of values under 5 in this type of data).

Academic and volunteer data were further compared for their ability to detect particular taxa in the invertebrate assemblage. The Jaccard Index of similarity was calculated for each site to evaluate agreement between methods on the invertebrate assemblages produced. Further, each invertebrate taxon was assessed for whether it was detected by both methods, by only the academic assessment, or only the volunteer assessment. McNemar exact tests were used to determine whether the probability of detection for each taxon was the same for each method. Fisher's exact tests were used instead where the sum of the discordant values in contingency tables was less than five. A Bonferroni correction was applied to control the familywise error rate due to multiple comparisons.

All data analysis was completed in the R statistical environment (R Core Team, 2016).

Results

All sites analyzed were of relatively low quality based on the indices examined for the academic assessment (Appendix B: Figure S1). Average diversity was 0.78 ±0.104 and taxon richness was 10.88 ± 1.452. Midges (Chironomidae) were the most common taxon (average of 379 ± 60 individuals per site) and were found in every sample. They composed between 28 and 93 percent of the invertebrate community. Aquatic worms (Oligochaeta) were the next most abundant taxon with 193 ± 91 individuals per site. The maximum EPT richness was three taxa. However,

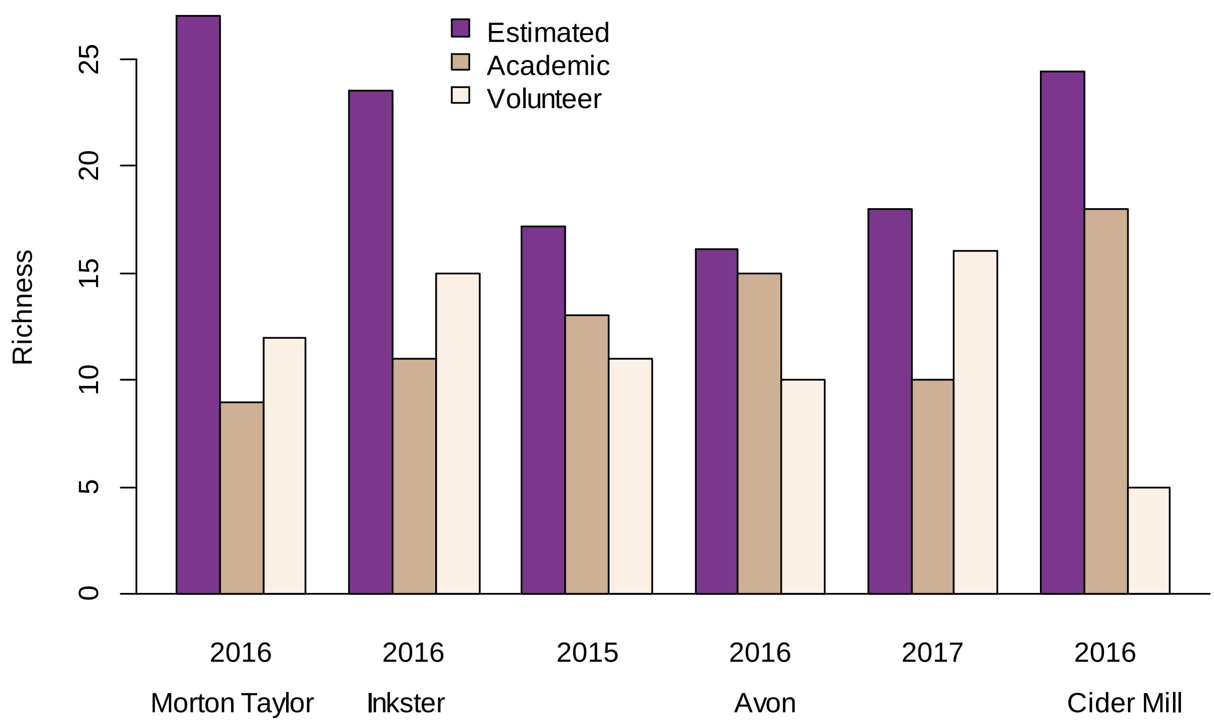


Figure 9. Richness as estimated by the Chao1 estimator which corrects for rare and unsampled taxa, and the observed richness from the academic and volunteer invertebrate samples. Only six sites are present because the Chao1 estimator cannot function with samples where no taxa were represented by two individuals (cannot divide by zero).

net-spinning caddisflies (Hydropsychidae), a relatively tolerant invertebrate, were the most abundant taxon in this group (found at all but one site). The EPT composition was otherwise only made up of occasional small minnow mayflies (Baetidae) or microcaddisflies (Hydroptilidae). No stoneflies (Plecoptera) were identified in any sample.

Estimated sample sizes from the Chao1 estimator (Chao et al. 2009) were only calculable for six of eight sites due to a lack of samples with only two individuals (cannot divide by zero). Based on the estimated richness values, the academic and volunteer assessments of site richness were lower than that of the estimator (Figure 9). Average estimated richness was 8.37 ± 2.26 taxa greater than was observed in academic samples and 9.53 ± 2.40 greater than the volunteer samples. The differences in abundances ranged from two to 18 taxa. However, no significant difference was found between estimated and observed richness values overall (χ^2= 5.26, P = 0.384 for academic data, and χ^2= 7.21, P = 0.205 for volunteer data).

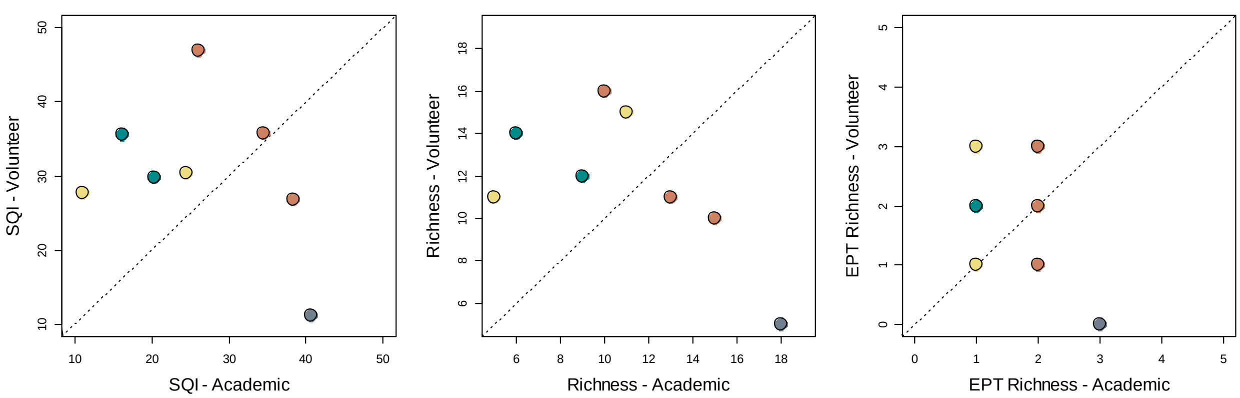


Figure 10. The SQI (left), taxon richness (middle), and EPT richness (right) as determined by the quantitative academic assessment and the qualitative volunteer assessment. The one-to-one lines are shown. If the two assessments came to the same conclusion about the site, all points would lie along the one-to-one lines. Points above the one-to-one line indicate a higher value was determined by the volunteers; points below the line indicate a higher value was determined by the academic assessment.

Comparison of the academic and volunteer assessments yielded generally similar results for each of the metrics compared. The SQI was higher for the volunteer data at six of eight sites, but the difference was not significant (χ^2 = 56; P = 0.229; Figure 10). Richness was similarly high for the volunteer assessments but was not significantly different between assessments (χ^2 = 48; P = 0.243). The EPT richness was the same at two sites and deviated by a single taxon at three other sites (P = 0.914). There was some similarity in which sites differed between academic and volunteer assessments. For example, the Cider Mill site on the Clinton River in 2016 was consistently rated much higher in the academic assessments than the volunteer assessments. This site had the second highest number of individuals sampled for any academic assessment (n=931 individual macroinvertebrates) and had the lowest proportion of the assemblage composed of midges (Chironomidae), the most abundant taxon across all sites.

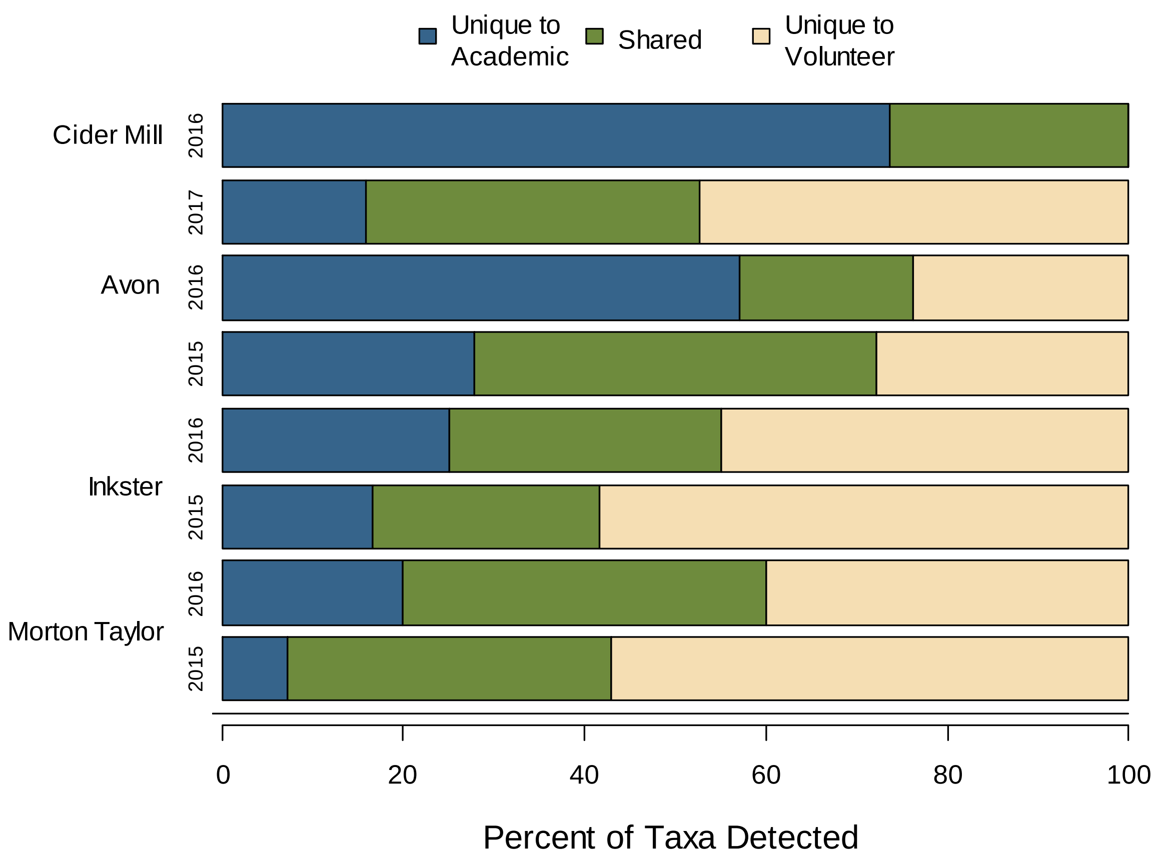


Figure 11. The percent of taxa shared and unique to each sample type. The percent of taxa which were detected by both methods (shared) corresponds to the Jaccard index of similarity.

Despite general similarities in the stream quality metrics measured, the Jaccard index of similarity remained relatively low for each site (Figure 11). Invertebrate assemblages from the academic and volunteer data were 32% similar on average and ranged from 19% to 44% similar. There were some patterns in which invertebrate taxa were detected by each group. Seven taxa were completely unique to the academic assessments across sites, never having been found in the volunteer data (Table 5). Likewise, ten taxa were consistently missed by academic assessments, despite having been found in volunteer data. Midges (Chironomidae) were the only taxon to be consistently detected by both groups. In total, the volunteer method detected 26 taxa while the academic detected 23 taxa. However, no statistical differences in probability of detection were identified in taxon-specific analyses.

Table 5. Taxa completely unique to each type of assessment across all sites. Scientific and common group names are given.

Academic assessment		Volunteer assessment	
Group	Common Name	Group	Common Name
Ceratopogonidae	Biting midge	Aeshnidae	Dragonfly
Collembola	Springtail	Ancylidae	Freshwater snail
Hydra	Freshwater cnidaria	Athericidae	Water snipe fly
Hydracarina	Water mite	Calopterygidae	Damselfly
Limpet	Freshwater snail	Decapoda	Crayfish
Ostracoda	Seed shrimp	Dryopidae	Freshwater beetle
Sciomyzidae	Marsh fly	Heptageniidae	Mayfly
		Turbellaria	Flatworm
		Sphaeriidae	Native bivalve
		Veliidae	True bug

Discussion

Citizen science efforts can provide a valuable contribution to research efforts but are often under-utilized because of lack of certainty about their accuracy. Studies that compare traditional quantitative assessments with citizen volunteer-obtained data allow for inclusion of this resource into traditional research and monitoring endeavors. Here I provide a site-by-site comparison of eight instances of qualitative volunteer monitoring with quantitative academic assessments on the same invertebrate communities.

The SQI, richness, and EPT richness metrics were higher for the volunteer-obtained data. Qualitative methods are known to produce a higher richness value due to their ability to explore multiple microhabitats (Lenat 1988). The larger difference between observed richness and estimated richness in the academic data corroborates a larger underestimate of richness in the academic data. This may highlight a potential weakness in quantitative academic sampling that is addressed by incorporating complementary qualitative sampling. While quantitative methods often used by trained ecologists may allow for increased accuracy and reproducibility, the trade-off is the restrictive nature of the microhabitat sampled. For example, the Hess stream bottom sampler used in this study explicitly targets riffles because they are known to harbor high invertebrate diversity (Brooks et al. 2005). However, in a heterogeneous stream reach, this focus on riffles may fail to identify taxa, which preferentially occupy other microhabitats (Brown and Brussock 1991). Further, limiting the area of the benthos sampled by quantitative methods (i.e., same number of replicates per site) may fail to fully capture the patchiness of invertebrate distribution, even within the same microhabitat type (Downes et al. 1993).

Differences between metrics were further illustrated by the low degree of similarity in the Jaccard comparison. Approximately one third of the invertebrate assemblage composition was shared between sampling types. These similarities were largely due to the most abundant taxa in the assemblage, especially midges. Midges are a highly tolerant invertebrate taxon which are common to a variety of habitats (Pinder 1986). Due to their high abundance and diversity, the likelihood of species discovery for midges is quite high. Conversely, the taxa which were unique to each sampling method highlighted some key differences in the probability of species discovery. Taxa unique to academic sampling were mostly small-bodied invertebrates with slow or limited motility that are typically quite low in abundance. Taxa unique to the volunteer data were largebodied and highly motile organisms. This pattern highlights an important difference in the strategies for sorting and identification. The volunteer method of sorting by eye on-site is more likely to miss small items that do not readily move around and so are more difficult to spot (Nichols and Norris 2006). When invertebrates are preserved and sorted by microscope, these individuals are much more likely to be discovered and accurately identified.

Despite the large proportion of taxa which were uniquely identified by each sampling type, no invertebrate taxon was identified to have a higher probability of detection for one method over the other. The majority of the taxa which were unique to one sampling method were collected in low abundances (i.e., one or two individuals). The probability of under-sampling naturally rare taxa is sufficiently high to skew the data in this manner (MacKenzie et al. 2005). While size variation among invertebrate taxa produced by each method may highlight consistent differences in the detection capabilities for each method, the resulting estimate of the invertebrate assemblage tends to vary largely due to rare taxa. Information on rare taxa can be cumbersome to include in such analyses due to the degree of chance in each sampling event but is ultimately important for accurate environmental assessment (Faith and Norris 1989). These challenges underscore the importance of long-term, repeated monitoring activities which can better inform assemblage dynamics over time.

Importantly, differences in species discovery did not correspond to significant differences in stream quality metrics between the academic and volunteer assessments, highlighting the utility of volunteer data, despite inherent differences in sampling methods. Further, volunteer data consistently discovered invertebrate taxa that were missed in academic sampling. While there are pros and cons to each method, understanding the goals of monitoring can help determine if volunteer data are suitable to incorporate in traditional research efforts. For example, if a detailed accounting of zooplankton and other small-bodied organisms is a priority, volunteer data may not be suitable. However, if the overall goal is to use invertebrate data to monitor changes in stream quality, this study suggests volunteer data are as reliable as quantitatively obtained academic data.

This finding is important for the incorporation of qualitative citizen-produced data into research and monitoring efforts. Particularly in urban centers like the ones in this study, regular monitoring is of critical importance. There has been a multitude of anthropogenic impacts on these rivers in the past (Beam and Braunscheidel 1998; Francis and Haas 2006) and invertebrates can provide an indication of how stream quality is changing over time (Firehock and West 1995). Invertebrate monitoring provides a valuable addition to measurements of contaminants and bacterial levels, like those typically done in urban wastewater effluents. Invertebrates, being a biological indicator, provide a more robust, long-term assessment of stream quality; because they can inhabit the stream for up to several years (Merritt et al. 2008), they allow for a more robust assessment of quality than snap-shot measurements of chemical contaminants. While regular invertebrate monitoring and identification can be expensive and laborious, citizen science assessments substantially reduce the cost and time required to do so (Bonney et al. 2009; Theobald et al. 2015). Citizen monitoring programs also provide regular activity on a variety of locations around a watershed. Such activity can be beneficial for early reporting of illegal dumping or new introductions of nonnative species (Gallo and Waitt 2011).

One caveat to the patterns observed here is that the Rouge and Clinton River watersheds where this study was located are highly urban (see Chapter 3 for more details). While this allows for a large population from which to draw volunteers, it does not allow for a wide range in site quality for these comparisons. As observed here, diversity and richness were consistently low, as is typical of urban watersheds (Moore and Palmer 2005). It is possible that these trends might differ in less disturbed sites. For example, when an invertebrate assemblage favors smaller-bodied, tolerant taxa (typical to urban systems), volunteer data might miss a larger proportion of the taxa present; conversely, in areas where larger, more sensitive taxa occur (e.g., in headwater streams), volunteer data might produce results more in line with traditional monitoring. Further, in more stable sites with reduced incidence of disturbance (as would be expected of more rural streams), microhabitat specialization among invertebrates is expected to increase (Death 2004). This shift in habitat use may further favor broad sampling methods like those utilized by citizen volunteers. Citizen science can be difficult to support in more rural areas due to the lower human population density but may prove to be valuable. Additional comparative studies in less disturbed watersheds could validate this hypothesis.

This study provided detailed information on two citizen science stream monitoring programs and how they compare to traditional academic assessments. These types of studies are increasingly common as appreciation for citizen science efforts increases and their value to agency and academic programs becomes clear. While academic and volunteer data in this study did not always perfectly align, the general assessment of site quality by citizen volunteers was validated. Importantly, if the goal of these programs is to monitor stream degradation, volunteer data may provide a more conservative estimate of degradation by exploring a larger variety of habitat within a stream reach. When coupled with detailed habitat assessments to account for the difference to academic methods, as is done in the MiCorps protocol, this can provide a valuable and cost-effective addition to monitoring efforts (Theobald et al. 2015). Additional information, when validated, can vastly improve regular monitoring of ecosystems and assist decision making for conservation and management purposes.

CHAPTER 3 THE IMPORTANCE OF ENVIRONMENTAL CONTEXT IN THE ROUND GOBY INVASION: NATIVE DIVERSITY AND RIPARIAN LAND USE INFLUENCE INVASION SUCCESS

Introduction

Invasive species can cause declines in native biodiversity, negatively affect water quality, and disrupt food webs and ecological processes, reducing overall environmental health (e.g., Ricciardi et al. 1998). Further, native biotas are susceptible to predation and competition associated with the introductions of novel species. In addition to negative environmental and ecological impacts of invasion, the economic consequences are significant, causing dramatic declines in revenue from recreational and commercial fisheries, a $4 billion per year industry (US EPA 1997), as well as altering water availability and quality for drinking water and hydropower (Pejchar and Mooney 2009), and decreasing property values (Horsch and Lewis 2009). Due to the degree of potential environmental and economic impact, nonnative species are one of the foremost concerns for ecosystem and species conservation (Wilcove et al. 1998). Effective prevention is by far the most suitable method of addressing invasive species problems compared to trying to control them after establishment, as mitigation and removal are more difficult and costly (Rout et al. 2011). This dilemma has motivated research to identify the environmental context in which invasion is most likely to occur. As a result, many attempts have been made to identify relationships between ecosystems of concern and the environmental factors associated with successful invasion or aggravate negative impacts following establishment.

The concept of 'invasibility', or how likely any system is to be invaded, evolved from ideas about competitive equilibria for species (MacArthur 1970) and was adopted for terrestrial species by Crawley (1988). Since this time, retrospective studies (e.g., Moyle and Light 1996; Gido and Brown 1999; Moyle and Marchetti 2006) have provided some indication of how these theories apply to freshwater habitats. Collectively, low native community diversity, available niche space (associated with disruption of habitat, recent extirpations, etc.), and environmental similarity to the nonnative species' range are commonly identified as potential local factors which can facilitate invasion. These attributes have variously been identified as key factors in known invasion events. For example, Moyle and Light (1996) suggested that complex communities pose increased biological resistance to new invaders due to their own elaborate histories of species assembly. With respect to niche space, Davis et al. (2000) identified resource availability as a key component in invasion success, either due to an increase in productivity or release from a pre-existing organism. Given this understanding, identifying natural ecosystem attributes can assist accuracy in understanding the environmental factors associated with invasion. However, human alteration of ecosystem structure and function can affect the degree of these relationships and must also be considered (Leprieur et al. 2008).

Nonnative introductions may occur over a large gradient of anthropogenic influence. As such, anthropogenic facilitation of invasion must be considered in addition to a system's natural attributes. When multiple stressors are already present, the impact on freshwater ecosystems from invasive species may be compounded (Bianchi and Morri 2000), and the potential for successful invasion is exacerbated (Strayer 2010). Specifically, human population size (McKinney 2006), contaminants (Hillery et al. 1997), nutrient runoff (Anderson et al. 2002), previous invasions (Glon et al. 2017), and propagule pressure (Lockwood et al. 2005) can act as stressors on native ecosystems. Changes in land cover can also impact ecosystems via multiple vectors across a landscape (Wolter et al. 2006). Because altered land cover can represent multiple individual impacts to an ecosystem, it can be used as a metric for overall degradation (Foley et al. 2009). Specifically, there can be large scale impacts from urban and agricultural development which have multiple, long-lasting consequences for freshwater biota. For example, in the Great Lakes, these widespread, landscape-scale stressors have been combined as a measure of 'cumulative stress' on freshwater ecosystems (Danz et al. 2007; Allan et al. 2013; Wang et al. 2015). This measure can thus be utilized to address how environmental degradation influences invasion success.

Conceptually there is reason to suggest invasion and disturbance theories would translate well to Great Lakes watersheds (Mills et al. 1991; Mills et al. 1994). However, research on inland lakes and tributaries has lagged in comparison to the Great Lakes. Yet, tributary and other lotic habitats represent an important vector for secondary dispersal of invasive species across the landscape (Bronnenhuber et al. 2011). For example, Bobeldyk et al. (2005) showed that streams provided vectors for dispersal of the invasive zebra mussel (*Dreissena polymorpha*) from source populations in lakes. Given the physical and biological differences between lake and river ecosystems, increasing the understanding of invasion in lotic waters may provide additional contextual information on species invasions, particularly those currently undergoing secondary spread.

The environmental conditions during and preceding an invasion can dramatically alter the likelihood of a nonnative species becoming established. An invader's broad tolerance to environmental conditions (Moyle 1986), the rates of predation encountered in the invaded area (Keane and Crawley 2002), and niche opportunity (Shea and Chesson 2002) have all been identified as important factors in predicting the success of an invasion. These characteristics, referred to as the 'context dependency' for invasion, are important components in determining best practices to limit the degree and extent of consequences from invasive species (Townsend 2003; Dick et al. 2017). Many of these components are subject to anthropogenic alteration which could either increase or decrease their relative influence on the success of any given invasion. Byers

(2002) found that anthropogenic alteration of habitats through eutrophication or trophic restructuring created an environment where the nonnative was favored because advantages associated with local adaptation were eliminated. It is thus important to consider the context-dependency surrounding invasion to inform best management practices and provide better assessments of the potential extent of invasion and the ultimate impact on native species and ecosystems.

While the specifics of any particular invasion can vary dramatically from one instance to another, I propose that system attributes create common opportunities that contribute to the successful establishment of nonnative species. Further, I suggest that anthropogenic influence can alter the nature and extent of invasion. Here I characterize system attributes that provide opportunities for nonnative establishment using the invasive round goby (*Neogobius melanostomus*) as a model. Building upon work done by the Michigan Rivers Inventory Project (Seelbach and Wiley 1997) and the Great Lakes Aquatic Habitat Framework (GLAHF 2015), I utilize land cover, watershed characteristics, habitat assessments, and biotic integrity to determine the abiotic and biotic context for successful species invasion by the round goby (*Neogobius melanostomus*) in seven Great Lakes tributaries in Michigan. Understanding what environmental characteristics need be present for invasion to occur, or the context-dependency, may ultimately be informative in limiting dispersal and impacts of non-native species.

Methods

In order to determine the environmental parameters associated with the invasion of round goby, I conducted a three-year survey in seven rivers where round goby was actively invading. The sample design included three sites on each river, some of which were previously invaded by round goby and hosted populations of varying size, and others which had yet to be invaded. Over the three years, round goby populations expanded into new sites (see Chapter 1 for further details).

Biotic parameters

Sampling occurred at wadable stream reaches approximately 282.81±99.26 meters in length on average and were chosen to represent a gradient in land use from primarily urban to primarily forest or wetland (see map of sites and location information from Chapter 1). We conducted a fish survey for approximately one hour at each site, one time per year to identify the relative species composition of the fish assemblage at each location. Fish were captured using a 3x1.5m nylon mesh seine (3.18mm mesh). Individual fish were identified on site and released, except for a subset which were euthanized via an overdose of MS222 (tricaine methanesulfonate) and preserved for species verification.

While current fish assembly characteristics are informative in identifying attributes common to round goby invasion, I was also interested in identifying the role of time since invasion in the characteristics of current round goby populations. Past fish survey data was gathered from several sources to narrow down timing of initial invasion for each sampling site. Fish collection information was downloaded from the FishNet2 online data repository (FishNet2 2017), the University of Michigan Museum of Zoology Fish Division online catalog (UMMZ 2017), the Midwest Invasive Species Network (MISIN 2017), the Great Lakes Aquatic Nonindigenous Species Information System (GLANSIS 2017), and the Global Biodiversity Information Facility (GBIF 2018). In addition, collection records from fisheries surveys and scientific permits were obtained from the Michigan Department of Natural Resources (DNR) (personal communication – T. Goniea and K. Wehrly, Michigan DNR). Where insufficient information occurred for any watershed, individual watershed status reports were obtained from the DNR where they included timing of invasion (Francis and Haas, 2006). Where collection record information was lacking, timing of invasion was informed by the fish survey data from this study. All records from the relevant watersheds which included round goby were mapped using ArcGIS. Where data points were spatially relevant, they informed a conservative estimate of the initial timing of round goby invasion.

Physical parameters

In addition to fish surveys, a suite of water chemistry and habitat assessments were completed at the time of sampling. Temperature, dissolved oxygen, conductivity, and pH were measured using a handheld YSI multiparameter instrument. Average stream depth was estimated for the reach following sampling. Stream width was measured following sampling using Google Earth measurement tools. Water samples were collected for analysis of copper concentration as a measure of chemical contamination associated with urban development (Van Metre and Mahler 2003). Fifteen mL samples were collected and preserved with concentrated nitric acid (HNO_3) and returned to the laboratory. Levels of dissolved copper (Cu) were measured in the lab using a Shimadzu AA-7000 Atomic Absorption Spectrophotometer. Concentrations were calculated from an average of three concurrent runs of each sample based on a calibration curve of laboratory standards at 0, 0.5, 1.0, and 1.5 mg L^{-1}.

Stream discharge is also an important component of the physical structure of the stream and was considered where possible. Stream gage data was obtained from the National Water Information System hosted by the United States Geological Survey (USGS 2018). A few of the sites sampled in this study have gages installed at the same location. For all others, the nearest available gage data was accessed (Appendix C: Table S2). Where necessary (due to lack of multiple gages), the same gage was used for all three sites in the watershed. Daily mean discharge (ft cm^{-3} or cfs) was calculated for each site on the day it was sampled.

Habitat parameters

I completed a habitat assessment during sampling for each site using the EPA's Rapid Habitat Assessment (RHA) protocol (Barbour et al. 1999) which identifies ten habitat parameters important to ecosystem function and allows the surveyor to rank the quality of each parameter on a scale of 0 to 20. The parameters address physical, chemical, and biological aspects of the system as reflected by the quality of the habitat. The measures include assessments of the substrate, channel morphometry, flow, and vegetation. Total RHA Scores were assigned to each site based on the sum of the ten parameters and expressed as a proportion of the total potential score.

In addition to the physical habitat of each stream reach, there are larger-scale physical attributes which have been documented to influence round goby invasion. Specifically, the presence of dams has been shown to be important in round goby invasion. Low-head dams can reduce stream flow to a pace navigable by round goby and provide pockets of lentic habitat which support step-by-step invasion across the landscape (Raab et al. 2018). To address the importance of this phenomenon in my study, the distance in river kilometers (linear distance along the stream flowline) between each sample site and the nearest downstream impoundment, and thus the corresponding reservoir, was included in analyses. Distance from the site to the mouth of the river in river kilometers was also measured using ArcGIS.

Land Cover

Finally, as a large-scale indicator of potential pathways of anthropogenic influence on invasions, land cover data was obtained from the United States Geological Survey National Land Cover Database (NLCD). The NLCD database provides mapping of the entire United States at a spatial resolution of 30 meters and categorizes each grid-cell relative to the main land cover type occupying that cell. There are twenty potential land cover types used by the NLCD. These categories were binned into five categories relevant to stream integrity (e.g. Ahearn et al. 2005; for full details on binning see Appendix C: Table S1). Land cover information from the years 2011 (Homer et al. 2015) and 1992 (Vogelman et al. 2001) were used to represent how the current landscape influences patterns of invasion and address the influence of land cover changes over time.

Land cover was identified for each watershed inside a 100m riparian buffer zone (from each bank) following the flowline of the stream using ArcGIS to account for the land cover most directly affecting stream form and function inside the watershed (Allan 2004). This method allows for the reduction of the data to only that which most directly affects each site. In addition, each watershed was clipped along the stream line such that only the land area upstream of each site was considered. This resulted in the downstream-most site incorporating the largest land area, so proportional abundances of each land cover type were assigned to each site. Current proportional composition of land cover (from the 2011 NLCD) and prior land cover composition (from the 1992 NLCD – to address any legacy affects) were evaluated as factors potentially influential to invasion.

Data Analysis

A modeling effort was used to determine which of these factors were correlated with round goby invasion success in the seven watersheds in this study. Each data type was processed as described below prior to developing the model.

Fish assemblage data from fish surveys from 2015 to 2017 were analyzed for assemblage diversity using the adjusted Shannon's diversity index proposed by Chao and Shen (2003). This adjusted index accounts for rare species that were missed during surveys so that the richness of the assemblage is not underestimated when computing diversity metrics. This method uses a maximum likelihood approach to assess the individual probabilities of species discovery relative to the total number of species to determine sample coverage. Traditional Shannon's diversity is then corrected using the estimation of coverage to account for rare and potentially present, but missed, species.

For various reasons, occasional missing data points existed for four of eight parameters in the physical data. Because missing data can skew results in multivariate analyses, missing points were imputed by multiple imputation using the supplemental R package, missMDA (Josse and Husson 2016). Data was assessed for normality of each variable using the Shapiro-Wilk test. All parameters which were not normal were transformed via ln(x+1) transformations where it aided in normality of the distribution.

The nature of lotic systems can cause collinearity between many of the measured parameters in this study. To address collinearity and to reduce bias toward the explanatory power of correlated variables, a Principal Component Analysis (PCA) was done on the physical data measured at each site using the add-on R package, FactoMineR (Lê et al. 2008). This allowed the reduction of these variables to Principle Component scores to be used in further analyses, while still representing the gradient of variation explained by covariable physical parameters. Analysis of scree plots allowed for selection of the number of PC scores which explained the greatest amount of variation (>85%) within the physical data while minimizing the number of parameters. Because multivariate analyses are sensitive to missing data, Stony Creek and the Ocqueoc River were excluded from the physical parameter PCA because discharge data was entirely lacking for these watersheds (no active USGS gages).

A similar PCA was conducted for proportional land cover data within the 100m upstream flowline buffer for each site. Binned land cover data from both 1992 and 2011 were analyzed. An arcsine square root transformation was done on each land cover category where it improved the normality of the data. The PC scores which explained the majority of the variation were retained for further analysis as above.

Model Building

All physical, habitat, biotic, and land cover parameters (Table 6) were assessed for their predictive power in the presence and proportional abundance of round goby using boosted regression trees (BRT; Elith et al. 2008). Using the add-on package, 'gbm', in the R statistical environment with additional source code by Elith et al. (2008), I used a stepwise selection procedure which optimized the number of trees, learning rate, and tree complexity. The model fit was evaluated with a cross-validation technique due to the relatively small sample size in this study (i.e., <1000 observations) with the aim of minimizing the model deviance from observed data. I used the model simplification backward selection function to eliminate variables with the least amount of contribution to the dependent variable variance based on the cross-validation error and ultimate model performance.

All analyses were done using the statistical software package, R (R Core Team, 2016). Additional packages were used for specific analyses as indicated above.

Table 6. Variables included in the BRT model for identifying the environmental context of round goby populations.

Independent Variables	Description
Physical PCA1	First principal component from the physical data PCA
Physical PCA2	Second principal component from the physical data PCA
Physical PCA3	Third principal component from the physical data PCA
Land PCA1	First principal component from the land use PCA
Land PCA2	Second principal component from the land use PCA
Land PCA3	Third principal component from the land use PCA
Land PCA4	Fourth principal component from the land use PCA
Land PCA5	Fifth principal component from the land use PCA
H_adjusted	Shannon's Diversity Index adjusted for rare species
RHA	Rapid Habitat Assessment scores
Dam_distance	Distance to the nearest downstream impoundment (rkm)
Copper	Measured concentrations of copper (mg/L)
Mouth_distance	Distance from site to the mouth of the river
Invasion_year	Estimated year of invasion

Results

Biotic parameters

Native diversity in the fish assemblage varied among watersheds and sites (Figure 12). The lowest diversity appeared in the midstream site in the Rifle River (1.08), while the highest was the upstream site in the Muskegon (2.05). The lowest watershed-level diversity was observed in the Rouge River (1.48), while the highest was in the Muskegon River (1.91). Round goby comprised an average of 15% of the fish assemblage at sites where it was present and was absent from 17 sites over the course of three years of sampling. There was an increase in proportional abundance of round goby and the number of sites at which it was present over time.

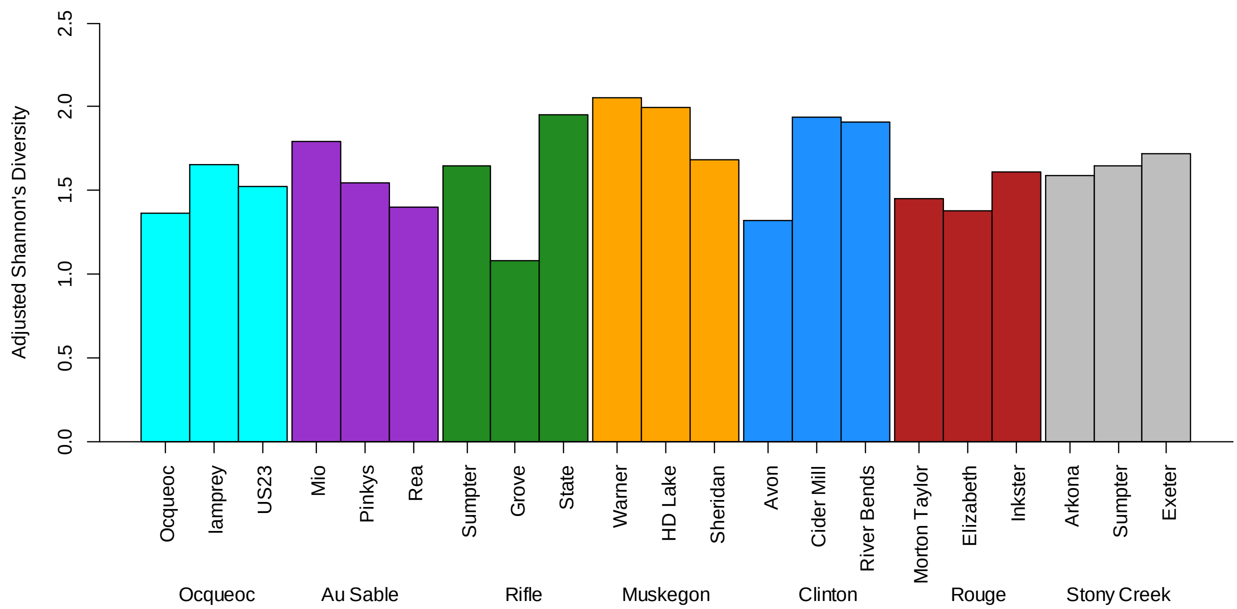


Figure 12. Adjusted Shannon's Diversity Index for each site (years combined). Diversity indices represent a mean for each site over the three years of sampling.

Fish collection information from online databases, reports, and survey information yielded 48 round goby collections that were spatially relevant to this study. A collection record was deemed relevant when it occurred at or upstream of a given site, indicating that round goby had successfully invaded or surpassed those sampling sites by the time of the collection. The dates for these records were interpreted as minimum estimates of initial round goby invasion (though actual invasion timing may have occurred earlier than these initial survey events). Combined with the data from my surveys, an approximate time of initial invasion was assigned to each site in the study (Table 7).

Table 7. Estimated timing of initial invasion and the sources from which the timing was derived. 'Krabbenhoft' denotes invasion timing derived from this study, acronyms refer to the online databases mentioned above, 'Permit reports' are scientific collector permits obtained from the Department of Natural Resources, 'Fisheries surveys' are surveys conducted by the DNR, and 'DNR reports' are special reports produced by the Fisheries Division of the DNR on the status of various watersheds as needed. 'NA' denotes sites that had not yet been invaded by the conclusion of this study.

River	Site	Estimated Year of Invasion	Source
Ocqueoc	Ocqueoc	NA	
	Lamprey	2017	Krabbenhoft
	US23	2016	Krabbenhoft
Au Sable	Mio	NA	
	Pinkys	NA	
	Rea	2002	GLANSIS
Rifle	Maple Ridge	2016	Krabbenhoft
	Grove	2014	MISIN
	State	2014	MISIN
Muskegon	Warner	2013	GLANSIS
	Holton Duck Lake	2013	GLANSIS
	Sheridan	2004	Permit report
Clinton	Avon	NA	
	Cider Mill	2001	DNR report
	River Bends	2001	DNR report
Rouge	Morton Taylor	2013	Permit report
	Elizabeth	2013	Permit report
	Inkster	2013	Permit report
Stony Creek	Arkona	2010	Fisheries surveys
	Sumpter	2010	Fisheries surveys
	Exeter	2010	Fisheries surveys

Physical parameters

The first principal component for the habitat parameters was largely correlated with differences in dissolved oxygen and stream width. The second component was driven largely by stream depth (Figure 13). Collectively, the first three components of the PCA explained 85.6% of the variation and were retained for further analysis (Appendix C: Table S3). These results suggest channel morphometry is in large part driving the variation in physical parameters among sites in this study.

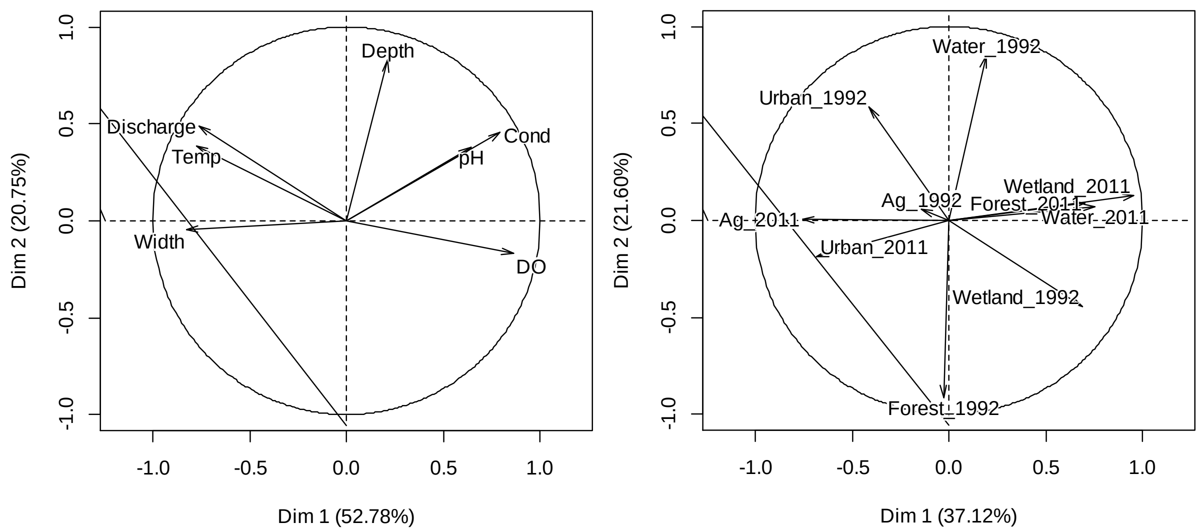


Figure 13. Loading plot of physical parameter PCA (left) and the land cover PCA (right). Length of arrows indicates relative contribution to variation. Total variation explained by the first two components is 73.53% for the physical data and 58.72% for the land use data.

Habitat parameters

The habitat assessment revealed a general decrease in RHA score along sites moving from north to south, corresponding to an increase in human population density (Appendix C: Table S4). The values were highest in the Ocqueoc and Au Sable watersheds, which have the highest proportion of forest and wetland land cover types, and lowest in Stony Creek, which was primarily agricultural. There was also a general upstream-to-downstream gradient, where most upstream sites were rated as higher quality than the downstream sites. However, in some cases the midstream site was rated highest; for example, both the Ocqueoc and Au Sable rivers had a site rated as 'Reference' quality at the midstream point.

Land Cover

The five binned land cover types were summarized by the percent area of the 100m buffer for all streamlines upstream of each site (Figure 14). The Rouge and Clinton River watersheds were by far the highest in urban development, while Stony Creek had relatively high contributions of agricultural land use. The Au Sable, Rifle, and Ocqueoc were the most forested watersheds. The Ocqueoc and Au Sable Rivers also had notable proportions of wetland in the riparian buffer area. Land cover varied widely between 1992 and 2011. There was an increase in urban development in all watersheds between 1992 and 2011, except the Muskegon River (Appendix C: Table S5). Forest, wetlands, and water decreased as agriculture and urban development increased over time, suggesting expansion of anthropogenic uses. The first component of the PCA corresponded largely

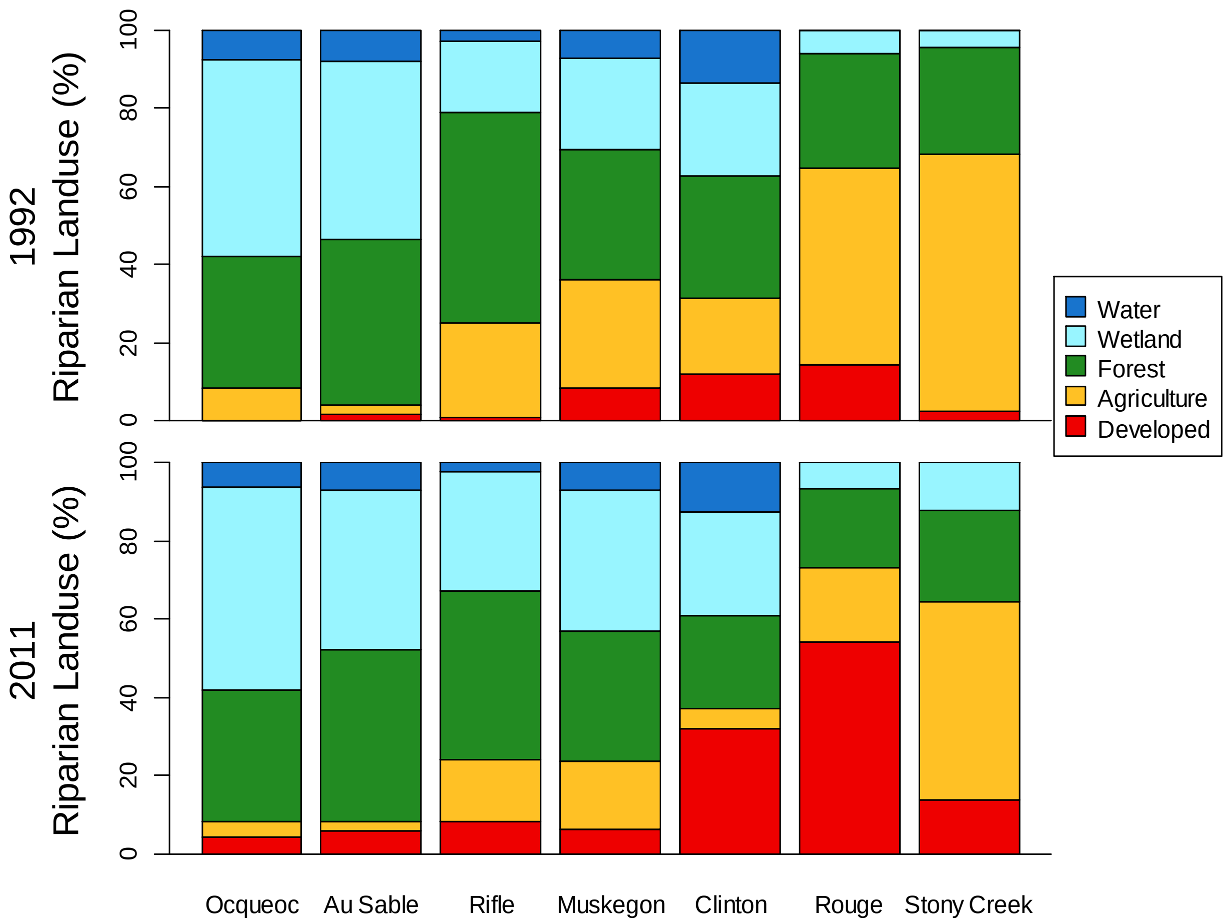


Figure 14. Upstream riparian land use inside a 100m buffer (from each bank) for all rivers in 1992 (top) and 2011 (bottom). Data was combined for all three sites for this figure.

to a trade-off between natural land cover types (forest, water, wetlands) and human development (urban, agriculture) in 2011 (Figure 13). The second component corresponded to variation in the 1992 data. The first five components of the PCA explained 92.3% of the variation among sites (Appendix C: Table S3) and were retained for further analysis.

Boosted Regression Tree Model

The final model for environmental conditions associated with round goby presence as determined by stepwise model selection used a learning rate of 0.0005, tree complexity of 4, and resulted in an optimal 4,550 trees. The final estimated deviance was 0.091 (SE = 0.011) and the correlation with training data from the cross-validation procedure was 0.768. The recursive elimination of model features resulted in the removal of eight variables to optimize model

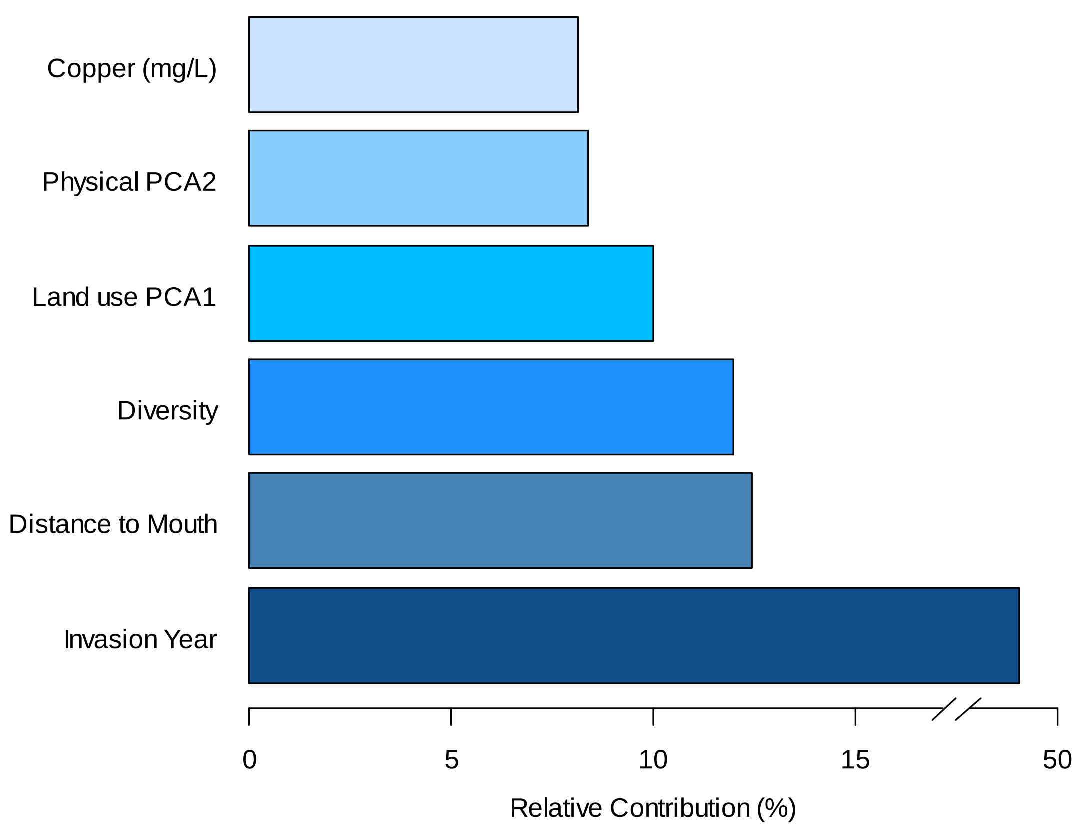


Figure 15. The relative contribution of the six most informative variables in determining round goby proportional composition in the fish assemblage across sites and years as assigned by the final BRT model. Note the break in the x-axis to accommodate the high explanatory power of estimated invasion year.

performance including the RHA, distance to dams, and several PC scores (Appendix C: Figure S1). The model's explanatory power was largely driven by the initial year since invasion and distance from the site to the mouth of the river. Site characteristics which contributed to remaining variation included native fish assemblage diversity, the first principal component for the riparian land cover PCA, the second principal component from the physical parameter PCA, and copper contamination (Figure 15).

Functions were fitted to each explanatory variable to indicate the direction of the relationship between the variable and the proportional abundance of round goby in the fish assemblage (Figure 16). These relationships collectively indicate that round goby is lower in abundance in areas where it has only recently invaded, it is farther from the source population (the Great Lakes), the native fish diversity is high, natural land uses are dominant, the river itself is of moderate size, and chemical contaminants are low.

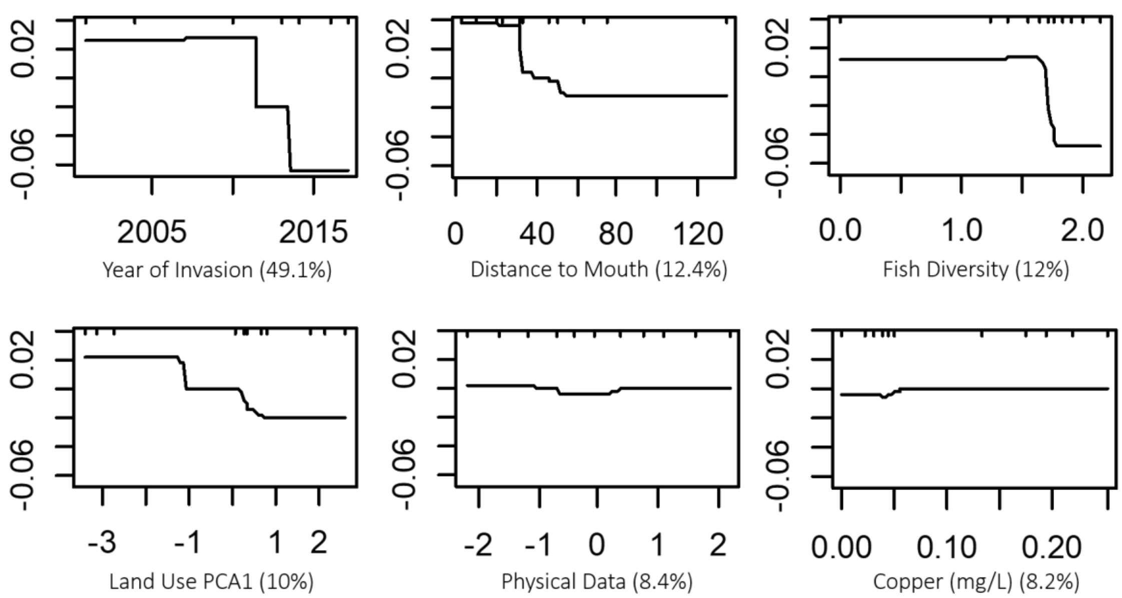


Figure 16. Fitted function plots for the six explanatory variables in the final BRT model. Y-axes are adjusted to a common scale to allow for comparison but represent the dependent variable (proportional abundance of round goby). Plots that decrease from left to right along the x-axis suggest a decrease in round goby abundance as the variable being measured increases.

Discussion

This study used several data types to develop a model for identifying the context-dependency of round goby invasion. Here I incorporate biotic, physical, chemical, and landscape parameters to identify the characteristics common to streams where round goby have successfully invaded and persisted in Michigan streams.

Physical and landscape characteristics identified a gradient in overall site quality, largely corresponding to a north-to-south gradient across the state of Michigan. More densely populated areas in southern Michigan typically had the lowest scores in the habitat assessment and the highest urban and agricultural land use. These findings indicate that round goby has successfully invaded a wide gradient of watershed and reach types, despite differences in stream morphometry and landscape composition. This corresponds well to the variety of habitats round goby exploits in the Great Lakes themselves, and the proposed broad environmental tolerance of round goby that has allowed it to become such a prolific invader (Kornis et al. 2012).

Several factors were removed from the final boosted regression tree model during backward elimination due to a lack of contribution to the explanatory power. Contrary to results by Raab et al. (2018), distance to the nearest impoundment was not correlated with round goby abundance. However, only ten of 21 sites in this study had significant impoundments downstream. This is a factor of sample design in this study as sites were chosen explicitly due to known existence of round goby populations in downstream reaches. Impoundments are relatively common in these watersheds, but sampling efforts were sometimes concentrated to downstream reaches because dams had thus far impeded invasion (though not in all cases). The habitat assessment (RHA) was similarly removed for lack of explanatory power. I suggest this is because many of the parameters assessed by the RHA are correlated with the PC scores from the measured physical parameters. While RHA does provide a robust assessment of overall habitat quality (Barbour et al. 1999), smaller scale parameters specific to the site (like those in the physical PCA) may provide a finer assessment of quality and render any extra variation explained by the RHA negligible.

The parameters included in the final BRT model indicated variation in round goby abundance was largely associated with time since invasion and the distance from the source population. These are logistic factors in any invasion that have previously been determined to be important factors in estimating the size of an invasive population. This analysis serves to support prior conclusions that propagule pressure is an important factor in determining the successful establishment of an invasive population (Lockwood et al. 2005). Proximity to the source population increases the likelihood of multiple introductions and increased time since invasion allows for the time necessary for the population to become established.

The sample site characteristics associated with the patterns observed in round goby populations were much more telling. The importance of native fish diversity potentially supports the biotic resistance hypothesis which states that more diverse communities are less likely to be invaded (Elton 1958). Ross (1991) posed that increased native diversity would decrease invasion potential because of established food web structure. Higher diversity may thus indicate a lack of available niche space for the invader because all available niche space is already exploited (Shea and Chesson 2002). However, it is unclear in this study whether low native diversity preceded round goby invasion or whether it changed over time coincident to invasion. Other aspects of the environmental context may independently influence both invasion and changes in native diversity, thus making this relationship coincidental. However, as identified in Chapter 1, some native species may decline in response to invasion. Thus, round goby invasion may have induced the decreases in native diversity, not the other way around. Determining whether this relationship is correlative or causal (in either direction) would require more precise estimates of invasion timing and a larger pool of historical fish assemblage data. Regardless, the relationship between round goby abundance and native assemblage diversity found in this study makes the case for conservation efforts maintaining native biodiversity in the face of invasion.

The first component of the land cover PCA was also an important site characteristic corresponding to round goby abundance. This component largely indicated a trade-off between anthropogenic and natural land cover types. Natural land cover types were negatively correlated with round goby abundance. This relationship supports the idea that anthropogenic development in a watershed can facilitate invasion, a concept previously identified in terrestrial systems (Hobbs 2000). Land cover has also been identified as an important driver of ecosystem integrity in streams (Allan 2004) but the application of this concept to invasion facilitation is relatively novel. While different land cover types impact streams through a variety of mechanisms (Allan 2004), I demonstrate here that an outcome for both urban and agricultural watersheds is an increased incidence of invasive species. This underscores the importance of considering ecosystem function during development for human endeavors. Specifically, maintaining natural land cover (and thus minimizing disturbance) in riparian corridors may decrease the abundance of invasive species, potentially minimizing ultimate environmental and economic consequences.

The physical data contributing to the model largely corresponded to stream morphometry. With stream depth largely driving the variation within this variable, this relationship suggests overall stream size can be an indicator of suitable habitat for round goby. The Great Lakes serve as the source for invasion and this finding may indicate a preference among initial invaders for large bodies of water that are more like the lacustrine habitat common to the source populations.

Round goby also have aggressive territory defense (Meunier 2009) so areas of initial range expansion need to be large enough to accommodate home range sizes for multiple individuals. While lake home ranges tend to be larger than those in streams (Ray and Corkum 2001), invading large rivers would ease the environmental transition when moving from lake to stream. This tendency also reflects similar preference for lakes and larger rivers as observed in the native range of round goby in Europe (Kornis et al. 2012).

Finally, copper concentration was identified as an important variable. Dissolved metals, can pose serious threats to aquatic organisms including fish and zooplankton (Griffitt et al. 2008). The mechanisms of toxicity can depend on the type of copper present, but Meng et al. (2007) found that reactive copper affects the body's ion balance and that copper accumulates in renal tissues. Copper is also a known trophic toxicant due to its ability to bioaccumulate (Zyadah and Abdel-Baky 2000). Copper levels tend to be higher in areas with high human population density and an abundance of motor vehicles. Tires and brake linings both contain copper and may serve as non-point sources for the metal contamination in freshwater systems (Paul and Meyer 2001). As such, copper concentration may be another consequence of anthropogenic land use that extends beyond the immediate riparian area. Dissolved metals may ultimately contribute to underlying levels of stress on native organisms leading to lower resistance to disturbances (Kashian et al. 2007), thus corresponding to an increase in the likelihood of successful round goby invasion.

In this study, I have identified six factors which correspond to round goby secondary invasion into Michigan streams. These characters collectively support long-standing ecological hypotheses (Moyle 1986) that loss of biodiversity and an increase in anthropogenic disturbance drive incidence of invasive species in streams. While proximity to source populations remains important in invasion prediction, the spatial barriers which have often precluded initial introductions may only provide short-term barriers to range expansion as human activities act as vectors for movement of species across the landscape (Davidson et al. 2015). Dreissenid mussels are a perfect example of how even species with limited motility can have enormous consequences when introduction is facilitated by humans (Cariton and Geller 1993). For fish, the regular use of round goby as a bait fish has subsequently led to introductions to inland lakes that were unreachable via natural migration (Kornis et al. 2012). The spatial challenges associated with invasion are thus becoming less important, suggesting that limiting destruction of riparian corridors, and maintaining habitat integrity are the best means available to increase system resistance to invasion.

SIGNIFICANCE

Through this research, system-specific data on round goby, one of the Great Lakes' most prolific invaders, was produced. This is an important step in understanding the round goby's function and impact in tributaries of three Great Lakes. Fully characterizing the biology of this invader can ultimately inform management of round goby populations throughout the Great Lakes basin and potentially elsewhere. Because this research occurred in systems with a range of population densities, pollution levels, and habitat types, this work has contributed information on the round goby's tolerance levels in its introduced range. In assessing correlation of round goby presence and abundance with various environmental characteristics, I have provided valuable information for the management and control of this invasive species.

This work further adds to the discussion of streams and rivers as an important vector for secondary dispersal of invasive species (Bronnenhuber et al. 2011). Although this concept is not new, work on invasion in lotic waters has lagged in comparison to research on the Great Lakes themselves. Because rivers are more intimately linked to the landscape than lakes, they provide important context for species invasions. Understanding the interaction between anthropogenic activities and the associated consequences for invasion success is critical for combatting the secondary spread of nonnative species (Blanchet et al. 2009). While the interaction between landscape and ecosystem function of rivers has been studied for a long time (Allan 2004), how these interactions influence invasion is relatively novel. This research has thus contributed to understanding the linkage between tributary, lake, and terrestrial ecosystems.

Finally, this work has resulted in a method of prediction for the continuing spread of round goby across the Great Lakes. Round goby distribution expanded over the course of this study and continues to do so. The impacts of this invader are being felt in both environmental and economic contexts and as shown here, populations do not appear to decline after initial invasion. There is much to be gained by curtailing round goby invasion where possible. Interest in predictive methods for addressing invasion has increased in recent years due to the difficulty in dealing with invaders after establishment. In identifying the environmental, biological, and chemical context associated with successful invasion, this work not only tests long-standing ecological hypotheses about invasion, but also adds to existing knowledge of the environmental tolerance of round goby. This information may ultimately contribute to efficient methods to identify areas potentially vulnerable to round goby invasion and contribute to conservation of native ecosystems and species.

APPENDIX A

Table S1. Site locations. Rivers listed from north to south, sites listed from upstream to downstream. HUC refers to the Hydrologic Unit Code assigned to the watershed by the United States Geological Survey. Coordinates according to GCS_WGS_1994.

River	Basin	HUC-8	Site Name	Latitude	Longitude
Ocqueoc River	Lake Huron	04070003	Ocqueoc	45°27'23"N	84°05'16"W
			Lamprey barrier	45°28'52"N	84°06'35"W
			US23	45°29'17"N	84°04'35"W
Au Sable River	Lake Huron	04070007	Mio	44°39'34"N	84°07'45"W
			Pinkys	44°30'13"N	83°47'55"W
			Rea	44°26'10"N	83°26'04"W
Rifle River	Lake Erie	04080101	Maple Ridge	44°08'30"N	84°02'37"W
			Grove	44°04'50"N	83°57'55"W
			State	44°02'05"N	83°50'48"W
Muskegon River	Lake Michigan	04060102	Warner	43°20'48"N	85°56'26"W
			Holton-Duck Lake	43°17'52"N	86°04'44"W
			Sheridan	43°15'43"N	86°10'54"W
Clinton River	Lake Erie	04090003	Avon	42°39'53"N	83°09'18"W
			Cider Mill	42°40'17"N	83°05'46"W
			River Bends	42°38'55"N	83°03'27"W
Rouge River	Lake Erie	04090004	Morton-Taylor	42°16'58"N	83°27'58"W
			Elizabeth	42°17'07"N	83°23'19"W
			Inkster	42°17'56"N	83°18'24"W
Stony Creek	Lake Erie	04100001	Arkona	42°05'56"N	83°36'17"W
			Sumpter	42°02'17"N	83°28'31"W
			Exeter	42°01'24"N	83°25'10"W

Table S2. Information on fish species binned for data reduction in PCA and MANOVA analyses of fish assemblage. Total N reported is for all years and sites combined.

	Common Name	Species	Total N
Bin 1	American brook lamprey	*Lethenteron appendix*	1
	unidentified lamprey ammocoete	unknown	1
Bin 2	Black redhorse	*Moxostoma duquesnei*	2
	Golden redhorse	*Moxostoma erythrurum*	24
	Silver redhorse	*Moxostoma anisurum*	3
Bin 3	Brown trout	*Salmo trutta*	1
	Chinook salmon	*Oncorhynchus tshawytscha*	13
	Coho salmon	*Oncorhynchus kisutch*	1
	Rainbow trout	*Oncorhynchus mykiss*	36
	unidentified trout parr	unknown	2
Bin 4	Northern common shiner	*Luxilus cornutus*	427
	Shiner hybrid	*L. cornutus* x *L. chrysocephalus*	18
Bin 5	Smallmouth bass	*Micropterus dolomieu*	99
	unidentified juvenile bass	*Micropterus* sp.	2
Bin 6	Bluegill	*Lepomis macrochirus*	62
	unidentified larval sunfish	unknown	1

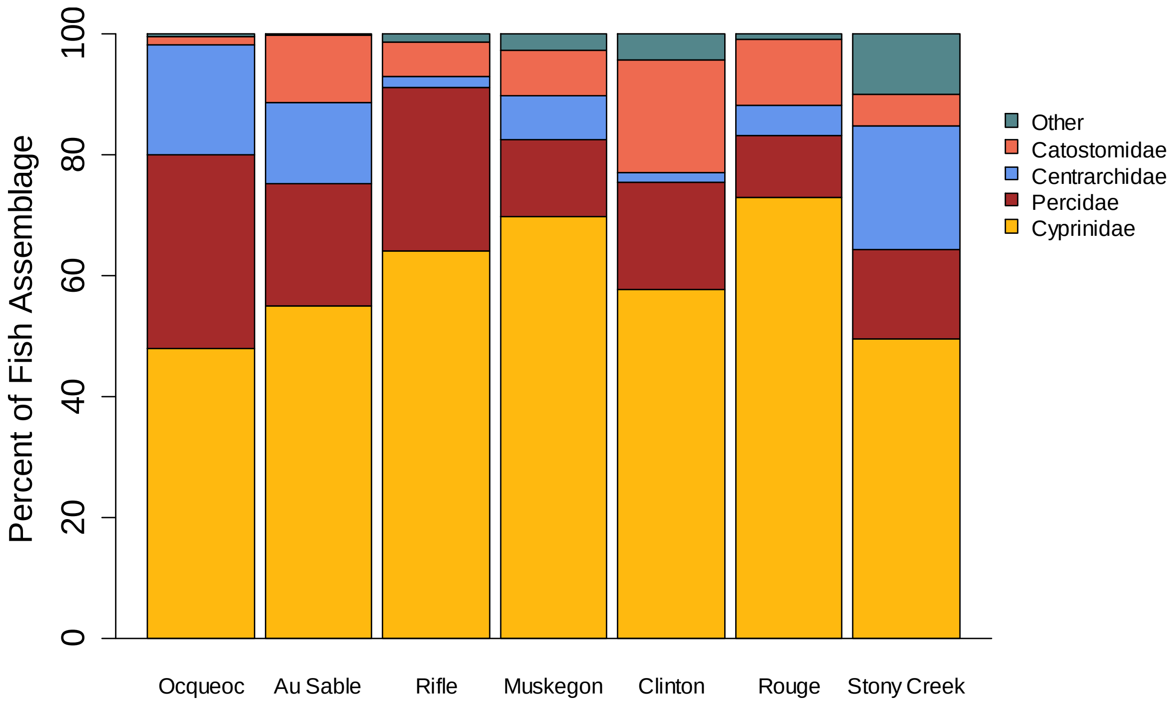


Figure S1. Native fish assemblage according to family for each river (all years combined). The four most abundant families are listed. The family group "Other" consists of Atherinidae, Cottidae, Esocidae, Fundulidae, Ictaluridae, Salmonidae, Petromyzontidae, and Umbridae.

APPENDIX B

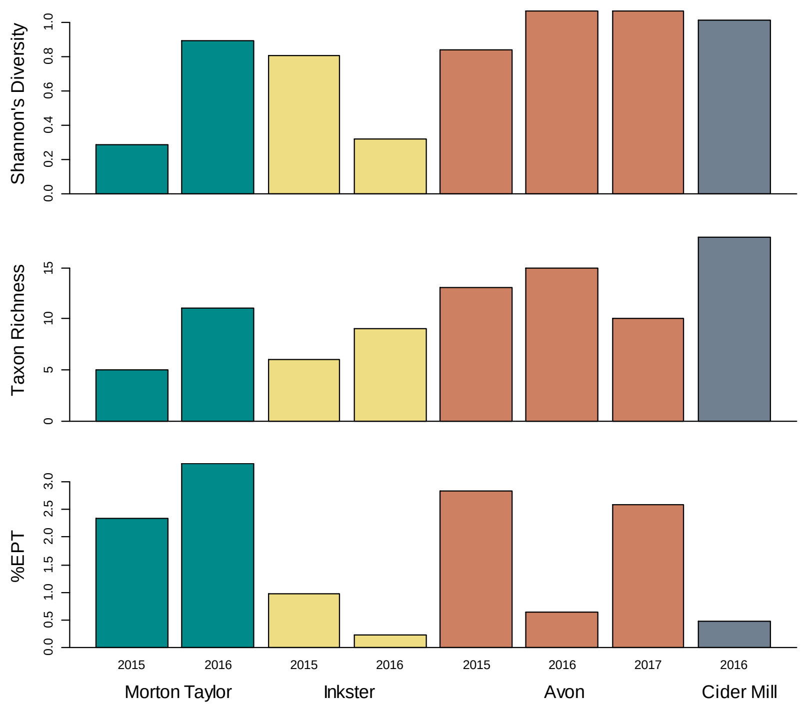


Figure S1. Diversity (top), richness (middle), and %EPT taxa (bottom) for all eight sites. Data is derived from the quantitative academic assessment.

APPENDIX C

Table S1. Full description of the NLCD land use categories from 1992 and 2011 and the corresponding bins that were used in this study. Not all twenty categories were present in the land area of this study; only those present are noted here.

1992 classification	Bins
Open Water	water
Low-Intensity Residential	developed
High-Intensity Residential	developed
Commercial/Indust./Transport.	developed
Bare Rock/Sand/Clay	developed
Strip Mine/Quarry/Gravel Pit	developed
Transitional Barren	developed
Deciduous Forest	forest
Evergreen Forest	forest
Mixed Forest	forest
Natural Grassland/Herbaceous	forest
Hay/Pasture	agriculture
Row Crops	agriculture
Urban/Other Grasses	developed
Woody Wetland	wetland
Herbaceous Wetland	wetland

2011 classification	Bins
Open Water	water
Developed, Open Space	developed
Developed, Low Intensity	developed
Developed, Medium Intensity	developed
Developed, High Intensity	developed
Barren Land	developed
Deciduous Forest	forest
Evergreen Forest	forest
Mixed Forest	forest
Shrub/Scrub	forest
Grassland/Herbaceous	forest
Pasture/Hay	agriculture
Cultivated Crops	agriculture
Woody Wetlands	wetland
Emergent Herbaceous Wetlands	wetland

Table S2. USGS operated stream gages for which discharge data was applied to each watershed. For the Au Sable, a different gage was applied to each site as appropriate. In the Rouge, the original gage service was discontinued after 2016 and a secondary gage was used instead. The Ocqueoc River and Stony Creek do not currently host any USGS stream gages and are excluded here. Coordinates according to NAD83.

Watershed	Site	Data Year	Gage ID	Nearest City	Latitude	Longitude
Au Sable	Mio	All	4136500	Mio, MI	44.6600	-84.1311
	Pinkys	All	4137005	Curtisville, MI	44.5608	-83.8028
	Rea	All	4137500	Au Sable, MI	44.4364	-83.4339
Rifle	All	All	4142000	Sterling, MI	44.0725	-84.0200
Muskegon	All	All	4121970	Croton, MI	43.4347	-85.6653
Clinton	All	All	4161820	Sterling Heights, MI	42.6145	-83.0266
Rouge	All	2015 & 2016	4168000	Inkster, MI	42.3006	-83.3002
	All	2017	4168400	Dearborn, MI	42.3084	-83.2537

Table S3. Statistical output of the physical data PCA (top) and the land use PCA (bottom). Components were kept if the cumulative variance explained was greater than 85%. Variables with a contribution greater than |0.4| are bolded.

Statistic	Comp 1	Comp 2	Comp 3
Eigenvalue	3.694	1.453	0.915
Proportion of Variance	52.777	20.752	13.069
Cumulative Variance	52.777	73.530	86.599
Variable		Loading Values	
Width (m)	**-0.828**	-0.043	0.390
Depth (m)	0.214	**0.827**	**-0.444**
Conductivity (µS/cm)	**0.794**	**0.455**	0.078
DO (mg/L)	**0.859**	-0.168	0.349
Discharge (cfs)	**-0.762**	**0.488**	0.198
pH	**0.640**	0.380	0.591
Temperature (°C)	**-0.777**	0.387	0.222

	Statistic	Comp 1	Comp 2	Comp 3	Comp 4	Comp 5
	Eigenvalue	3.712	2.160	1.531	1.042	0.782
	Percentage of variance	37.125	21.605	15.309	10.419	7.822
	Cumulative variance	37.125	58.730	74.039	84.458	92.280
	Variable			Loading Values		
2011	Agriculture	**-0.756**	0.005	**-0.482**	-0.259	0.069
2011	Urban	**-0.685**	-0.187	**0.605**	0.236	-0.064
2011	Forest	**0.701**	0.090	**-0.525**	-0.275	0.022
2011	Water	**0.755**	0.072	**0.474**	0.251	0.019
2011	Wetland	**0.955**	0.129	-0.102	0.034	-0.023
1992	Agriculture	-0.144	0.058	**-0.531**	**0.815**	0.116
1992	Urban	**-0.412**	**0.583**	0.228	-0.221	**0.589**
1992	Forest	-0.030	**-0.912**	0.062	-0.228	-0.193
1992	Water	**0.192**	**0.850**	0.216	-0.110	-0.383
1992	Wetland	**0.686**	**-0.446**	0.197	-0.033	**0.478**

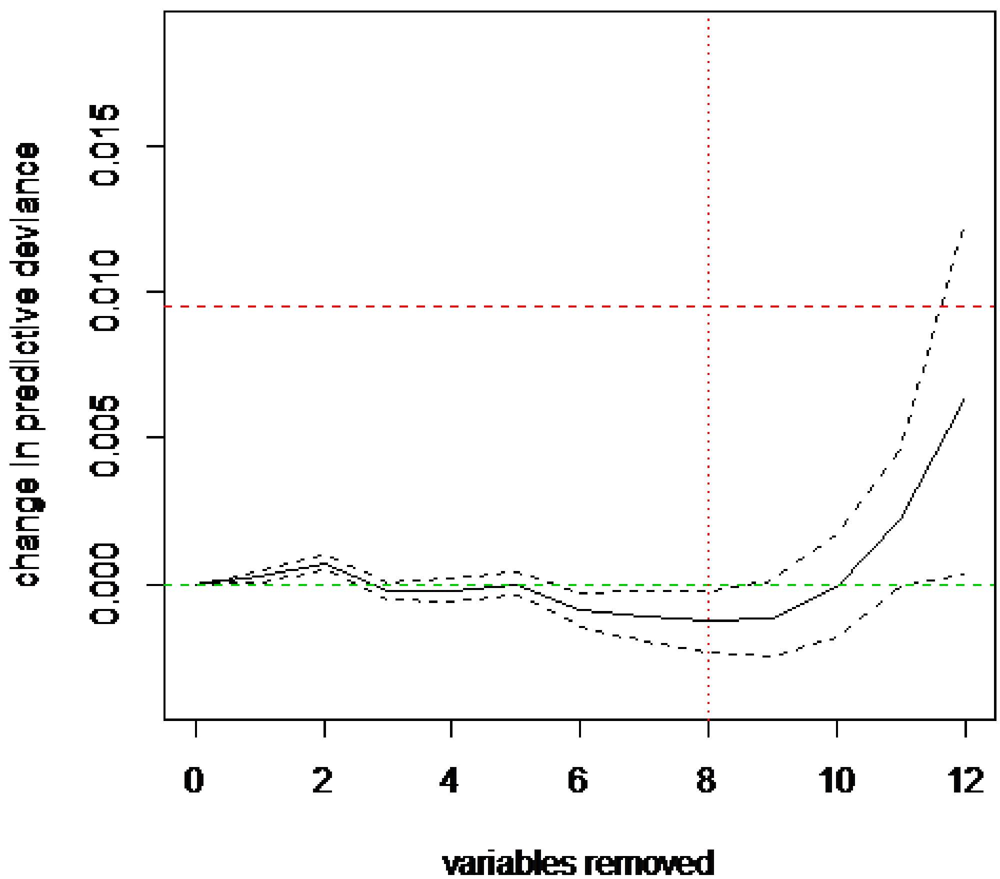
Table S4. Change in percent land use from 1992 to 2011 inside a 100m buffer (from each bank) for all sites. Negative values indicate a decrease in that land use type.

River	Site	Urban	Agriculture	Forest	Wetland	Water
Ocqueoc	Lamprey barrier	3.98	-4.20	-1.42	3.21	-1.57
	Ocqueoc Rd	4.38	-5.03	2.75	-0.99	-1.12
	US23	3.95	-4.19	-0.26	2.03	-1.52
Au Sable	Mio	4.40	-0.58	-1.80	-0.28	-1.75
	Pinkys	4.71	-0.78	-1.82	-0.94	-1.17
	Rea Rd	4.58	-0.75	-1.54	-1.15	-1.14
Rifle	Grove	7.29	-6.49	-6.77	6.36	-0.38
	Maple Ridge	7.01	-12.31	-25.44	32.38	-1.65
	State Rd	7.20	-7.14	-5.70	5.97	-0.34
Muskegon	Holton Duck Lake	-1.69	-10.41	-0.61	12.81	-0.11
	Sheridan Rd	-1.61	-10.18	-0.86	12.67	-0.01
	Warner	-2.05	-10.56	-0.49	13.34	-0.24
Clinton	Avon Rd	21.82	-10.83	-12.32	2.75	-1.42
	Cider Mill	17.72	-13.13	-6.79	2.96	-0.76
	River Bends	21.02	-16.17	-6.77	2.72	-0.80
Rouge	Elizabeth	41.94	-32.31	-10.47	0.81	0.03
	Inkster	40.24	-30.50	-10.58	0.81	0.03
	Morton-Taylor	32.54	-29.62	-3.50	0.30	0.28
Stony Creek	Arkona	7.57	-10.60	-3.37	6.28	0.12
	Exeter	11.08	-14.99	-3.68	7.57	0.02
	Sumpter	11.63	-15.80	-4.41	8.55	0.02

Table S5. Results of the Rapid Habitat Assessment for each site. Habitat parameters are Epifaunal Substrate/Available Cover (ES); Pool Substrate Characterization (PS); Pool Variability (PV); Sediment Deposition (SD); Channel Flow Status (CF); Channel Alteration (CA); Channel Sinuosity (CS); Bank Stability (BS); Bank Vegetative Protection (BV); Riparian Vegetative Zone Width (RP).

River	Site	ES	PS	PV	SD	CF	CA	CS	BS	BV	RV	Total	RHA Index	Quality Assignment
Ocqueoc	Ocqueoc Rd	16	15	16	15	17	19	14	10	9	11	142	0.71	Good
	Lamprey barrier	20	20	18	17	18	16	15	17	19	18	178	0.89	Reference
	US23	17	16	5	19	19	8	12	10	16	18	140	0.7	Good
Au Sable	Mio	16	18	7	18	19	13	6	16	10	10	133	0.665	Good
	Pinkys	16	18	16	17	19	19	14	14	19	20	172	0.86	Reference
	Rea Rd	19	15	17	19	19	19	6	10	19	19	162	0.81	Good
Rifle	Maple Ridge	16	12	2	15	17	18	5	12	19	19	135	0.675	Good
	Grove	20	17	18	18	19	18	7	10	12	10	149	0.745	Good
	State Rd	2	6	8	14	15	14	7	6	16	16	104	0.52	Fair
Muskegon	Warner	14	12	17	18	19	13	7	12	14	12	138	0.69	Good
	Holton Duck Lake	4	11	13	8	18	19	8	10	16	18	125	0.625	Fair
	Sheridan Rd	16	14	13	19	19	15	12	15	12	9	144	0.72	Good
Clinton	Avon Rd	4	7	12	15	15	15	14	10	10	10	112	0.56	Fair
	Cider Mill	15	18	15	16	19	14	15	5	7	5	129	0.645	Fair
	River Bends	16	14	15	8	18	19	15	9	10	8	132	0.66	Good
Rouge	Morton-Taylor	5	8	13	12	16	19	12	10	19	19	133	0.665	Good
	Elizabeth	12	10	10	15	17	9	5	10	18	20	126	0.63	Fair
	Inkster	14	10	12	15	18	18	14	15	10	14	140	0.7	Good
Stony Creek	Arkona	11	13	10	13	16	19	13	15	17	7	134	0.67	Good
	Sumpter	1	7	13	12	18	10	8	5	10	10	94	0.47	Fair
	Exeter	3	6	1	5	17	11	4	10	16	15	88	0.44	Fair

Figure S1. The change in predictive variance in the full BRT model (no parameters removed). An abrupt increase in the predictive variance as variables are removed indicates the ideal number of variables to remove from the model to increase explanatory power while decreasing model deviance.



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ABSTRACT

DRIVERS AND IMPACTS OF THE INVASIVE ROUND GOBY (*NEOGOBIUS MELANOSTOMUS*) IN MICHIGAN TRIBUTARIES TO THE GREAT LAKES

by

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The abundance and persistence of the invasive round goby (*Neogobius melanostomus*) has often resulted in antagonistic interactions between the invasive and its native competitors. In this study, I sought to quantify the consequences and environmental context of these interactions in Great Lakes tributaries. Specifically, I aimed to identify changes in feeding and reproductive behavior in a native competitor in response to round goby invasion, identify potential solutions to increase regular stream monitoring by tapping into citizen science programs, and quantify the environmental context associated with successful goby invasion. Surveys of fish communities were conducted over three years in seven Michigan tributaries to the Great Lakes. Each site was evaluated for fish assemblage composition, round goby abundance, and habitat quality. Individual round goby and a native competitor, the Johnny darter (*Etheostoma nigrum*), were dissected for a diet comparison and to identify investment in reproduction to illustrate changes in feeding and reproductive behavior by the native species. To inform better practices for stream management and invasion detection, a quality assessment of two citizen science programs in the area was completed. Citizen data was directly compared to traditional research focused sampling methods to verify the validity of the data and its potential inclusion in ecological research. Finally, a model

was developed to identify the environmental context common to sites invaded by round goby. Results suggest that Johnny darter diet diversity decreases, trophic position increases, and reproductive timing changes when goby are present. Citizen science may provide a way to monitor stream degradation which can facilitate these negative interactions. Despite differences in sampling methodology, qualitative citizen data reached similar conclusions about site quality as quantitative research methods. As identified by the environmental context model, altered riparian land use and decreased native species diversity are common characteristics of sites invaded by round goby. Regular monitoring for these characteristics may help identify locations vulnerable to round goby invasion so prevention and mitigation resources can be efficiently allocated. This research provides background on round goby invasion that can be utilized to better manage native species and ecosystems to increase resistance to and reduce the impacts of invasion.

AUTOBIOGRAPHICAL STATEMENT

I grew up in Albuquerque, New Mexico. I am one of those who came to biology later in life, never having been properly exposed to the diversity of life as a kid. I forced my way through introductory biology classes as an undergraduate, but never really fell in love with the subject until I took an Ichthyology class during my time as an undergraduate at the University of New Mexico. To say this course changed my life is an understatement. Among many other things, this course introduced me to a subject for which I truly had a passion. I got a job working in the Fish Division of the Museum of Southwestern Biology and never looked back.

Moving to the Great Lakes area after finishing my undergraduate and masters work in the arid southwest was quite the change. I enjoyed learning about an entirely different ecosystem with its own challenges. However, I found that despite the many differences between the southwest and the Great Lakes, we are all mostly concerned with the same ideals: what can we do to protect native ecosystems and species? I have kept this idea as a central theme to my work and will continue to do so as I move forward in my career.