The effect of invasive plant species on invertebrate biodiversity in Great Lakes coastal wetlands

Heather Siersma

Co authors: Wesley Johnson and Vince McKeon

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Abstract

This study investigates whether or not non-native graminoid invasions impact invertebrate biodiversity in Great Lakes coastal wetlands, and if so, whether those impacts are specific to or independent of the invasive plant species. Two invasive plants species were investigated with the establishment of four test areas, two for *Typha x glauca* with one containing the exotic and the other containing native vegetation. The same setup was done with *Phragmites australis*. Invertebrates were sampled using two techniques one of which captured mostly aerial and the other ground dwelling. Variation in order richness in *Typha* plots was found but no variation was found to exist in the *Phragmites* plots.

Introduction

The biodiversity of the state of Michigan is threatened as habitats have become fragmented and reduced in the wake of widespread logging and development over much of its lower peninsula (Penskar et al, 2001). Reductions in biodiversity have multiple deleterious effects in many areas, including damages to the economy and recreation, a lessening of human health, threats to human rights, and assaults on the intrinsic or spiritual value of nature (National Geographic Society, 2006). Awareness of these impacts can motivate individuals toward efforts to preserve biodiversity. For example, some would argue that maintaining biodiversity is essential for the development of new medications, foods, and other products with potential economic impacts; others would contend that the importance of preservation lies in the recreational opportunities that biodiversity within the environment can provide; still others would fight to preserve biodiversity to allow for the continuation of traditional lifestyles among indigenous peoples and of the vital processes of exchange among the flora and fauna themselves. Though their reasons for doing so vary, most people agree that it is important to try to prevent species extinction. According to recent public opinion polls, "… more than 60% of Americans describe themselves as active environmentalists or sympathetic to the environment…Americans overwhelmingly support our nation's major environmental

laws and more than 80% of Americans favor strengthening these environmental standards." (US Mission, 2006). Such statistics suggest that the American people are behind efforts to restore and preserve native biodiversity, thereby improving the natural quality of their environments.

Also important to improving the natural quality of Michigan environments, over 12,000 km of which are Great Lakes coastlines (Smith et al, 1991; Keough et al, 1999), is the maintenance of the Great Lakes coastal wetlands, which have escaped being drained for agriculture or other development (Dahl, 2000) only to be faced with threats from climate change, eutrophication in response to nutrient inflows increased by human activities, and the invasion of exotic plant species or genotypes (Goldberg, 2007). These wetlands not only play an ecological role in Great Lakes ecosystems by providing habitat and food for a variety of native plant and animal species, but they also play economic roles by protecting waterways for recreation and navigation and by serving to filter potential Great Lakes pollutants, including excess nutrients, from the waters which flow through them into those lakes (Mitsch and Gosselink, 2000; Zedler and Kercher, 2004; McClain et al, 2003; Krieger 2003; Mitsch and Wang, 2000). The degree of water quality degradation, particularly in terms of excess inputs of nutrients, has been shown to cause substantial changes in the richness, composition, and density of aquatic plant species in and around lakes (Lougheed et al, 2001; Toivonen and Huttunen, 1995; Bini et al, 1999: Magee et al, 1999). Further, studies have shown that where wetlands receive runoff from urban or agricultural landscapes, *Typha spp*. (cattails) and other invasive plants including *Phragmites australis* (common reed) and *Phalaris arundinacea* (reed canary grass), often displace the native vegetation (Woo and Zedler, 2002; Boutt et al, 2001; Wayland et al, 2002; Wayland et al, 2003; Duckles et al, in review).

The invasiveness of plant species depends both on plant traits and habitat modifications (Mooney et al, 1986; Galatowisch et al, 1999; Mack et al, 2000). Richardson et al (2000) defines invasive plants as those which "produce reproductive offspring, often in very large numbers, at considerable distances form parent plants (approximate scales: >100 m, <50 years for taxa spreading by seeds and other propagules; >6m / 3 years for taxa spreading by roots, rhizomes, stolons, or creeping stems), and thus have the potential to spread over considerable area" (p. 98). In Michigan

coastal wetland systems, prominently problematic emergent graminoids include *Typha*
Typha x glauca (a hybrid cattail formed from the invasive cattail *T. angustifolia* and the
native cattail *T. latifolia*), and the M Eurasian haplotype of *Phragmites australis*
(common reed) (Galatowitsch et al, 1999; Zedler and Kercher, 2004). These invaders
share the key traits of large size (1-4 m in height), rapid growth rate (up to 4X as
compared to the native), prodigious litter production (up to 14X as compared to the
native) extensive clonality (can spread large distances by rhizomes or rhizome fragments
and form dense monotypic stands) (Galatowisch et al, 1999; Grace and Harrison, 1986;
Mal and Narine, 2004; Zedler and Kercher, 2004; Herrick and Wolf, 2005; Salstonstall,
2002; USDA Forest Service, 2007; Rook, 2004; Driscoll, 1999; Howard et al, 2007,
Nancy Tuchman, personal communication). Additionally, the invaders respond quickly
to habitat nutrient additions which allows them to increase their primary productivity
over that of coexisting species thereby facilitating their dominance in nutrient enriched
wetland communities and a subsequent shift from native plant biodiversity to a non-
native vegetative monoculture (Tilman and Wedin, 1991; Miao and Sklar, 1998; Keddy,
1990; Woo and Zedler, 2002; Green and Galatawisch, 2001; Lavergne and Molofsky,
2004; Zedler and Kercher 2004, Schooler et al, 2006; Boers et al, 2007).

In addition to causing changes in biodiversity, non-native plant invasions can also
cause significant changes to a wetlands nutrient cycling processes by the increased
nutrient retention that their rapid substantial biomasses accumulation requires (Kercher
and Zedler, 2004; Ehrenfeld, 2003). These nutrient cycling changes can in turn produce
soil modifications in the invasion sites (Goldberg, 1990; Hobbie, 1992; Wilson and
Agnew, 1992; Lavorel & Garnier, 2002; Eviner & Chapin, 2003; Ehrenfeld, 2003). Such
plant-mediated environmental changes have been proposed to be responsible for
generating positive feedbacks which allow invading plant species to increase their own
populations as they shift the environment towards better meeting their needs over those
of their competitor species (Goldberg, 2007). However, it has also been proposed that
negative feedbacks, which cause the invader populations to decrease, are also generated
as these same environmental changes accumulate such that the environment becomes
substantially different than what was favorable for the initial invasion (Debra Goldberg,
personal communication). If the latter hypothesis is correct, the environmental changes

induced by one plant invasion may facilitate further invasions by other species, which could further reduce native biodiversity.

As is evidenced by the studies cited above, much investigation has been done on the impact of invasive plants on native plant biodiversity in wetlands and the resulting ecological and economic consequences these biodiversity impacts underlie in wetland systems. Few studies, however, have considered the impact plant invasions have on native animal biodiversity in wetland communities. According to the Intermediate Disturbance Hypothesis, small amounts of disturbance tend to increase biodiversity whereas too much disturbance tends to decrease it (Connell, 1978). Therefore, if plant invasion causes minor habitat disturbance, animal biodiversity in that habitat can be expected to increase; but if plant invasion causes major habitat disturbance, animal biodiversity in that habitat can be expected to decrease. Brown et al (2006) determined that invasive Eurasian purple loosestrife (*Lythrum salicaria*) negatively impacted the survival, development rate, and diet of American toad (*Bufo americanus*) tadpoles in North American freshwater wetlands, and they predicted the invader's impact on wetland ecological processes and aquatic food webs was more general and therefore of potential harm to other wetland amphibians. In contrast, Schwarz et al (2005) discovered increased diversity in tephritid fuitflies (*Rhagoletis pomonella*) following invasion by non-native honeysuckles (*Lonicera spp.)* in the northeastern United States, though this was due to rapid animal hybrid speciation. Gordon (1998) and Molnar (1990) each found evidence that invasions of non-indigenous species may increase both animal and plant species richness of native communities, at least in the short term. Gordon (1998) further suggested that our currently limited ability to predict and prevent negative impacts on native biodiversity of non-native plant invasions necessitates prioritizing research and management efforts in this area.

With this in mind, this study examines the invertebrate biodiversity in two Great Lakes coastal wetlands, comparing that discovered among sites with only native vegetation to that discovered in sites with either invasive *P. australis* or *T. x glauca* and otherwise similar environmental and vegetative conditions. Specifically, the study aims to address:

1) whether invasion by a non-native graminoid species with the abilities to produce large amounts of litter and shade, to quickly respond to nutrient enrichments, and to alter local soil characteristics and hydrology will produce an observable change in the biodiversity of invertebrates in Great Lakes coastal wetland systems; and

2) whether observed differences in invertebrate biodiversity are specific to or independent of the invading plant species.

Methods

Study Sites

Sturgeon Bay, located on Lake Michigan in northwestern lower Michigan's Emmet County 12 miles southwest of Mackinaw City, provided the sites used for comparing invertebrate biodiversity within and outside of invasive *P. australis* stands (see Fig 1 and Fig 2). Pointe La Barbe, located on Lake Michigan in the south central Upper Peninsula of Michigan's Mackinaw County 2 miles west of St. Ignace, provided the sites used for comparing invertebrate biodiversity within and outside of *T. x glauca* stands.

Vegetative and Environmental Condition Sampling

At each of the two sampling sites, stands of the invasive plant were located and five $1m^2$ quadrats were randomly placed within the stands. Five $1m^2$ quadrats were also randomly placed outside of the stands. The present plant species, their relative abundances, and the richness of the plant species within each quadrat was determined by on site identification of known plant species and by collection and subsequent keying of unknown plant species using the Herbarium at the University of Michigan Biological Station and dichotomous keys – primarily the *Manual of Vascular Plants of Northeaster United States and Adjacent Canada* (Gleason and Cronquist, 1991), the *Illustrated Companion to Gleason and Cronquist's Manual: Illustrations of the Vascular Plants of Northeastern United States and Adjacent Canada* (Holmgren, 1998), and *Michigan Flora*, parts I-III (Voss, 1972, 1985, 1996). and the Herbarium at the University of Michigan Biological Station.

Also measured in each quadrat were average leaf litter depth (the average of measurements made in each quadrat corner and in its center), % litter cover, % native v.

invasive litter cover, % light lost between the top of the vegetation and the litter layer (photometer reading at litter level subtracted from at eye level above vegetation, divided by reading at eye level), and distance of quadrat from open water, and quadrat elevation, latitude, and longitude (determined using GPS). Finally, soil cores were collected from the W and S corners of each quadrat for subsequent determination of % soil moisture (determined by massing before and after drying in a 105 degree Celsius oven overnight), % soil organic matter (determined by massing dry soil samples before and after burning off the organic material in a 550 degree Celsius muffle furnace for 2 hours), and soil pH (determined by mixing 20 grams of soil with 20 mL of deionized water, allowing the sediment to settle back out of water column, and measuring the water pH).

Invertebrate Sampling

The following steps were performed in each quadrat at each of the sampling sites:

1) Two 28 cm long x 14 cm wide pieces of bright yellow cardstock, coated on one side with Tanglefoot ©, was staked in the quadrat, one each in the N and S corners, such that they faced into the wind. At Pt. LaBarbe,the traps faced SW, and the northern trap was staked such that its base was 50 cm above the liter level in that corner while the southern trap was staked such that its base rested on the litter. At Sturgeon Bay the traps faced N and the southern trap was staked such that its base was 50 cm above the liter level in that corner while the northern trap was staked such that its base rested on the litter. The traps were collected from the field after three days.

2) A pitfall trap consisting of an 10 cm diameter, 625 cm^3 plastic cup buried to its rim in soil and partially filled with a liquid preservative (70% EtOH) was placed in the center of the quadrat (in the hole created from the sediment/litter sample) and collected after three days.

Invertebrates collected by each of the above described methods were keyed to order, and the counts from both the high and low Tanglefoot © traps as well as from the pitfall traps were combined to allow for data analysis of total measured invertebrate biodiversity in the four sampling sites.

Data Analysis

DCA was performed to analyze the similarity of the vegetation and of the environmental factors between the 20 replicates. DCA was also performed separately

between to 10 replicates at each site to analyze the invertebrate order richness between invasive + and − sites. Finally, Student's T tests were performed to test the significance of the differences in invertebrate numbers observed per order between *Typha* + and − sites and between *Phragmites* + and − sites.

Results

DCA of native vegetative similarity across sampling sites shows that the differences between sites were greater than differences within sites (see Fig 3). DCA of measured environmental factors supports the greater similarity within sites as compared to between sites. These results suggest that invertebrate biodiversity should be compared within sites rather than across sites.

DCA of invertebrate order richness inside and outside of a *Typha x glauca* stand shows that invertebrate order richness tended to be higher within the *Typha* stand than in the area lacking *Typha* (see Fig 4). Four of the five *Typha* + stands had species richness values ranging from 10.5 to 12.5 (outlier value = 9.5). Four of the five stands lacking Typha had species richness values ranging from 8.5 to 9.5 (outlier value = 10.5). DCA of invertebrate order richness at the Sturgeon Bay sampling site revealed species richness values ranging from 7-10.5 within the Phragmites stand and ranging from 7.5-11 outside of the stand.

The overlay of environmental factors onto the DCA of invertebrate order richness for *Typha* + and − sites revealed that major differences between the sites include soil moisture and average litter depth, both of which tended to be greater in the *Typha* stand (See Fig 5). Overlaying invertebrate orders onto the same DCA revealed that eight invertebrate orders were important in describing the differences in invertebrate order richness within and outside the *Typha* stand (see Fig 6). Running Student's T tests on log transformed data to compare the mean number of each of these eight invertebrate orders indicated that only the Aranea (df=8, p = 0.040) and Plecoptera (df=8, p = 0.040) numbers differed significantly between *Typha* + and - sites. Running the T test on untransformed data indicated that Plecoptera (df = 8, p = 0.037) and Diptera (df = 8, p = 0.035) numbers also differed significantly between these sites. Running this test on the log transformed data from within and outside the Phragmites stand indicated no

significant differences, while running it on the untransformed data set indicated that Aranea (df=8, p = 0.023) numbers differed significantly.

Discussion

The results of this study suggest that invasion by *Typha x glauca* is a minor disturbance to Great Lakes coastal wetland systems and therefore increases invertebrate biodiversity within that habitat. It further suggests that invasion by the M Eurasian haplotype of *Phragmites australis* does not produce an observable change in invertebrate biodiversity in these systems. Together, these results indicate that plant invasion induced differences in invertebrate biodiversity are specific to the invading plant species.

DCA of invertebrate order richness run independently for the Pt. La Barbe (see Fig 4) and the Sturgeon Bay sampling sites show that order richness tended to be higher for *Typha* + areas as compared to *Typha* − areas, but tended to be similar in both *Phragmites* + and *Phragmites* − areas. This suggests that while *Typha* stands are supportive of a more varied invertebrate community than are areas lacking *Typha*, *Phragmites* stands do not exhibit the same increased supportiveness. However, while increases in order richness also indicate changes to order composition, comparable order richness values are not descriptive of order composition. With this in mind, the raw data was revisited to examine whether or not comparable numbers of different invertebrate orders were indeed producing the observed species richness similarity inside and outside of *Phragmites* stands. While two invertebrate orders were found only within *Phragmites* stands and three were found only outside of those stands, four of these five order differences were produced by the presence of a sole invertebrate and the Student's T tests run confirmed their insignificance. More invertebrates of the fifth order, found only outside of *Phragmites* stands, were present, but the T test also revealed their presence to be insignificant. Thus, neither invertebrate richness or composition was significantly different between *Phragmites* + and − sites, though both tended to be higher in *Typha* + stands as compared to *Typha* − stands.

Evaluation of major environmental differences between the *Typha* + and − sites, revealed greater soil moisture and greater average plant litter depths in the *Typha* + stands than in the *Typha* − stands (see Fig 5). While *Typha x glauca* is believed to change local

hydrology in such a way that it dries out the areas it invades rather than increases soil moisture (Nancy Tuchman, personal communication), it has been shown to produce up to 14 times the amount of litter made by native cattails (Nancy Tuchman, personal communication). Therefore, the soil moisture differences between $Typha +$ and $-$ sites is more likely a factor underlying *Typha x glauca* invasion locations than it is a factor underlying changes to invertebrate biodiversity in existing *Typha x glauca* stands. The environmental pre-condition of high soil moisture then, and not the presence of Typha x glauca, could explain the increased presence of stoneflies (order Plecoptera), which are aquatic throughout their lifecycles (Merritt and Cummins, 1996), and of true flies (order Diptera), which are typically aquatic as juveniles (Merritt and Cummins, 1996), in Typha + sites as compared to Typha − sites. However, increased spider (order Aranea) presence in Typha + sites as compared to Typha − sites seems driven by increased litter depths as spiders are not typically aquatic. Perhaps the spiders follow their food sources, the dipterans, into the Typha stands. Interestingly, increased spider (order Aranea) presence was the only significant difference in invertebrate biodiversity between Phragmites. Perhaps the taller height of both invasives as compared to their native counterparts allows the spiders to construct higher webs and thereby catch food that would otherwise fly above their reach. More research is necessary to understand the reasons underlying the observed differences in spider numbers between invasive + and − stands.

   More research is also necessary to support the major claims of this study that invasion by *Typha x glauca* increases invertebrate biodiversity while invasion by the M Eurasian haplotype of *Phragmites australis* does not produce an observable change in invertebrate biodiversity. The sites from which these results were derived had average invasive plant covers of 6.2%, sd = 5.17%, and 20.8%, sd = 14.01%, respectively, and these results may be specific to these levels of invasion. For example, these low levels of *Typha x glauca* may be minor disturbances, but as hybrid *Typha* stands become densely monotypic, the invertebrate biodiversity within them may in fact decrease. Similarly, lower or higher densities of invasive *Phragmites* may actually impact invertebrate biodiversity.

   For the reason described above, repetitions of this experiment need to be performed at various invasive plant densities. Further, replicate similarity should be

better established prior to invertebrate sampling by doing stem counts rather than percent cover estimates of the native vegetation and by measuring environmental factors prior to invertebrate sampling.   Additionally, more invertebrates should be collected. This could be accomplished by increasing the number of replicates or by increasing the numbers and.or types of invertebrate traps used within each quadrat. For example, Tanglefoot © and pitfall traps could be reset multiple times, pitfall traps could be filled more frequently with EtOH or filled with a less volatile chemical to lessen the number of invertebrates which escape from them, and soil invertebrates could be collected using Tullgren funnels. Lastly, invertebrates could be identified more specifically, to family or genus, to better describe their biodiversity.

Overall, the findings of this study suggest that invertebrate biodiversity in Great Lakes coastal wetlands is indeed impacted by plant invasions, but that the impact is specific to the invading plant species. This reinforces the importance of effective invasive plant species management planning and implementation in the long-term preservation of Michigan's biodiversity and the maintenance and improvement of the natural quality of Michigan's environments.

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Figure Legend

Fig 1: GIS map of study sites

Fig 2: GIS map of study sites – close up

Fig 3: DCA of native vegetative similarity across sampling sites. Group 1 represents
Typha + replicate; group 2 represents *Typha* -replicates; group 3 represents
Phragmites + replicates; and group 4 represents *Phragmites* - replicates. Thirty
five total native plant species were found within the study's quadrats; however,
this figure shows only the 12 most important in describing the differences
between the sites. These are *V-Eleoch – Eleocharis palustris*, *V-Sacutu –
Schoenoplectus acutus*, *V-Jbrevi – Juncus brevicaudatus*, *V-Typha – Typha x
glauca*, *V-Jnodos – Juncus nodouis*, *V-Carexv – Carex viridula*, *V-Sciram –
Scirpus americanus*, *V-Potent – Potentilla anserina*, *V-Eupper – Eupatorium
perfoliatum*, *V-uforbs* – unkeyable forbs, *V-ugrams* – unkeyable graminoids, and
V-Phrag – Phragmites australis.

Fig 4: DCA of invertebrate order richness inside and outside of a *Typha x glauca* stand.

Fig 5: DCA analysis of invertebrate order richness overlaid with environmental factors.
Environmental factors measured include: E-wet – % soil moisture, E-.linv – %
invasive plant litter cover, E-orgC – % soil organic carbon, E-avgld – average
plant litter depth, E-PAR - % light lost between the top of the vegetation and the
litter layer, E-elev – elevation, E-pH – soil pH, E-dist – distance from quadrat to
Lake Michigan, E-.lnat - % native plant litter cover.

Fig 6: DCA analysis of invertebrate order richness overlaid with invertebrate orders.
Orders observed include: IArane = Aranea; IOrthp = Orthopoda; IPleco =
Plecypoda; IHymen = Hymenoptera; IDipter = Diptera; IColeo = Coleoptera;
IIsopo = Isopoda; and ISpiro = Spirobolida.
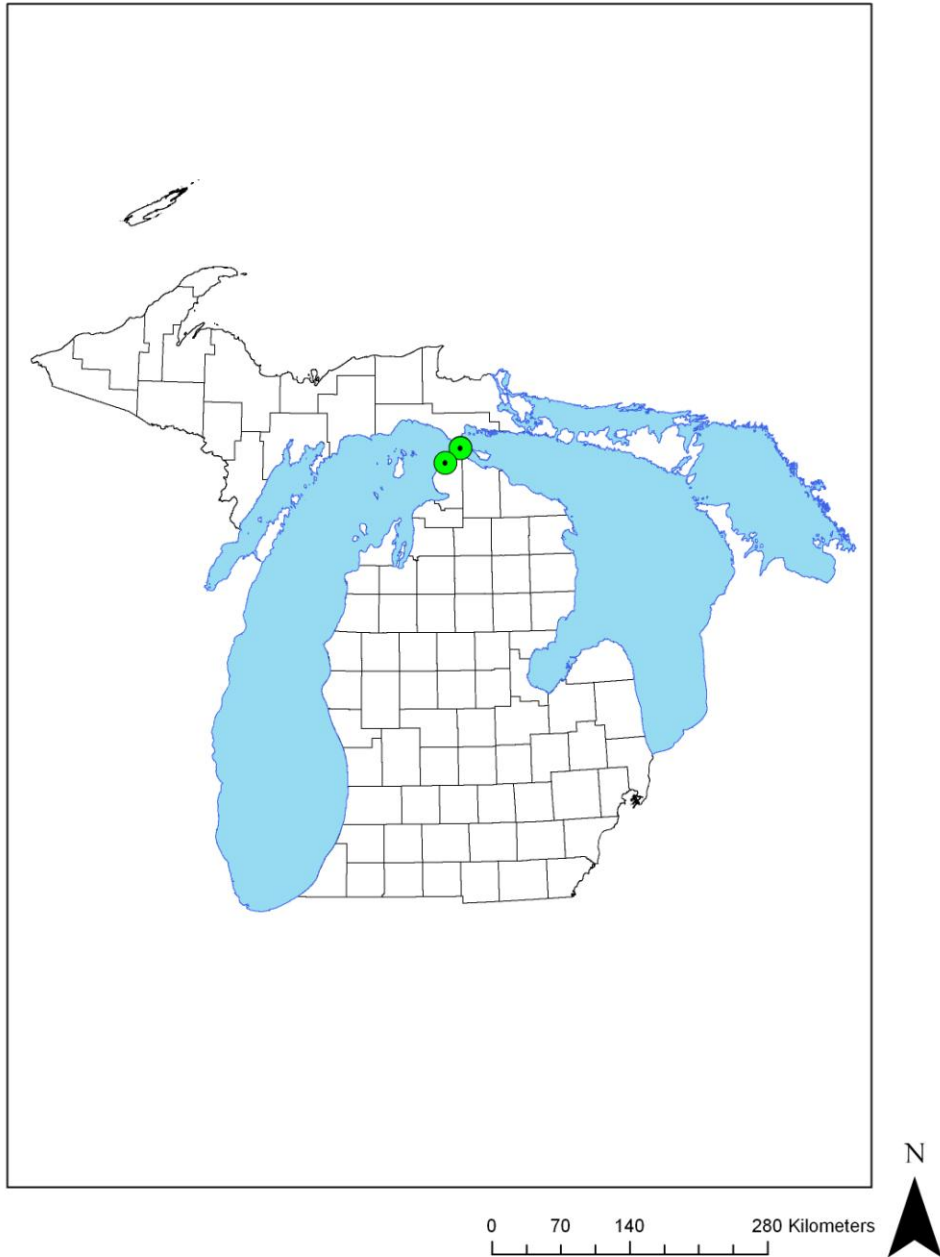
Figures

Fig 1

Study Sites
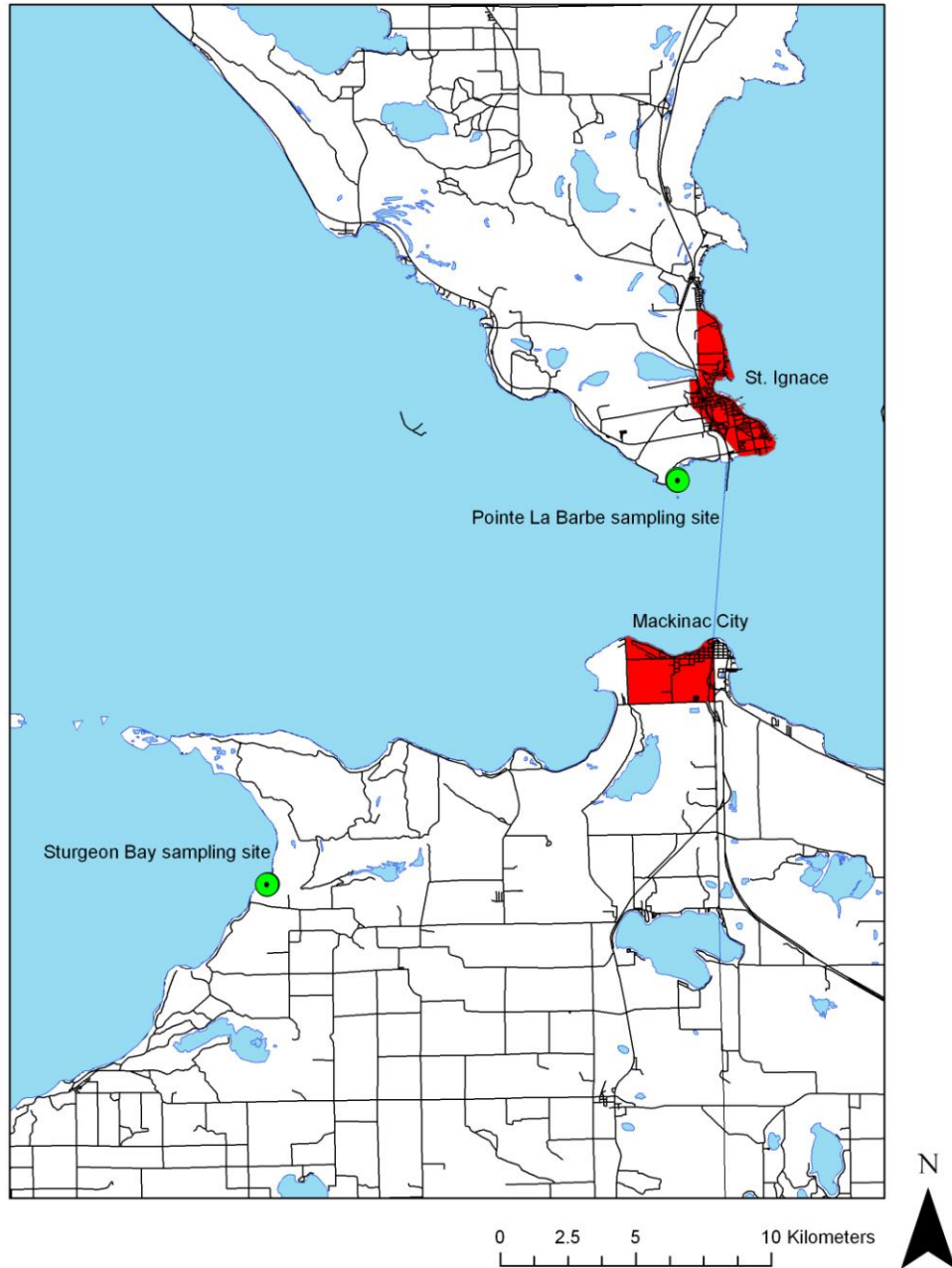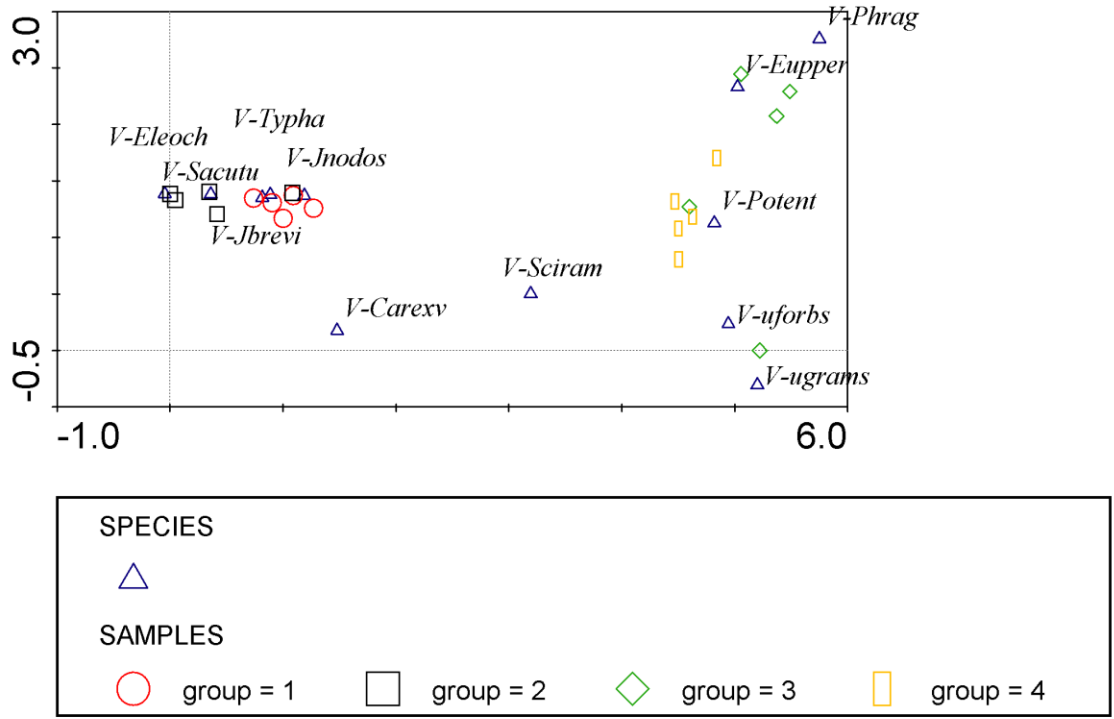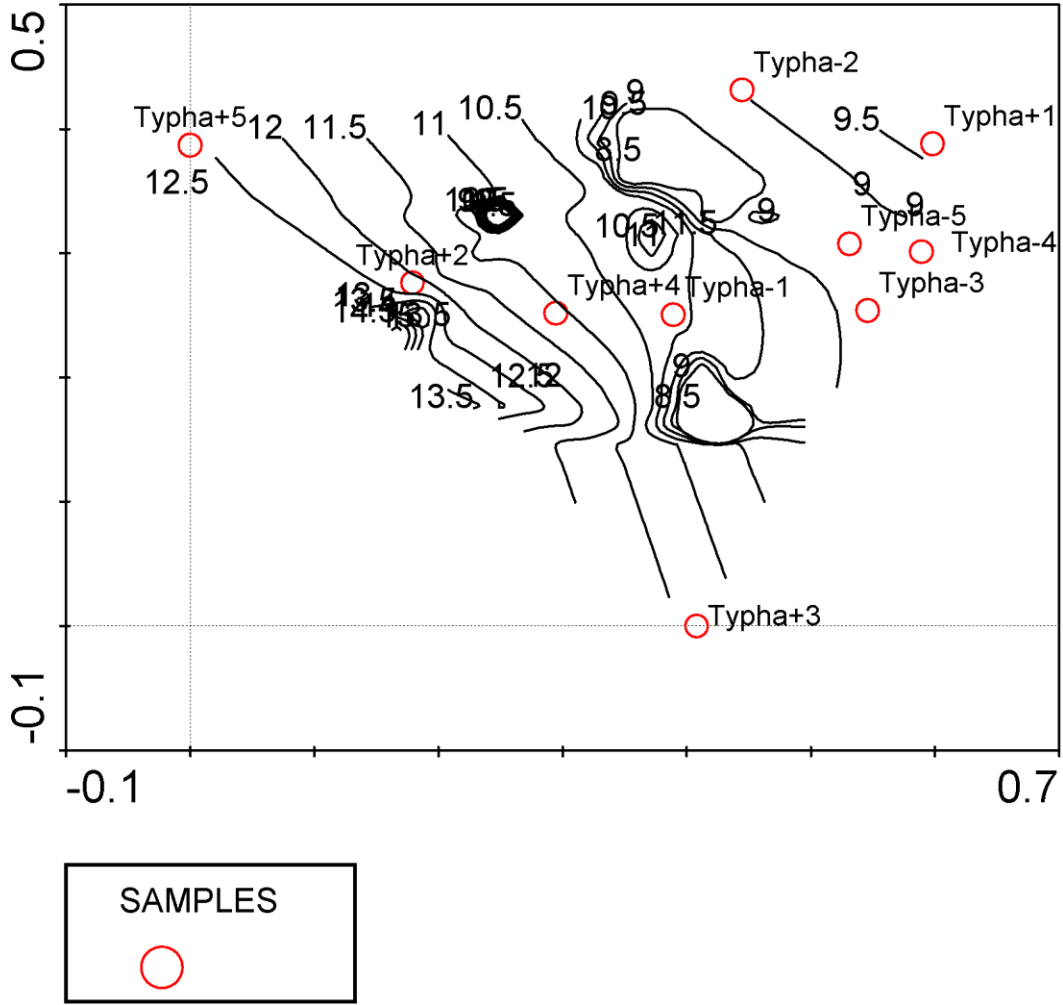
Fig 2

Study Sites- Close Up

St. Ignace

Pointe La Barbe sampling site

Mackinac City

Sturgeon Bay sampling site

N

0   2.5   5   10 Kilometers

Fig 3

Fig 4

Fig 5

Fig 6