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Lake Erie coastal marsh aquatic invertebrate community structure across habitats dominated by two different emergent macrophytes

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Lake Erie Coastal Marsh Aquatic Invertebrate Community Structure Across Habitats Dominated by Two Different Emergent Macrophytes

by

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Abstract

The objective of this observational study was to determine if there was a difference in aquatic invertebrate communities between areas dominated by *Phragmites australis* and areas dominated by *Typha spp.* in a freshwater coastal marsh. The hypothesis was that aquatic invertebrate diversity and abundance would be greater in *Typha*-dominated locations as opposed to *Phragmites*-dominated locations. Sampling took place at Lake Erie Metropark in southeast Michigan during the summer of 2013. Invertebrates were collected using Hester-Dendy samplers and identified in the laboratory. Invertebrates were assessed using the Shannon-Wiener Index, taxon richness, and abundance values which were all analyzed using t-tests and Mann-Whitney U-tests. Invertebrate community structure was analyzed using principal component analysis (PCA) and multivariate analysis of variance (MANOVA) was used to compare factor scores. Environmental variables of water temperature, pH, dissolved oxygen concentration, and percent dissolved oxygen saturation were measured and analyzed using repeated measures analysis of variance and Spearman correlations. There was no significant difference in invertebrate richness or diversity ($p>0.05$), nor were there any significant differences in the abundance of individual invertebrate taxa between the two plant types ($p>0.05$), except for *Helobdella modesta*, which was significantly more abundant in *Typha*-dominated areas ($p<0.05$). PCA of invertebrate taxa captured >70% of total variance in community structure in the first two factors (55.4% and 15.6%, respectively). Plotted sites on PCA axes showed no grouping patterns with respect to dominant plant species, suggesting invertebrate communities were not different based on plant type, and MANOVA confirmed the lack of groupings based on plant type ($p>0.05$). PCA suggested three groupings of invertebrate taxa which occurred together frequently. In regards to invertebrate functional feeding groups (FFG), there were no significant differences in mean FFG

abundance based on plant type (p always >0.05), except for the predator group, which was statistically greater at *Typha* sites ($p<0.05$). PCA of FFG captured >80% of total variance in community structure in the first two factors (46.1% and 34.3%, respectively). Plotted sites on PCA axes showed no grouping patterns based on plant type, suggesting FFGs were not different based on plant type, and MANOVA confirmed the lack of groupings ($p>0.05$). In conclusion, these findings suggest that freshwater *Phragmites* and *Typha* marshes are equally capable of supporting abundant and diverse aquatic invertebrate communities.

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Introduction

Coastal wetlands across North America are under siege by the invasive macrophyte *Phragmites australis*, also known as the common reed. *Phragmites australis* has been present in North America for at least 3,000 years in the form of two native subspecies (*Phragmites australis americanus* and *Phragmites australis berlandieri*), but during recent decades it has rapidly replaced other native flora (Osgood et al. 2003; Meyerson et al. 2010a). Previous research suggests that *Phragmites australis* invasion may be due to an aggressive genotype of the subspecies *Phragmites australis australis* from Europe (Able and Hagan 2000; Saltonstall 2002). This aggressive genotype likely originated from the United Kingdom and colonized North America as the result of multiple introductions over time, leading to genetic recombination and greater genetic diversity than native North American lineages (Hauber et al. 2011; Plut et al. 2011). A controlled study indicated that native and introduced subspecies of *Phragmites australis* successfully hybridized, allowing allelic recombination which could increase fitness of *Phragmites australis* in North American habitats (Meyerson et al. 2010b). Hybridized *Phragmites australis* are highly prevalent in the Great Lakes region and the northeast coast of New England (Paul et al. 2010). In contrast, other DNA analyses indicate that some native colonies maintain genetic diversity with no presence of hybridized *Phragmites australis* even in close proximity to the non-native stands (Saltonstall 2011). The differences in these studies reveal one of the challenges of determining the invasion process of *Phragmites australis.*

Phragmites australis invasion is catalyzed by disturbances such as restricted tidal flow, diking, and nitrogen eutrophication from shoreline development (Able et al. 2003; Fell 2006). These disturbances open a niche in the environment allowing *Phragmites australis* the opportunity to colonize and spread. This reed has high phenotypic plasticity, allowing it to

survive in changing habitats such as those subjected to anthropogenic disturbance (Hauber et al. 2011; Trebitz and Taylor 2007). *Phragmites australis* is a marsh species that is highly salt tolerant, meaning it has a greater capability of replacing less salt-tolerant plants in saltwater marshes with fluctuating salt concentrations and also in areas where salt concentrations are temporally high as a result of disturbance such as along roads, highways, and parking lots treated with deicing solutions during winter (Galatowitsch et al. 1999; Meyerson et al. 2000). Despite being most associated with altered habitats, *Phragmites australis* has also been documented in relatively undisturbed habitats. For example, *Phragmites australis* first became present on Hog Island, New Jersey, in 1971 and had replaced 83.1% of the island's native vegetation by 1999, even though this island is considered an undisturbed tidally-flooded marshland (Windham and Lathrop 1999).

In North America, *Phragmites australis* has been known to heavily out-compete other native plants and alter marsh structure. Biologically available nitrogen can be limiting in aquatic habitats, but in the rhizosphere of *Phragmites australis* oxidation is high and nitrogen readily attaches to the *Phragmites australis* rhizome (Meyerson et al. 2000). The rhizosphere is generally oxidizing, a strategy plants use to absorb nitrogen and prevent other plants from taking up the nutrient; in combination with the monotypic tendency of *Phragmites australis*, it is difficult for other plant species to get access to nutrients and to persist amongst a *Phragmites australis* monoculture (Meyerson et al. 2000). Research at the Mullica River showed that the soil in *Phragmites australis* marshes had higher redox potentials compared to nearby shortgrass marshes containing *Spartina patens* and *Distichlis spicata* (Windham and Lathrop 1999). The dense stands, tall stems, and leaf litter also reduce the amount of sunlight that reaches the water surface and marsh soil (Fell et al. 2003; Meyerson et al. 2000), which can inhibit the growth of

other marsh plants. This excess amount of leaf litter changes marsh topography by smoothing the surface and eliminating water-filled impressions common in marshes dominated by *Spartina spp.* (Able and Hagan 2000; Fell et al. 2003; Lathrop et al. 2003). Another reason why *Phragmites australis* is a successful invader is because it reproduces through rhizomes, and the stalks can easily recover from damage due to herbivory, fire, and clear-cutting (Meyerson et al. 2000). Data has shown that *Phragmites australis* is such a successful invader because it can reproduce through three mechanisms: 1) colonization of new patches through sexual seed production, 2) linear clonal growth of rhizomes which occurs along a preferred axis, and 3) circular clonal growth of rhizomes which can be random and non-directional (Lathrop et al. 2003). These biological characteristics of *Phragmites australis* increase its invasiveness in marsh habitats.

Replacement of native plants with *Phragmites australis* can cause dramatic changes in the native flora and possibly the fauna communities as well. Coastal marshes act as vital breeding grounds for many organisms because the areas between vegetation act as a safe place to deposit eggs and raise young (Able and Hagan 2000). Marshes are crucial nurseries for larvae, juveniles, and adults of smaller-bodied fish species that live and forage in these marshes year-round (Able and Hagan 2000). Because of the importance of coastal marshes to different life-stages of these organisms, conservation of marsh habitats and the biota within them is essential for preservation of the ecosystem. Alterations in marsh structure and function as a result of *Phragmites australis* introduction and invasion can have deleterious effects on the ability of a marsh ecosystem to provide adequate habitat for marsh species, which could cause irreversible trophic cascades within the ecosystem.

In a positive light, *Phragmites australis* has been known to produce habitat and food for birds, mammals, invertebrates, and some fishes, especially in Europe and areas in the U.S. that

have been affected by habitat destruction (Fell et al. 2003; Hunter et al. 2006; Gratton and Denno 2005; Hanson et al. 2002). In addition, the common reed has been cultivated and used for phytoremediation of wastewater to ameliorate eutrophication events and sequester nutrients, remove heavy metals, reduce erosion of shorelines, and combat sea level rise (Able and Hagan 2000; Hauber et al. 2011; Raichel et al. 2003). The presence of *Phragmites australis* reduces tidal flood frequency, flood depth, flood duration, and increases marsh elevation which work together to buffer the marsh from erosion and sea level rise (Osgood et al. 2003) that are inevitable outcomes of global climate change.

Due to the predominantly negative effects of *Phragmites australis* in North American marshes, efforts to combat the spread of the invasive *Phragmites australis* by means of biomass removal have been implemented across the United States, including the East Coast and Great Lakes region, to encourage regrowth of native plant populations and restore attractiveness and functionality of the wetlands (Kimball and Able 2007; Warren et al. 2001; Weis and Weis 2003). Removal strategies for *Phragmites australis* include the application of herbicides followed by the burning of dead aboveground biomass or harvesting of the plant matter (Fell et al. 2006; Hagan et al. 2007).

Much is known about the effects of *Phragmites australis* invasions on plant communities, and information on how plant communities respond to *Phragmites australis* control is becoming more available as research progresses. However, little to no research has been published on the effects of *Phragmites* on animal populations, such as invertebrates, in freshwater coastal marsh habitats including those containing the invasive *Phragmites australis* monocultures and locations consisting of other non-invasive plants. Previous research on the invasion of *Phragmites australis* on community structure is somewhat limited, but some trends

and patterns have been documented regarding the effects of expansion of common reed populations on faunal communities. There are few research papers relating faunal diversity and abundance to freshwater *Phragmites australis* and *Typha spp.* marshes, but research performed in saltwater marshes could provide further evidence for the differences in community structure across these two marsh types.

Phragmites australis invasion has been well-documented along the eastern coast of the U.S, including Connecticut, Delaware, Maryland, New Jersey, New York, South Carolina, and Virginia. In the marshes of the Mullica River of New Jersey, mummichog, *Fundulus heteroclitus*, and spotfin killifish, *Fundulus luciae*, were more abundant in habitats naturally dominated by *Spartina alterniflora* than *Phragmites australis* (Able and Hagan 2000; Able and Hagan 2003). Alternatively, there were only slight differences in biomass and abundance of fish species and benthic communities between *Phragmites australis* and *Spartina alterniflora* marshes within Chesapeake Bay (Meyer et al. 2001; Posey et al. 2003) and no differences in nekton use across plant type in the Charles Wheeler Salt Marsh on the lower Housatonic River estuary in Connecticut (Osgood et al. 2003). *Fundulus spp*. biomass and abundance were greater in *Spartina alterniflora/Typha*–dominated creeks than *Phragmites*–dominated creeks of the Lower Connecticut River (Warren et al. 2001). In the Hackensack Meadowlands of New Jersey, harpacticoid copepods, oligochaetes, ostracods, and sabellid polychaetes were significantly more abundant in *Spartina*-dominated habitats, while in the *Phragmites*-dominated habitat chironomids, gastropods, and gammarids were more abundant (Raichel et al. 2003). Another study of the Hackensack Meadowlands showed that nematodes, copepods, amphipods, mites, and all "other" invertebrates were more abundant at *Spartina alterniflora* sites than at *Phragmites australis* sites (Robertson and Weis 2005). The same study had field sites at Long

Island; results showed no difference in abundance across the different vegetation types for amphipods and mites, while nematodes, copepods, and "other" invertebrates were significantly more abundant in *Spartina alterniflora* sites than in *Phragmites australis* sites (Robertson and Weis 2005). Overall invertebrate density for both studies was greater in *Spartina alterniflora* sites than in *Phragmites australis* sites at all locations (Raichel et al. 2003; Robertson and Weis 2005). Macroinvertebrates are a large food source for marsh fishes, and alterations in macroinvertebrate populations could have cascading effects on fish populations. In general, the results from coastal wetland studies suggest overall faunal mean abundance and biodiversity is either reduced in the presence of *Phragmites australis* (Able and Hagan 2000; Able and Hagan 2003; Raichel et al. 2003; Robertson and Weis 2005; Warren et al. 2001) or that *Phragmites australis* causes no difference in faunal mean biomass or mean abundance (Meyer et al. 2001; Fell et al. 2003; Posey et al. 2003; Osgood et al. 2003).

A similar pattern can also be seen across habitats having different levels of invasion. In New England estuaries, mummichog and spotfin killifish populations were greatly reduced as *Phragmites australis* invasion stage increased; populations were highest in habitats with no presence of *Phragmites australis* and habitats recovering from *Phragmites australis* removal (Able et al. 2003; Hunter et al. 2006; Kimball and Able 2007). This suggests that restoration of the habitat from *Phragmites australis* dominance back to *Spartina alterniflora* or other native plant could reverse the effects of invasion on mummichog populations in the Delaware Bay area. Among *Typha angustifolia*-dominated, *Phragmites australis*-dominated, and treated *Phragmites australis*-dominated marshes along the lower Connecticut River, there was no significant difference in fish species composition (Fell et al. 2003). Another study of Alloway Creek showed weak differences in overall fish populations between sites dominated by *Phragmites australis*

and adjacent sites that had been treated, suggesting slow or very little response to removal (Grothues and Able 2003). In the Lieutenant River, a river that joins the Connecticut River before flowing into Long Island Sound, fish diversity and abundance did not differ between *Phragmites australis* sites and treated sites; however, *Phragmites australis* removal did have a negative effect on the abundance of mummichog (Fell et al. 2006). When observing the effects of *Phragmites australis* treatment in the lower Connecticut River and the Lieutenant River, invertebrate densities greatly varied with no clearly significant differences relative to plant type or control treatment (Warren et al. 2001). The lack of significant difference between treated and non-treated sites could be due to the removal of the *Phragmites* leaf litter which attracts macroinvertebrates that are a nutritious food source for mummichog and related species.

Research comparing invertebrates in freshwater sites dominated by *Phragmites* to unaffected sites is limited, but trends in freshwater marshes may differ from those of saltwater marshes. A study that focused on eight marshes along the southern coast of Lake Erie in Ohio observed benthic community structure of habitats dominated by *Phragmites australis*, *Typha spp.*, and native *Sagitarria spp.* and *Sparganium spp.* (Holomuzki and Klarer 2010). These results showed a positive relationship between benthic macroinvertebrate taxa diversity and the presence of *Phragmites australis*; greater benthic macroinvertebrate diversity associated with emergent macrophyte cover than floating macrophyte cover, and no influence of stem density on benthic macroinvertebrate diversity (Holomuzki and Klarer 2010). This means that the presence of reed-type macrophytes supported a more diverse community than floating-leaved macrophytes, sites where *Phragmites australis* was present supported the greatest invertebrate diversity, and benthic macroinvertebrate diversity was not a function of plant density. Macroinvertebrate density was slightly greater in *Phragmites australis* sites than sites with

Typha spp. and floating macrophytes; however, statistical analysis showed the average densities were not significantly different (Holomuzki and Klarer 2010). Epiphyton and diatoms, which are important food sources for macroinvertebrates, did not have different taxa diversities across plant types; however, densities were significantly greater in *Phragmites australis*-dominated sites than *Typha*-dominated sites (Holomuzki and Klarer 2010). Because the macroinvertebrate and epiphyton densities were greater at sites with *Phragmites australis*, we could infer that fish diversity and abundance, had it been measured, might also have been greater in *Phragmites australis* sites of the eight test marshes along Lake Erie's southern shoreline. These findings stand in contrast to much of the research previously discussed regarding the relationship between the presences of *Phragmites australis* and subaquatic community structure in saltwater marshes.

Leaf litter, a byproduct of both *Phragmites australis* and *Typha spp.*, constitutes a reliable food source for animals such as grazing invertebrates, and the species of the food source may not have a large impact on invertebrate abundance or community structure. Observing the activity of a specific freshwater macroinvertebrate, *Hyalella azteca* amphipods from Old Woman Creek National Estuarine Reserve, trends were seen in relation to the fungal community and the amount and type of leaf litter consumed. There was a weak, but significant, relationship between the amount of amphipod growth and biomass of litter-associated fungi regardless of the species of leaf litter (either *Phragmites australis, Typha angustifolia,* or treated *Phragmites australis*; Kulesza and Holomuzki 2006). This research suggests that as long as leaf litter, litter-associated fungi, and litter shredding amphipods are present that fish could also thrive in the presence of treated and untreated *Phragmites australis.* It was also observed that fungal biomass was greater and leaf toughness was lower in *Phragmites australis* leaf litter that had been treated with herbicide (Kulesza and Holomuzki 2006). Because the leaves of treated *Phragmites australis*

were less tough and more readily colonized by fungi, the researchers expected *Hyalella azteca* growth to be greater on the treated *Phragmites australis* as well; however, amphipod growth was not statistically significantly different across the plant types (Kulesza and Holomuzki 2006). This could suggest that the quality of the leaf litter or the presence of fungi may not actually be playing a role in amphipod abundance, but that some other variable could be driving amphipod abundance. Other invertebrates were not studied so this research does not give a community-wide view of the feeding association with plant type. This could also suggest that marshes rich in plant species which shed leaves at different times throughout the season could see a shift in abundance, being greater at one site during the early season to being more abundant at another site later in the season, which could only be observed using temporal analysis.

The goal of this study was to compare the diversity and abundance of the benthic macroinvertebrate community within Lake Erie Metropark, a freshwater marsh habitat, in sites dominated by two different vegetation types: *Phragmites australis* and *Typha spp*., in order to generate additional data on invertebrate responses to *Phragmites australis* in freshwater wetlands. Such information is needed, since the only previously-published study of this topic from Great Lakes wetlands (Holomuzki and Klarer 2010) found an unusual positive response of invertebrate diversity to *Phragmites australis*. Based on the generally negative responses of invertebrates to *Phragmites australis* in coastal marshes, I hypothesized that invertebrate community characteristics will differ across the two types of vegetation, with the expectation that *Typha spp*. sites will have greater density and diversity of invertebrates than sites dominated by *Phragmites australis.* A secondary goal of this work is to relate patterns in wetland invertebrate communities to site-specific environmental factors, such as temperature and oxygen availability.

Methodology

Site Selection:

Study sites were located within Lake Erie Metropark in south-eastern Michigan and consisted of isolated backwater channels lined by *Phragmites australis* and *Typha* spp. [Figure 1]. Sites were accessed by flat-bottomed jon boat powered manually by oars. Five locations within the metropark were categorized as *Phragmites*-dominated and denoted with the initial "P"; however, only four were described in the study due to the loss of sampling apparatus at one of the *Phragmites* locations. Five locations were categorized as *Typha*-dominated and denoted with the initial "T". Categorization of either P or T was determined based on the dominant plant type adjacent to each site. *Phragmites australis* within the marsh had been previously treated with herbicide in preparation for burning and harvesting, so some *Phragmites*-dominated were non-living, while all *Typha*-dominated sites consisted of living stands. Approximate water depth at each site was one meter to ensure submersion of the sampling apparatus.

Site P1 (N42 03.426, W083 11.511) consisted of living *Phragmites australis* monocultures, surrounded by nearly 35 meters of living *Phragmites* downstream, 24 meters of living stands upstream. P2 (N42 03.520, W083 11.601) was nestled in a monoculture of living *Phragmites australis* for 15 meters upstream and almost 17 meters downstream with a meter of mixed *Phragmites* and *Typha spp.* and continuous dead *Phragmites* further downstream. P4 was a mixture of treated and living *Phragmites australis* that was mostly dead treated stalks with some regrowth intermixed and more abundant green regrowth closer to the land. P4 (N42 03.825, W083 11.769) was flanked by approximately 16 meters of dead *Phragmites* upstream with some regrowth preceded by no regrowth further upstream, and 58 meters of dead

Phragmites downstream with some regrowth developing into living monocultures. P3 (N42 04.023, W083 11.848) was located in a separate intersecting channel on the other side of a vehicle bridge away from the other *Phragmites*-dominated sites. This location was originally dominated by dead, treated reeds but became inundated with lots of mixed regrowth over the season. P3 is flanked by approximately 89 meters of mixed regrowth downstream towards where the previous channel intersects and 120 meters of dead and living *Phragmites australis* with green monocultures further upstream towards a pedestrian bridge.

Typha-dominated sites were all located among stands of living *Typha spp.* T5 (N42 04.116, W083 11.862) was 64 meters upstream from a pedestrian bridge with approximately 30 meters of living *Typha spp.* stands further upstream for approximately 30 meters before transitioning to dead *Phragmites australis* monocultures. The remaining *Typha*-dominated sites were located downstream near an arched vehicle bridge where the channel opens up into Lake Erie. T3 (N42 03.889, W083 11.623) was placed nearly 94 meters upstream from a vehicle bridge with about 32 meters of green *Typha spp.* before continuing into dead *Phragmites australis* monocultures even further upstream. T1 (N42 03.858, W083 11.591) was located roughly 64 meters downstream from T3. T4 sat (N42 03.843, W083 11.562) nearly 15 meters downstream of the vehicle bridge with 22 meters of living *Typha spp.* leading into treated *Phragmites australis* towards the open waters of Lake Erie. T2 (N42 03.842, W083 11.595) is the only *Typha*-dominated site located on the opposite side of the channel from the other *Typha spp.* sites, 28 meters upstream of the vehicle bridge near the mouth of the channel. The distance downstream towards the bridge is dominated by living *Typha spp.* with dominance of *Phragmites australis* on the shoreline past the bridge. The shoreline upstream of T2 is dominated entirely by *Typha spp.*

Field Sampling and Data Collection

Hester-Dendy samplers were used to evaluate the invertebrate community. These samplers consist of layered plates, which allow invertebrates to attach and colonize within its layers. The samplers were anchored to metal stakes inserted into the substrate. Stakes were placed in the wetland channel two meters or less from the dominant emergent vegetation. This distance was dependent on water depth for placement. Samplers were suspended from the stakes in below the water surface but not deep enough in the water column to graze the sediment, typically in about 1 m of water. The Hester-Dendy samplers were deployed June 20, 2013 and collected July 22, 2013. The samplers were delicately lifted to the surface of the water where resealable plastic bags were used to carefully enclose the samplers, after which they were transported to the laboratory in the resealable plastic bags. In the laboratory, individual invertebrates were manually removed from the samplers without magnification, sorted into categories, and preserved in scintillation vials filled with 70% ethanol. Preserved invertebrate specimens were identified in the laboratory using a dissecting microscope to the lowest practical taxonomic level and enumerated. Pennak (1953) and Merritt et al. (2008) were the primary taxonomic references used. In addition to taxonomic identification, invertebrates were grouped together based on standard functional feeding groups (Barbour et al. 1999). Zooplankton species were not target organisms for this study, but some zooplankton were included in collections. These organisms, which consisted of ostracods and copepods, were excluded from all data analysis.

Water variables were recorded using a YSI multi-parameter sonde at each of the sites to determine variability across sites throughout the sampling season and to determine if environmental factors may have affected the invertebrate diversity, richness, and abundance data.

Temperature and pH were measured approximately twice weekly from June 20 to July 22, 2013. Dissolved oxygen (mg/L) and percent dissolved oxygen concentration were measured less frequently due to the availability of equipment on June 20, 21, 24, 25, and July 9, 2013. Measurements were recorded during the late morning to early afternoon each sampling day, and the order in which sites were measured each day was random.

Data Analysis

Taxonomic diversity for invertebrates was calculated using the Shannon-Wiener Index (H') for each site. The total H' for each plant type was calculated for *Phragmites*-dominated sites and *Typha*-dominated sites. The mean H' across replicates for each plant type was compared using a two-tailed separate variance t-test at $\alpha = 0.05$. Total taxon richness (R) of each site was recorded and mean taxon richness across replicates was calculated and analyzed using a two-tailed separate variance t-test at $\alpha = 0.05$ for all invertebrates to compare plant types. Mean abundance of each invertebrate category across sites was analyzed using non-parametric Mann-Whitney U tests at $\alpha = 0.05$ to determine if any differences in the abundance of individual taxon were present for *Phragmites australis*-dominated sites and *Typha*-dominated sites. Invertebrate taxa were also categorized based on primary functional feeding group, and these groupings were analyzed using non-parametric Mann-Whitney U tests at $\alpha = 0.05$ to observe potential differences based on plant type.

Invertebrate community structure was analyzed using principal components analysis (PCA) to visualize differences in community structure between *Phragmites*-dominated and *Typha*-dominated sites. Relative abundances of each invertebrate taxonomic category were expressed as percentages and PCA was conducted using correlation coefficients as the measure

of similarity, thereby weighting all invertebrate taxonomic categories equally. Both factor scores and factor coefficients were plotted to visualize patterns in community composition amongst sites and patterns in co-occurrence of invertebrate taxa. To determine if any consistent patterns were observed in PCA, factor scores were analyzed using a multivariate analysis of variance (MANOVA) with $\alpha = 0.05$ to compare *Typha* and *Phragmites* sites. Community structure was also analyzed according to functional feeding groups using PCA and MANOVA in the same manner by which the taxonomic categories were assessed.

Water temperature (C), pH, dissolved oxygen concentration (mg/L), and percent dissolved oxygen saturation were measured throughout the duration of the Hester-Dendy trap submersion. These variables were analyzed through time using repeated measures analysis of variance tests with $\alpha = 0.05$ and a Huynh-Feldt correction to determine if any significant differences occurred between *Phragmites australis* and *Typha spp.* sites which could have influenced the invertebrate diversity, abundance, or community structure. A Spearman correlation was also performed to determine the relationships between individual taxon and the mean value of each recorded environmental variable at each site.

Results:

Invertebrate Taxa Composition

Upon identification, invertebrates were categorized into the following 22 groups for analysis of richness, diversity, and community composition:

A: Amphipoda
AO: Annelida, Oligochaete
DCE: Diptera, Ceratopogonidae
DCH: Diptera, Chironomidae
DO: Decapoda, *Orconectes spp.*
E: Ephemeroptera
ES: *Erpobdella spp.*
F: Turbellaria flatworm
G: Gastropoda unknown
HG: Hemiptera, Gerridae
HP: Hemiptera, Pleidae
HM: *Helobdella modesta* (c.f. *stagnalis*)
I: Isopoda
L: Limpet spp.
OZ: Odonata, Zygoptera/Anisoptera
P: Planorbidae
PHG: Physidae, *Physa heterostropha/gyrina*
S: Sphaeriidae
T: Trichoptera
TH: Trombidiformes, Hydrachnidae
PR: Placobdella rugosa
VB: Viviparidae/Bithyniidae

The total taxon richness for all *Phragmites*-dominated sites was 18 and for *Typha*-dominated sites was 21, indicating a greater number of taxon were identified at *Typha*-dominated sites. The sum diversity values were H'=1.853 for *Phragmites*-dominated sites and H'=1.986 for *Typha*-dominated sites, suggesting overall diversity was greater at *Typha*-dominated sites. Statistical tests could not be performed using these data due to the lack of site replication, but the results of the total richness and overall diversity were used to visually assess these attributes across the two plant types.

Invertebrate taxon richness varied across each site [Figure 2]. The mean ($\pm$1 SD) taxon richness of *Phragmites*-dominated sites and *Typha*-dominated sites was 7.3$\pm$5.1 taxa per site and 11.6$\pm$3.0 taxa per site, respectively, resulting in a t-test output of t=-1.508, df= 4.578, p=0.197 [Figure 3], indicating there was no significant differences in the mean number of invertebrate taxa for each plant type.

Taxonomic diversity (H') was also calculated for each individual site using the Shannon-Weiner index [Figure 4]. These values show that T3 had the greatest H' while P6 had the lowest H'. Mean taxon diversity (±1SD) for *Phragmites* and *Typha* sites was 1.197±0.502 per site and 1.716±0.429 per site, respectively [Figure 5]. The t-test of H' resulted in t=-1.645, df= 5.995, p=0.151, indicating no significant difference in taxon diversity across plant type.

Total numbers of benthic invertebrates were 221 individuals collected from *Phragmites*-dominated sites and 444 individuals collected from *Typha*-dominated sites, but these raw values do not suggest a real difference in total abundance because there were more *Typha* sites than *Phragmites* sites sampled. The mean (±1SD) abundance of invertebrates for *Phragmites* sites was 55.3±43.5 individuals per site, and the mean abundance (±1SD) for *Typha* sites was 88.8±47.1 individuals per site with t-test results of t= -1.097, df= 7, p=0.309, indicating that plant type did not affect overall invertebrate abundance [Figure 6].

Individual invertebrate taxonomic categories were assessed across each site as a sum of site type. Both the total number of each organism at each site type [Figure 7] and mean (±1SD) of each invertebrate for *Phragmites* sites and *Typha* sites were calculated [Figure 8]. Abundances of each invertebrate taxonomic category are discussed in the following text.

Only one crayfish, *Orconectes spp.*, was captured which was at P2, although other decapod crustaceans were observed in the area. Results of non-parametric testing showed p=0.264, indicating the presence of this one crayfish was not a significant find.

Total amphipods sampled at *Phragmites* sites and *Typha* sites were 9 and 10, respectively, with the most amphipods being sampled at P4 and no amphipods present at P1, P3, and T5. Mean (±1SD) amphipod abundance at *Phragmites* sites was 2.3±3.9 individuals per site

and 2.0±1.9 individuals per site at *Typha* sites. A Mann-Whitney U-test revealed p= 0.610. This p-value suggests no significant difference in amphipod abundance across the two plant types.

Isopods were present at a total of 18 individuals at *Phragmites* sites and 72 individuals at *Typha* sites, with the most collected from T2 and none collected from P1 and P3. Mean (±1SD) number of isopods from *Phragmites* sites was 4.5±6.6 individuals per site and the mean abundance of *Typha* sites was 14.4±9.6 individuals per site. A Mann-Whitney U-test resulted p=0.085, suggesting no significant difference between plant types.

Annelids comprised 111 individuals captured at *Phragmites* sites and 195 individuals at *Typha* sites, with the highest number of annelids sampled at T3 and none collected from P3. The mean (±1SD) abundance of annelids at *Phragmites* sites was 27.8±19.8 individuals per site and a mean of 39.0±37.6 individuals per site at *Typha* sites. A Mann-Whitney U-test resulted in p=0.061, indicating no significant difference in annelid abundance based on plant type.

Flatworms were collected from P4, the only *Phragmites* site to have these organisms, with a total of 3. Flatworms at *Typha* sites totaled 13 with the most collected from T4 and none from T2 or T5. Mean (±1SD) abundance of flatworms at *Phragmites* sites was 0.8±1.5 individuals per site with mean abundance at *Typha* sites being 2.6±3.0 individuals per site. A Mann-Whitney U-test resulted in p=0.283 indicating that there was no difference between *Typha*-dominated and *Phragmites*-dominated sites.

Three taxa of leeches were identified in the samples. *Erpobdella spp.* was present as 4 individuals at *Phragmites* sites and a single individual at T2. Mean (±1SD) abundance of *Erpobdella spp.* was 1.0±0.8 individuals per site at *Phragmites* sites and 0.2±0.4 per site at *Typha* sites. These abundances were not significantly different between plant types (p= 0.100). *Helobdella modesta* (c.f. *stagnalis*) was present in *Phragmites australis* sites as 2 individuals

from P2 and 1 from P3, while 23 individuals of this species were collected from *Typha*-dominated sites, with the greatest number found at T1 with a total of 9 individuals. Mean (±1SD) *H. modesta* abundance at *Phragmites* sites was 0.8±1.0 individuals per site and at *Typha* sites was 4.6±3.0 individuals per site. A Mann-Whitney U-test indicated these abundances were statistically significantly different, indicating *H. modesta* had a greater mean abundance in *Typha* sites than *Phragmites australis* sites (p=0.024). The final leech species, *Placobdella rugosa*, was present as one individual from P2 and 3 individuals from T2. *P. rugosa* had a mean (±1SD) abundance of 0.3±0.5 individuals per site in *Phragmites australis*-dominated sites and a mean (±1SD) abundance of 0.6±1.3 individuals per site in *Typha*-dominated sites. These abundances did not test as significantly different (p=0.100). Combined, leeches contributed 8 individuals for *Phragmites* sites and 27 individuals for *Typha* sites. The mean (±1SD) of total leeches for *Phragmites* sites was 2.0±1.6 individuals per site and for *Typha* sites was 4.6±2.6 individuals per site. The abundance of total leeches was not significantly different (p=0.125).

Only two Ceratopogonids were captured in *Phragmites australis*-dominated sites at P2, while 2 were collected at T1 and 2 from T4 for a total of 4 individuals in *Typha*-dominated sites. The mean (±1SD) number of Ceratopogonids for *Phragmites* sites was 0.5±1.0 individuals per site, and 0.8±1.1 individuals per site at *Typha* sites. A Mann-Whitney U-test suggested no difference between plant types for these organisms (p=0.655). Chironomids were more abundant with 20 sampled from *Phragmites* sites and 39 sampled from *Typha* sites, with the most collected at T4 and none collected from P2 and P3. Mean (±1SD) Chironomid abundance at *Phragmites* sites equaled 5.0±6.6 individuals per site and the mean (±1SD) at *Typha* sites was 7.8±5.4 individuals per site. A Mann-Whitney U-test shows that the differences observed for Chironomids was not statistically significant difference between *Phragmites*-dominated sites and

Typha-dominated sites (p= 0.264). Total Dipterans (Ceratopogonids and Chironomids combined) equaled 22 and 43 at *Phragmites*-dominated sites and *Typha*-dominated sites, respectively. Mean (±1SD) Dipteran abundance at *Phragmites*-dominated sites was 5.5±6.2 individiuals per site while mean (±1SD) Dipteran abundance at *Typha*-dominated sites was 8.6±6.2 individuals per site. Total Dipterans abundance was not significantly different based on plant type (p= 0.389).

A total of 26 Ephemeroptera individuals were collected at P4, the only *Phragmites* site to have these organisms, and a total of 29 from *Typha* sites with the most sampled from T4 with 22 individuals. Mean (±1SD) Ephemeropteran abundance at *Phragmites* sites equaled 6.5±13.0 individuals per site and 5.8±9.1 individuals per site at *Typha* sites. No significant difference was observed for these invertebrates across plant type (p= 0.368).

Two groups of Hemipterans were collected, organisms from the families Gerridae and Pleidae. In the Gerridae family, no individuals were collected at *Phragmites* sites and only one individual was collected in *Typha* sites at T5 for a mean (±1SD) at *Typha* sites of 0.2±0.4 individuals per site. A Mann-Whitney U-test suggests the presence of this one organisms was not statistically significant (p=0.371). In the Pleidae family, none were collected at *Phragmites* sites, and four were collected in *Typha* sites split between T2 and T3 for a mean (±1SD) at *Typha* sites of 0.8±1.1 individuals per site. A significant difference in the mean number of these organisms collected was not observed (p=0.176). No Hemipterans were sampled from *Phragmites australis* sites, while a total of 5 were collected from *Typha* sites for a mean (±1SD) at *Typha* sites of 1.0±1.0 individuals per site. The difference in the capture abundance of total Hemipterans was not statistically significant either (p=0.079).

Odonata frequented *Typha* sites with a total of 12 individuals and only 2 individuals at *Phragmites* sites. Mean (±1SD) Odonata abundance at *Phragmites* sites was 0.5±0.6 individuals

per site and a mean (±1SD) abundance at *Typha* sites was 2.4±2.9 individuals per site. These abundances are not statistically different across the two plant types (p= 0.304).

Trichoptera larvae were rarely observed, with only 1 individual present in *Phragmites australis* sites at P4, and only 3 individuals in *Typha* sites at T3. Abundance of Trichoptera was not statistically different across plant types with a *Phragmites* site mean (±1SD) abundance of 0.3±0.5 individuals per site and a *Typha* site mean (±1SD) abundance of 0.6±1.3 individuals per site (p= 1.000).

Members of the Hydrachnidae family were also rarely captured, with 2 individuals obtained from P4 and 1 individual obtained from each of T2, T3, and T5. Mean (±1SD) Hydrachnid abundance in *Phragmites* sites was 0.5±1.0 individuals per site and 0.6±0.5 individididuals per site at *Typha* sites. With such a low sampling size, analysis determined these abundances to not be statistically significant (p= 0.584).

A single unidentified gastropod snail was sampled from T2 which was not found at any other site. The presence of this organism was deemed statistically insignificant using a Mann-Whitney U-test (p=0.371). Limpets were collected exclusively in *Typha*-dominated sites totaling 13 individuals with the most sampled from T4. Statistical analysis revealed the abundance of these organisms was not significantly different across plant types (p=0.081). Members of the Planorbidae family were present as 4 individuals from P8 and one individual from each T1 and T2. Planorbid mean (±1SD) abundance at *Phragmites* sites was 1.0±2.0 individuals per site and mean (±1SD) abundance at *Typha* sites was 0.4±0.5 individuals per site. Planorbid abundance did not test to be significantly different across plant types (p= 0.884). Snails identified as members of the Physidae family were present at P2 and P4 for a total of 3 organisms in *Phragmites australis*-dominated sites, and were present at T1, T2, and T3 for a total of 13

individuals in *Typha*-dominated sites. Mean (±1SD) *Phragmites* site abundance of Physids was 0.8±1.0 individuals per site and mean (±1SD) *Typha* site abundance was 2.6±2.6 individuals per site. Similarly, abundance did not differ across plant types (p= 0.306). Individual snails that were either family Bithyniidae or Viviparidae were grouped together for a total of 4 individuals in *Phragmites australis* sites from the singular P4 site while 13 were gathered from *Typha* sites with the greatest abundance of 7 individuals at T4. Mean (±1SD) abundance of these organisms at *Phragmites* sites was 1.0±2.0 individuals per site and at *Typha* sites was 2.6±3.0 individuals per site. Abundance was not statistically significant between *Phragmites australis* and *Typha spp.* sites for this gastropod category (p= 0.345). The final type of gastropod collected was determined to be of the family Sphaeriidae. A total of 7 Sphaeriids were identified in *Phragmites australis* sites at P2 and P4, while a total of 3 were identified in *Typha* sites split across T1, T3, and T4. Mean (±1SD) abundance of Sphaeriids collected from *Phragmites* sites was 1.8±2.9 individuals per site and from *Typha* sites was 0.6±0.5 individuals per site. Parallel to the other gastropod groups, there was no statistically significant difference in the abundance of Sphaeriids across plant types (p= 0.893). As individual categories, there are no statistically significant differences between *Phragmites*-dominated sites and *Typha*-dominated sites. When looking at total gastropods, *Typha* sites had a sum of 45 individuals and *Phragmites* sites had a sum of 18 gastropods. Mean (±1SD) abundance of total gastropods at *Phragmites* sites was 4.5±5.3 individuals per site and the mean abundance of total gastropods at *Typha* sites was 9.0±6.4 individuals per site. As an all-encompassing gastropod group there was still no significant difference in the abundance across plant types (p=0.213).

Invertebrate taxa were also categorized based on primary functional feeding group. Five functional feeding groups were identified: filter/collector (FC), gatherer/collector (GC), predator

(PR), scraper (SC), and shredder (SH). Total number of FC at *Phragmites* sites was 7 individuals and at *Typha* sites was 3 individuals. Mean abundance of FC at *Phragmites* sites was 1.8±2.9 individuals and mean abundance of FC at *Typha* sites was 0.6±0.5 individuals. There was no significant difference in abundance of FC based on plant type (p=0.893). Total number of GC at *Phragmites*-dominated sites and *Typha*-dominated sites was 185 individuals and 348 individuals, respectively. Mean (±1SD) abundance of GC at *Phragmites* sites was 46.3±37.0 individuals and mean abundance at Typha sites was 69.6±43.7 individuals. The abundance of GC was not significantly different based on plant type (p=0.327). GC was the dominant functional feeding group in both *Phragmites australis* and *Typha spp*. locations. Total number of PR at *Phragmites* sites was 17 individuals and was 64 individuals at *Typha* sites. Mean (±1SD) abundance at *Phragmites*-dominated sites and *Typha*-dominated sites was 4.3±2.1 individuals and 12.8±5.3 individuals, respectively. A significant difference in abundance was observed for PR based on plant type (p=0.049). Total SC captured at *Phragmites* sites was 11 individuals with 29 individuals collected from *Typha* sites. Mean (±1SD) abundance of SC at *Phragmites* sites was 2.75±4.3 individuals and mean (±1SD) abundance at *Typha* sites was 5.8±6.0 individuals. SC abundance was not significantly different between plant types (p=0.455). Only 1 individual fell into the SH category which was observed at site P3. Mean (±1SD) abundance of SH at *Phragmites* sites was 0.25±0.5 individuals and 0.0±0.0 individuals at *Typha* sites. The presence of this single organism did not produce a statistically significant difference between the two plant types (p=0.264).

Community Composition

The first two PCA factors explained >70% percent of the total variance in benthic invertebrate community composition based on taxonomy (55.4% and 15.6%, respectively) and were thus used to analyze the invertebrate community structure of the different sampling sites. The factor coefficients obtained suggest three groups of co-occurring taxa [Figure 9]. The first grouping located along the positive factor 1 axis includes the invertebrate categories Hemiptera Pleidae (HP), Odonata Zygoptera/Anisoptera (OZ), unknown Gastropod (G), members of the families Viviparidae and Bithyniidae (VB), and *Placobdella rugosa* (PR). The second grouping located along the negative factor 1 axis includes the invertebrate categories Hemiptera Gerridae (HG), *Erpobdella spp.* (ES), *Helobdella modesta* (c.f. *stagnalis*) (HM), Annelida Oligochaetes (AO), Diptera Chironomidae (DCH), and Ephemeroptera (E). The third grouping which is located along the negative factor 2 axis includes the invertebrate categories Decapoda *Orconectes spp.* (DO), Sphaeriidae (S), Flatworms (F), Diptera Ceratopogonidae (DCE), Limpets (L), and Amphipods (A). The lack of a grouping pattern of plotted sites with respect to dominant plant present suggested that invertebrate communities were not different based on vegetation type as the grouping factor [Figure 10]. A MANOVA performed using the factor scores of the first two factors obtained from PCA indicated no distinct grouping of the factors based on the effect of the grouping variable which was plant type (Pillai Trace F= 0.450, df= 2,6, p= 0.658), which supports the visual observations from the plot of the factor scores. Comparing the factor scores and factor coefficient plots, some patterns are observed. Site T2 was rich in the positive factor 1 taxa group with these grouped taxa found more frequently together. T5 and P1 were rich in the second taxa group, while P2 and T1 were visually richer in the taxa found in the third group.

The first two factors explain >80% of the variance in community composition (46.1% and 34.3%, respectively) based on categorization into functional feeding groups and were used to further explore the relationship between functional feeding group and dominant marsh plant type. The factor coefficients suggest that the placement of each feeding group is separate from the others except for SH and FC, which occur on the axes very close together [Figure 11]. The factor scores of Factor 1 and Factor 2 were plotted and suggest that seven of the nine sampling locations were grouped together along the negative Factor 2 axis while P3 and P2 were separated along the positive Factor 2 axis at opposite extremes of the Factor 1 axis [Figure 12]. Results of the MANOVA using the factor scores of Factor 1 and Factor 2 indicated no significant grouping of the factors based on the effect of the grouping variable which was plant type (Pillai Trace F=1.182, df= 2,6, p= 0.369).

Water Environment Variables

Temperature (C), pH, dissolved oxygen concentration (mg/L) and percent dissolved oxygen saturation were analyzed to observe potential differences across sites and as a factor of plant type. Mean ($\pm$1SD) temperatures of *Phragmites* sites were P1= 25.0$\pm$2.4C, P2= 25.4$\pm$2.5C, P3= 24.9$\pm$1.6C, P4= 25.3$\pm$2.1C [Figure 13]. The mean of the mean ($\pm$1SD) *Phragmites* site temperatures was 25.2$\pm$0.2C [Figure 14]. Mean ($\pm$1SD) temperatures of *Typha* sites were T1=25.1$\pm$2.0C, T2=25.5$\pm$1.6C, T3=25.0$\pm$1.8C, T4= 25.4$\pm$2.0C, T5= 24.3$\pm$1.9C [Figure 13]. The mean of mean ($\pm$1SD) *Typha* site temperatures was 25.1$\pm$0.5C [Figure 14]. A repeated measures analysis of variance showed there was no significant difference in temperatures between the two site types (p=0.756). This test revealed there was a significant change in temperature over time (p<0.001) but that the different sites did not have different

patterns of change over the course of sampling (p=0.096). A Spearman correlation determined there was no significant correlation between mean water temperature and the mean abundance of any of the identified taxa (p always >0.05). When observing herbicide treatment of *Phragmites* sites, no observable trends were seen between herbicide treatments (P3 and P4) and untreated *Phragmites*-dominated sites (P1 and P2) [Figure 13].

Mean (±1SD) pH readings of *Phragmites* sites were P1=7.3±0.2, P2= 7.4±0.1, P3= 7.1±0.2, P4= 7.4±0.2 [Figure 15]. The mean of the mean (±1SD) *Phragmites* site pH readings was 7.3±0.1 [Figure 16]. Mean (±1SD) pH readings of *Typha* sites were T1=7.6±0.4, T2=7.6±0.4, T3=7.4±0.6, T4= 7.7±0.5, T5= 7.1±0.2 [Figure 15]. The mean of mean (±1SD) *Typha* site pH readings was 7.5±0.2 [Figure 16]. A repeated measures analysis of variance showed there was no significant difference in pH readings between the two site types (p=0.165). This test revealed there was a significant change in pH over time (p<0.001) and that the *Phragmites* sites had different patterns of change than the *Typha* sites over the course of sampling (p=0.002). A Spearman correlation determined a significant positive correlation between mean pH and mean abundance of one taxon, total gastropods, which suggested that the sites with higher pH had higher mean gastropod abundance (r_s=0.797, p<0.05). There was also a significant positive correlation between mean sum of all taxa and mean pH (r_s=0.717, p<0.05). When observing treatment of *Phragmites* sites, no observable trends were seen between herbicide treatment (P3 and P4) and untreated sites (P1 and P2) [Figure 15].

Mean (±1SD) dissolved oxygen (mg/L) of *Phragmites* sites were P1=3.4±1.1 mg/L, P2= 3.8±0.8 mg/L, P3= 4.3±2.4 mg/L, P4= 4.05 ±1.3 mg/L [Figure 17]. The mean of the mean (±1SD) *Phragmites* site dissolved oxygen was 3.9±0.4 mg/L [Figure 18]. Mean (±1SD) dissolved oxygen of *Typha* sites were T1=4.9±3.1 mg/L, T2=6.4±3.2 mg/L, T3= 6.1±4.5 mg/L,

T4= 6.4±3.7 mg/L, T5= 5.3±1.2 mg/L [Figure 17]. The mean of mean (±1SD) *Typha* site dissolved oxygen was 5.8±0.7 mg/L [Figure 18]. A repeated measure analysis of variance showed there was a significant difference in dissolved oxygen (mg/L) between the two site types (p=0.001) with *Typha*-dominated sites having higher mean dissolved oxygen than *Phragmites*-dominated sites. This test revealed there was a significant change in dissolved oxygen (mg/L) over time (p<0.001) but that patterns of change between the two plant types did not significantly differ over the course of sampling (p=0.064). A Spearman correlation determined no significant correlation between mean dissolved oxygen (mg/L) and any of the taxa mean abundances (p always >0.05). When comparing herbicide treated *Phragmites australis* sites (P3 and P4) to untreated sites (P1 and P2), treated sites appeared to have greater variability but no differences were observed [Figure 17]. Mean (±1SD) percent dissolved oxygen saturation of *Phragmites* sites were P1=41.1±12.5%, P2= 46.0±9.9%, P3= 50.5±26.5%, P4= 48.6 ±13.9% [Figure 19]. The mean of the mean (±1SD) *Phragmites* site percent dissolved oxygen saturation was 46.5±4.1% [Figure 20]. Mean (±1SD) percent dissolved oxygen saturation of *Typha* sites were T1=57.7±36.9%, T2=77.6±38.2%, T3= 70.2±51.0%, T4= 78.0±49.3%, T5= 61.8±14.4% [Figure 19]. The mean of mean (±1SD) *Typha* site percent dissolved oxygen saturation was 69.1±9.2% [Figure 20]. A repeated measures analysis of variance suggested there was a significant difference in percent dissolved oxygen concentration between the two site types (p=0.003) with *Typha*-dominated sites having higher mean percent dissolved oxygen saturation than *Phragmites*-dominated sites. This test revealed there was a significant change in percent dissolved oxygen concentration over time (p=0.001) and that the patterns of change seen in *Phragmites* sites was significantly different than *Typha* sites over the course of sampling (p=0.040). A Spearman correlation determined no significant correlation between mean percent

dissolved oxygen saturation and any of the taxa mean abundances (p always >0.05). Treated *Phragmites* sites (P3 and P4) appeared to have greater variability than untreated sites (P1 and P2), but no differences in mean percent dissolved oxygen saturation were observed [Figure 19].

Discussion

Invertebrate Richness and Diversity

My research indicated there were no statistically significant differences in mean invertebrate richness or diversity for *Phragmites*-dominated sites and *Typha*-dominated sites. Total invertebrate taxon richness was greater for *Typha spp.* sites than *Phragmites australis* sites with 21 and 18 taxa, respectively. However, more sites were sampled in *Typha*-dominated marshes than *Phragmites*-dominated marshes, which may explain the greater richness. Similar results occurred with invertebrate taxa diversity (H'). Overall H', the diversity index for each site type inclusive of all sites within that type, was greater for *Typha spp.* sites than *Phragmites australis* sites. Statistical analysis could not be performed on these data due to a lack of site replication as a result of the limited accessibility and the availability of manpower and sampling equipment, but had yearly replicates been taken, there could be observable trends toward greater sum richness and diversity for one plant type over the other. Data analysis of the mean richness and diversity of invertebrates showed there were no statistically significant differences in average invertebrate richness and diversity for *Phragmites*-dominated sites and *Typha*-dominated sites. The results of this study suggest that *Phragmites australis*-dominated marshes and *Typha*-dominated marshes are equally capable of supporting similar communities of surface-associated aquatic invertebrates in terms of richness and diversity.

Previous research of coastal saltwater marshes revealed conflicting evidence regarding differences and similarities of invertebrate communities in coastal marshes; in general, invertebrates were either negatively or unaffected by *Phragmites australis* in such habitats. Total invertebrate richness, which included many of the same taxa collected at Lake Erie Metropark, was not significantly different between *Phragmites*-dominated and *Spartina*-dominated sites in the Charles Wheeler Salt Marsh, Connecticut (Osgood et al. 2003). However, data suggested that invertebrate richness was greater in *Spartina*-dominated than *Phragmites*-dominated sites during May and October (Osgood et al. 2003), which indicated that temporal variation at the beginning and end of the summer warm season could have influenced invertebrate abundance. Invertebrate diversity and richness results found in previous research of Alloway Creek in southern New Jersey suggested that richness and diversity were lower in invasive *Phragmites australis* sites than native *Spartina spp.* sites (Gratton and Denno 2005). Diversity data from the Hackensack Meadowlands and Long Island in 2001 also showed greater diversity at *Spartina alterniflora* marshes than *Phragmites australis* marshes, but these differences were not statistically significant across all sampling years (Robertson and Weis 2005).

In contrast to the negative or neutral responses observed in coastal marshes, research performed along the Lake Erie shoreline in Ohio indicated invertebrate diversity was positively correlated with *Phragmites australis* reed cover rather than with *Typha spp.* or other emergent macrophyte cover, and no effect of plant type on invertebrate abundance (Holomuzki and Klarer 2010). My study of Lake Erie Metropark (LEM) marshes found a generally neutral response of invertebrate abundance and diversity to *Phragmites australis*. Both sets of research found that there was no effect of plant type on invertebrate abundance, while a positive effect on diversity was found along the southern Lake Erie shoreline (Holomuzki and Klarer 2010) but no effect

was observed at LEM. Reasons for the contrasting effects of *Phragmites australis* on invertebrate diversity could be attributed to differences in sampling methods, time of the year in which sampling took place, and abiotic and biotic differences between the study systems. The research performed by Holomuzki and Klarer (2010) took advantage of laboratory microcosms to grow amphipods as well as using leaf litter bags submerged in approximately 0.4 m depth at the soil surface and 1–2 m distance from shoreline at Old Woman Creek National Estuarine Research Reserve (OWC) for collection. Hester-Dendy samplers used at LEM were placed at the same distance from the shoreline but submerged in at least one meter of water and above the sediment surface. Leaf litter bags offer the promise of food to attract amphipods, while Hester-Dendy samplers recruit invertebrates as a source of shelter. Not only were the sampling methods different, but the timing in which sampling was performed between the two studies is drastically different. Leaf litter bags were placed in field at OWC during October 2004 and collected monthly throughout the winter (Holomuzki and Klarer 2010), while Hester-Dendy samplers were only deployed at LEM from June 20 to July 22, 2013. The nine-year gap between studies and the extreme differences in seasonality that the Lake Erie region is subjected to could lead to widely different results in invertebrate diversity. In addition to the differences in sampling methods and timing, the two sampling areas have different hydrologies. OWC is a drowned river mouth being fed by inflows from Old Woman Creek and backflows from Lake Erie during storm events (Holomuzki and Klarer 2010). Sites at LEM are flooded marshlands with input from precipitation which flows downstream into Lake Erie and flooding from Lake Erie after storm and wind events. The two sites do not experience the same hydrology, and coupled with the differences in the seasonal timing of the samples could produce very different habitats in which to collect invertebrates. The disparity between the types of collection devices, timing of

sampling, and site specific differences could have led to different results in terms of calculated invertebrate diversity.

Taken together, my results, those of Holomuzki and Klarer (2010) for other Lake Erie marshes, and the coastal march literature could suggest that *Spartina alterniflora* (generally the non- *Phragmites* vegetation in coastal saltwater marsh studies) and *Typha spp.* from Lake Erie wetlands do not function equally in terms of maintaining greater richness and diversity. Invertebrate richness and diversity were either enhanced or unaffected by *Phragmites australis* relative to *Typha spp.* in Lake Erie marshes, but were unaffected or reduced by *Phragmites australis* relative to *Spartina alterniflora* in saltwater marshes.

Invertebrate Abundances

Overall mean invertebrate abundance was slightly greater in *Typha spp.* sites than in *Phragmites australis* sites; however, the difference in overall mean abundance was not statistically significant. The mean abundances of invertebrate categories were not significantly higher or lower in *Phragmites australis* sites compared to *Typha spp.* sites for all taxa, except for the mean abundance of *Helobdella modesta* that was significantly greater for *Typha spp.* sites than *Phragmites australis* sites. These results suggest that *Phragmites*-dominated sites and *Typha*-dominated sites were able to maintain similar abundances of most invertebrate taxa. No differences in mean abundance were observed when taxa were organized by functional feeding groups ($p>0.05$), except for the predator category which showed a statistically different abundance of organisms based on plant type with higher abundances observed in *Typha*-dominated sites ($p<0.05$).

Previous research of invertebrate abundance and its relationship to dominant marsh plant type have provided conflicting results. Some studies of invertebrates have failed to observe any significant differences in invertebrate abundances or densities between sites dominated by *Phragmites australis* and sites dominated by other marsh plants such as *Spartina alterniflora* (Abel and Hagan 2000; Fell et al. 2003; Fell et al. 2006; Gratton and Denno 2005; Holomuzki and Klarer 2010; Kulesza and Holomuzki 2006; Osgood et al. 2003; Posey et al. 2003; Warren et al. 2001). During 2001 at the Hackensack Meadowlands research sites, overall densities were not significantly different for *Spartina alterniflora* and *Phragmites australis* areas (Robertson and Weis 2005). In the Charles Wheeler Salt Marsh, total invertebrate density was not significantly different between *Phragmites australis* and *Spartina alterniflora* marshes (Osgood et al. 2003). In contrast, other work has reported that mean invertebrate abundances were reduced in *Phragmites australis*-dominated areas compared to sites dominated by the native marsh plant of those locations (Robertson and Weis 2005). In 2000 and 2002 at the Hackensack Meadowlands research sites, epi-faunal densities were greatest in *Spartina alterniflora* locations than *Phragmites australis* locations for all taxa identified (Robertson and Weis 2005).

Some previously published studies (Fell et al. 2003; Fell et al. 2006; Posey et al. 2003; Robertson and Weis 2005) included many of the same invertebrate groups (e.g. amphipods, gastropods, beetles, mites, oligochaetes, isopods, and insects) that were collected during this study. These group-specific findings were compared to observe patterns. Amphipods collected from Long Island in 2000 showed statistically equal abundances in *Phragmites australis* sites as *Spartina alterniflora* sites (Robertson and Weis 2005), which is similar to the research performed at LEM. Amphipods, specifically *Gammarus mucronatus*, were more abundant in *Spartina alterniflora* than *Phragmites australis* sites when collected using 12cm sediment cores from the

Chester River and Prospect Bay regions of Chesapeake Bay (Posey et al. 2003). Amphipods in the Connecticut River did not have significantly different mean abundances when captured in Breder traps; however, the same amphipod species captured using pit traps had higher mean abundances in *Phragmites australis* locations than *Typha spp.* locations in 2000, but mean abundances were not significantly different during 2001 (Fell et al. 2003). However, in 2006, amphipods were significantly greater in treated *Phragmites australis* locations than either untreated *P. australis* or *Typha angustifolia* locations (Fell et al. 2006). Dipterans sampled from the Connecticut River were also evenly distributed across plant type (Fell et al. 2003). The similarities of amphipod abundance and Dipteran distribution between plant types parallel what was observed in samples from LEM. Gastropod mean abundance and the mean abundances of individual gastropod taxa were not significantly different according to plant type in LEM. These results are similar to the abundance of Hydrobiid snails and *Stagnicola catascopium* in the Lieutenant River (Fell et al. 2006), even though neither of these types of snails were identified at LEM. Alternatively, in Chesapeake Bay, the snail *Hydrobia minuta* was more common in *Spartina alterniflora* marshes than *Phragmites australis* marshes (Posey et al. 2003). *H. minuta* was captured using different methods in Chesapeake Bay than those used at LEM and was not identified in LEM either. Mean abundances of diving water beetles, mites, oligochaetes, and isopods were greater in *Typha angustifolia* sites than treated and un-treated *Phragmites australis* sites in the Lieutenant River (Fell et al. 2006), which disagrees with the results from the described research conducted in the freshwater marshes of LEM. However, in 2000 at Long Island and in 2001 at both Long Island and Hackensack Meadowlands sites, mite densities were considered statistically equal in *Spartina alterniflora* and *Phragmites australis* marshes (Robertson and Weis 2005). Insects had statistically greater abundances in *Spartina alterniflora*

than *Phragmites australis* areas during 2001 and 2002 at Long Island and the Hackensack Meadowlands (Robertson and Weis 2005), which disagrees with the abundance data collected from LEM. In general, it is apparent that the relationship between invertebrate abundances and dominant macrophyte type is not consistent across invertebrate groups or across studies.

Invertebrate Community Structure

Invertebrate community structure analyzed using PCA and MANOVA methods also revealed no statistically significant difference in overall community structure between *Phragmites australis*-dominated sites and *Typha*-dominated sites ($p>0.05$). These results imply that the two different plant types function similarly in maintaining surface-associated aquatic invertebrate communities at a small scale. A similar study of the community structure of macroinvertebrates and fishes in the Lieutenant River revealed the communities were not statistically different for treated *Phragmites australis* marshes, untreated *Phragmites australis* marshes, and *Typha angustifolia* marshes (Fell et al. 2006). During that study it was observed that herbicide and mowing of *P. australis* had no significant impact on macroinvertebrate community structure and use of the marsh during the months following *P. australis* treatment (Fell et al. 2006). This could suggest that the influence of the plant could likely be the result of sub-aquatic interactions rather than the effect of aboveground plant type or biomass. Along the Lake Erie coast in Ohio, community composition and dominant functional feeding group varied significantly among wetland sites (Holomuzki and Klarer 2010); however, in Lake Erie Metropark, *Phragmites australis* sites and *Typha spp.* sites were both significantly dominated by gatherer/collectors with much lower abundances of other functional feeding groups. The predator

functional feeding group was the only category to show significant differences according to plant type ($p<0.05$).

Leaf litter is an important component in the diet of many marsh invertebrates and other organisms. It could be hypothesized that differences that occur between *Phragmites*-dominated sites and *Typha*-dominated sites could be a result of the change in water chemistry or marsh topography, but differences could also be due to differences in the quality of leaf litter produced by the two plant types. Because there were no observable differences in invertebrate abundance or community structure of sites within Lake Erie Metropark, it could be inferred that the quality of leaf litter as a food source was not significantly different between *Phragmites australis* and *Typha spp.* plants. Research of amphipods collected from Lake Erie in Huron, Ohio supports this conclusion (Kulesza and Holomuzki 2006). Amphipod abundance was positively correlated to fungal growth and biomass on leaf litter in which mean fungal biomass was the greatest on *Phragmites australis* treated with herbicide and natural *Typha angustifolia* but lowest on un-treated *Phragmites australis* (Kulesza and Holomuzki 2006). Despite the differences in amphipod abundance and fungal biomass, there was no significant difference in leaf litter breakdown by these organisms or rates of growth of these organisms between *Phragmites australis* and *Typha angustifolia* (Kulesza and Holomuzki 2006), which suggests that these plant types are both sufficient food sources for the identified fungi and amphipods.

Three groups of co-occurring taxa were extrapolated from PCA. The first group of co-occurring taxa included Hemiptera Pleidae (HP), Odonata Zygoptera/Anisoptera (OZ), unknown Gastropod (G), members of the families Viviparidae and Bithyniidae (VB), and *Placobdella rugosa* (PR). The second group included Hemiptera Gerridae (HG), *Erpobdella spp.* (ES), *Helobdella modesta* (c.f. *stagnalis*) (HM), Annelida Oligochaetes (AO), Diptera Chironomidae

(DCH), and Ephemeroptera (E). The third grouping included Decapoda *Orconectes spp.* (DO), Sphaeriidae (S), Flatworms (F), Diptera Ceratopogonidae (DCE), Limpets (L), and Amphipods (A). These groupings could be the result of similar habitat preferences among organisms within the groups. Even though no significant differences in invertebrate richness, diversity, or abundances were observed based on plant type, there may have been subtle differences between each site which could have driven the different taxa into these observable groupings. Aside from habitat preferences, it could be hypothesized that habitat partitioning may explain how the taxa grouped together. For example, the organisms in the first grouping may have adequately filled the niches in those sites and excluded taxa from the other two groupings. Much more field research and an extensive knowledge of each taxon would be required to further delve into why each taxon fell into which group and what specifically influenced those observations if they were not the result of random chance or sampling bias.

Influence of Environmental Variables

Environmental variables of the water column which included mean temperature (°C), pH, dissolved oxygen concentration (mg/L), and percent dissolved oxygen saturation were analyzed to determine if any differences occurred across site type which could have influenced the invertebrate community. Statistical analysis of water temperature indicated that throughout the season there was a significant change in the mean water temperature over the course of sampling, but that the patterns of change were not significantly different, and there was not a significant difference in mean water temperature across site type. There was also no correlation between mean temperature and invertebrate abundance, suggesting that temperature was not influencing invertebrate abundance in this study. In the Mullica River, Lieutenant River, and Chesapeake

Bay, mean water temperatures were also similar amongst *Phragmites australis* areas and the native *Spartina spp.* areas (Able and Hagan 2003; Able et al. 2003; Meyer et al. 2001). However, a later study of the Lieutenant River showed that mean water temperature was greater in *Phragmites australis* sites that had been treated versus non-treated *Phragmites australis* and the native *Typha angustifolia* (Fell et al. 2006). These differences in mean water temperature could be a result of the treatment process of herbicides and mowing in the treated *Phragmites australis* sites, allowing greater sunlight to reach the water's surface. At Lake Erie Metropark, there was no obvious difference in mean temperature between herbicide treated *Phragmites* sites (P3 and P4) and untreated sites (P1 and P2), but no statistical analysis was performed due to low replication. The lack of mowing in Lake Erie Metropark treated *Phragmites* sites might have reduced any local temperature difference. Mowing removes shade which could result in increased local water temperatures. Results of another study which included the Mullica River as well as Blackbird Creek and Monie Creek suggested that water temperature of natural marshes did not differ from marshes considered initial, early, or late invasion stages of *Phragmites australis* (Hunter et al. 2006), which more closely resembles the type of trends observed at Lake Erie Metropark when observing sites as herbicide treated *Phragmites* sites, untreated *Phragmites* sites, and unaffected *Typha spp.* sites. Water motion at these sites may have been sufficient enough to mix water across sites, thereby reducing or preventing local thermal differences, but the water motion at LEM may not be strong enough to mix across sites.

Overall, the pH of *Phragmites australis* sites and *Typha spp.* sites significantly changed over the course of sampling and the temporal pattern of pH change was statistically different between the two plant types; however, there was no statistically significant difference in the mean pH between *Phragmites*-dominated sites and *Typha*-dominated sites. A difference in mean

pH was not observed when taking into consideration herbicide treatment of *Phragmites* sites either. There appeared to be a significant positive correlation between pH and gastropod abundance, suggesting gastropods were more abundant at sites which had higher mean pH readings. The gastropods sampled could favor higher mean pH water environments due to specific life history traits of that particular taxon, but there are no published scientific literature articles to support this postulation for the species in question. One study performed in Rio de Janeiro suggests that gastropods, specifically *Physa marmorata*, had a positive correlation between mean abundance and mean alkalinity (Giovanelli et al. 2005). This research would support the significant positive correlation between pH and gastropod abundance observed from the data collected at Lake Erie Metropark.

Mean dissolved oxygen concentration (mg/L) showed statistically significant changes in concentration during the sample period, although the temporal pattern of change in dissolved oxygen concentration (mg/L) was not statistically significant based on plant type meaning this environmental variable changed in *Phragmites* sites with a similar pattern as *Typha* sites. The patterns of changes were similar, but the mean dissolved oxygen concentration (mg/L) was significantly different in *Phragmites*-dominated sites and *Typha*-dominated sites with greater mean for *Typha*-dominated sites. Within *Phragmites* sites, herbicide treated sites appeared to have greater variability in mean dissolved oxygen concentration (mg/L) than untreated sites, but the data does not suggest any considerable difference in means. Despite the significant differences in mean dissolved oxygen concentration (mg/L), there were no statistically significant correlations observed between dissolved oxygen concentration (mg/L) and any of the taxa mean abundances, suggesting that this variable was not an influencing factor in the abundance of any of the identified taxa in this study.

Percent dissolved oxygen saturation changed throughout the season with statistically different patterns of change observed for *Phragmites*-dominated sites and *Typha*-dominated sites with a significantly greater mean percent dissolved oxygen saturation at *Typha*-dominated sites than *Phragmites*-dominated sites. Within *Phragmites* sites, herbicide treated sites appeared to have greater variability in mean dissolved oxygen concentration (mg/L) than untreated sites, but no considerable difference in means was observed. Regardless of this observation, there was no significant correlation between mean percent dissolved oxygen saturation and the mean abundance of any of the taxa groups, indicating that this variable was not an influencing factor in invertebrate abundance.

Potential Influence of Temporal Variation and Methodology

Invertebrate response to *Phragmites* may vary temporally. In 1997, plant type showed no effect on invertebrate abundance while in 1998 the opposite effect was seen in the Mullica River (Able and Hagan 2000). Amphipod abundances in the Connecticut River sampled using pit traps had higher mean abundances in *Phragmites*-dominated sites than *Typha*-dominated sites in 2000, but mean abundances were not significantly different during 2001 (Fell et al. 2003). This interannual variability in the effects of plant species on invertebrate abundance in the Mullica River and Connecticut River suggests that differing patterns may have been observed for Lake Erie Metropark (LEM) had subsequent years of sampling been performed. Thus, additional, multi-year sampling may be required to fully examine the influence of *Phragmites* on wetland invertebrates. Differences may also exist on a seasonal time-scale. For example, invertebrate densities were greater in *Spartina alterniflora* marshes than *Phragmites australis* marshes during May and October at Charles Wheeler Salt Marsh even though temporally-integrated invertebrate

density was not statistically different (Osgood et al. 2003). Similarly, greater invertebrate densities were associated with *Phragmites australis* areas early in the sampling season but with *Spartina alterniflora* areas later in the sampling season at the Hackensack Meadowlands and Long Island sites in 2001 and 2002 (Robertson and Weis 2005). Thus, temporal variation at the beginning and end of the summer season could be influencing the patterns of invertebrate abundance but not significantly affecting overall abundances of invertebrates. Seasonal differences in regard to seasonality could be observed at LEM through the use of multi-year sampling with a focus on temporal variation. If the samplers had been emptied multiple times and redeployed during the field season, a more broad view of the invertebrate community could have been obtained.

Observed richness, diversity, abundance, and community structure of invertebrates can be dependent on specific sampling techniques or environmental factors. Macroinvertebrates such as *Calinectes sapidus*, *Paleomonetes pugio*, and *Rithropanopeus harissii* did not have significant differences in abundance between *Phragmites australis*-dominated sites and *Spartina*–dominated sites when collected in flume traps, but when collected in pit traps, *C. sapidus* and *P. pugio* were more abundant in *Spartina spp.* marshes than *Phragmites australis* marshes, while *R. harissii* was more abundant in *Phragmites australis*-dominated marshes (Able and Hagan 2000). Amphipod abundances of the Connecticut River were not significantly different when captured using Breder traps; however, the same amphipod species captured using pit traps had higher mean abundances in *Phragmites australis* sites than *Typha* sites in 2000 with no differences in abundance observed during 2001 (Fell et al. 2003).These observations suggests that different trapping methods can produce different results, which could indicate that had multiple sampling techniques been used rather than only Hester-Dendy samplers, different trends in invertebrate

abundance could have been seen in the freshwater marshes at LEM. Members of the Pleidae, Gerridae, Limpets, and the unknown gastropod group were only recorded at *Typha* sites, while *Orconectes spp.* was only collected in *Phragmites australis* sites within LEM. Because Hester-Dendy samplers are artificial substrate samplers, they are intended to collect surface-associated organisms rather than zooplankton or invertebrates free floating in the water column. Unlike flume traps and pit traps, Hester-Dendy samplers are recruitment devices, meaning an invertebrate is likely collected because it is inhabiting the sampler rather than passing through. These samplers give a view of resident invertebrates and not those which lifestyle consists of living and moving through the marsh passively. Also, the samplers have a fixed surface area and a fixed spacing between layers which only allows a certain size organism and a maximum number of organisms to persist at one time. Any of the mentioned trapping methods could provide useful information regarding invertebrate diversity and richness, but Hester-Dendy samplers were chosen because they were readily available, have the ability to collect resident invertebrates, and the ease at which they could be deployed and collected was convenient.

Ecosystem Implications

The Detroit River watershed is home to hundreds of species that are ecologically and economically important to the western Lake Erie region. Three hundred bird species have been documented in the Detroit River watershed, which includes LEM (U.S. Fish and Wildlife Service 2008). Approximately 3 million waterfowl, including ducks, geese, swans, and coots are estimated to migrate through the Detroit River watershed yearly (International Association for Great Lakes Research 2012). In 1991, $20.1 million in retail sales related to waterfowl hunting was brought into the state of Michigan and another $192.8 million from bird watching,

photography, and other non-consumptive uses of waterfowl (International Association for Great Lakes Research 2012). In addition to birds, 117 species of fish are known to use components of the Detroit River watershed (U.S. Fish and Wildlife Service 2008). Walleye fishing contributes $1 million dollars to communities along the Detroit River, and ten million walleye move up the Detroit River during the spring to spawn (International Association for Great Lakes Research 2012).

Birds and fish living in the Detroit River watershed are highly dependent on the wetlands for survival. Wetlands, especially coastal marshlands, provide crucial breeding grounds for birds and fishes because the areas between vegetation act as a safe place to deposit eggs and raise young (Able and Hagan 2000). These animals may also use the coastal marshes as pit-stops during migration or as a home year-round. Birds and fish in a marsh habitat rely on an abundant and diverse aquatic invertebrate community. Invertebrates are a food source for many of the larval, young-of-year, and adult fishes and waterfowl that reside within the wetland. These fishes and waterfowl are in turn a food source for people living within the watershed. As previously stated, waterfowl and fishes are not only economically important, but are also biologically important to the people living in the Detroit River watershed. The food web of this habitat becomes unbalanced when any of its components are affected, even its smaller organisms including the invertebrates.

Although the research performed at Lake Erie Metropark showed no significant effect of *Phragmites australis* on invertebrate abundance, diversity, or community structure, this research was just a snapshot of the ecosystem. If *Phragmites australis* affects the diversity and abundance of aquatic invertebrates on a longer time-scale, we might expect to see a trophic cascade which could potentially reduce the diversity and abundance of the 300 bird species and 117 species of

fish recorded in the Detroit River watershed. A reduction in the number of birds present could reduce the amount of revenue injected into the Michigan economy from birding activities. Because so many fishes use the marsh during one or more life-stages, a reduction in game species resulting from wetland vegetation alterations such as the *Phragmites australis* invasion could have long-term effects upon the fishing industry as well.

Phragmites australis removal has been implemented in the region during the past few years. The outcomes of my research could provide useful information about the relationship between *Phragmites australis* and epi-faunal communities to aid management decisions regarding *Phragmites australis* removal and the potential benefit or harm to these aquatic communities. There is not much information regarding how *Phragmites australis* affects bird populations or the economic value of birding activities and walleye fishing to the local community as it relates to *Phragmites australis*, but the valuable insights into the relationship between *Phragmites australis* and invertebrates gained by this research could provide information concerning these fields as well as direct management of *Phragmites australis.* More research is needed to understand the ecology of *Phragmites australis* before any further or dramatic means of removal of the plant material should be implemented. A multiyear study with a consistent sampling timeline and multiple collection events over a larger sampling area containing site replicates would provide a more detailed comprehension into how invertebrates use the marsh. Further analysis of the potential spread of *Phragmites australis* alongside the effects of controlling the reed in a freshwater environment to determine the projected affects on wildlife abundance and diversity will give greater insight to how these wetlands function.

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References

Able KW, Hagan SM (2000) Effects of common reed (*Phragmites australis*) invasion on marsh surface macrofauna: Response of fishes and decapod crustaceans. Estuaries 23: 633-646.

Able KW, Hagan SM (2003) Impact of common reed, *Phragmites australis*, on essential fish habitat: Influence on reproduction, embryological development, and larval abundance of mummichog (*Fundulus heteroclitus*). Estuaries 26: 40-50.

Able KW, Hagan SM, Brown SA (2003) Mechanisms of marsh habitat alteration due to *Phragmites*: Response of young-of- the-year mummichog (*Fundulus heteroclitus*) to treatment for *Phragmites* removal. Estuaries 26: 484-494.

Barbour MT, Gerritsen J, Snyder BD, Stribling JB (1999) Rapid Bioassessment Protocols for Use in Streams and Wadeable Rivers: Periphyton, Benthic Macroinvertebrates and Fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency; Office of Water, Washington DC.

Fell PE, Warren RS, Light JK, Rawson RL Jr., Fairley SM (2003) Comparison of fish and macroinvertebrate use of *Typha angustifolia*, *Phragmites australis*, and treated *Phragmites* marshes along the Lower Connecticut River. Estuaries 26: 534-551.

Fell PE, Warren RS, Curtis AE, Steiner EM (2006) Short-term effects on macroinvertebrates and fishes of herbiciding and mowing *Phragmites australis*-dominated tidal marsh. Northeastern Naturalist 13: 191-212.

Galatowitsch SM, Anderson NO, Ascher PD (1999) Invasiveness in wetland plants in temperate North America. Wetlands 19: 733-755.

Giovanelli A, da Silva CLPAC, Leal GBE, Baptista DF (2005) Habitat preference of freshwater snails in relation to environmental factors and the presence of the competitor snail

Melanoides tuberculatus (Müller, 1774). Memorias do Instituto Oswaldo Cruz 100: 169-176.

Gratton C, Denno RF (2005) Restoration of arthropod assemblages in a *Spartina* salt marsh following removal of the invasive plant *Phragmites australis*. Restoration Ecology 13: 358-372.

Grothues TM, Able KW (2003) Response of juvenile fish assemblages in tidal salt marsh creeks treated for *Phragmites* removal. Estuaries 26: 563-573.

Hagan SM, Brown, SA, Able KW (2007) Production of mummichog (*Fundulus heteroclitus*): Response in marshes treated for common reed (*Phragmites australis*) removal. Wetlands 27: 54-67.

Hanson SR, Osgood DT, Yozzo DJ (2002) Nekton use of a *Phragmites australis* marsh on the Hudson River, New York, USA. Wetlands 22: 326-337.

Hauber DP, Saltonstall K, White DA, Hood CS (2011) Genetic variation in the common reed, *Phragmites australis*, in the Mississippi River Delta marshes: Evidence for multiple introductions. Estuaries and Coasts 34: 851-862.

Holomuzki JR, Klarer DM (2010) Invasive reed effects on benthic community structure in Lake Erie coastal marshes. Wetlands Ecology and Management 18: 219-231.

Hunter KL, Fox DA, Brown LM, Able KW (2006) Responses of resident marsh fishes to stages of *Phragmites australis* invasion in three mid Atlantic estuaries. Estuaries and Coasts 29: 487-498.

International Association for Great Lakes Research (2012) Great Lakes Science & Policy: Conserving Detroit River Habitats.

Kimball ME, Able KW (2007) Nekton utilization of intertidal salt marsh creeks: Tidal influences in natural *Spartina*, invasive *Phragmites*, and marshes treated for *Phragmites* removal. Journal of Experimental Marine Biology and Ecology 346: 87-101.

Kulesza AE, Holomuzki JR (2006) Amphipod performance responses to decaying leaf litter of *Phragmites australis* and *Typha angustifolia* from a Lake Erie coastal marsh. Wetlands 26: 1079-1088.

Lathrop RG, Windham L, Montesano P (2003) Does *Phragmites* expansion alter the structure and function of marsh landscapes? Patterns and processes revisited. Estuaries 26: 423-235.

Merritt RW, Cummins KW, Berg MB (2008) An introduction to the aquatic insects of North America 4th edition. Kendall/ Hunt Publishing Company, Dubuque.

Meyer DL, Johnson JM, Gill JW (2001) Comparison of nekton use of *Phragmites australis* and *Spartina alterniflora* marshes in the Chesapeake Bay, USA. Marine Ecology- Progress Series 209: 71-84.

Meyerson LA, Saltonstall K, Windham L, Kiviat E, Findlay S (2000) A comparison of *Phragmites australis* in freshwater and brackish marsh environments in North America. Wetlands Ecology and Management 8: 89-103.

Meyerson LA, Lambert AM, Saltonstall K (2010a) A tale of three lineages: Expansion of common reed (*Phragmites australis*) in the U.S. Southwest and Gulf Coast. Weed Science Society of America 3: 515-520.

Meyerson LA, Viola DV, Brown RN (2010b) Hybridization of invasive *Phragmites australis* with a native subspecies in North America. Biological Invasions 12: 301-111.

Osgood DT, Yozzo DJ, Chambers RM, Jacobson D, Hoffman T, Wnek J (2003) Tidal hydrology and habitat utilization by resident nekton in *Phragmites* and non-*Phragmites* marshes. Estuaries 26: 522-533.

Paul J, Vachon N, Garroway CJ, Freeland JR (2010) Molecular data provide strong evidence of natural hybridization between native and introduced lineages of *Phragmites australis* in North America. Biological Invasions 12: 2967-2973.

Pennak RW (1953) Fresh-water invertebrates of the United States. The Ronald Press Company, New York.

Plut K, Paul J, Ciotir C, Major M, Freeland JR (2011) Origin of non-native *Phragmites australis* in North America, a common wetland invader. Fundamentals of Applied Limnology 179: 121-129.

Posey MH, Alphin TD, Meyer DL, Johnson JM (2003) Benthic communities of common reed *Phragmites australis* and marsh cordgrass *Spartina alterniflora* marshes in Chesapeake Bay. Marine Ecology Progress Series 261: 51-61.

Raichel DL, Able KW, Hartman JM (2003) The influence of *Phragmites* (common reed) on the distribution, abundance, and potential prey of a resident marsh fish in the Hackensack Meadowlands, New Jersey. Estuaries 26: 511-521.

Robertson TL, Weis JS (2005) A comparison of epifaunal communities associated with the stems of salt marsh grasses *Phragmites australis* and *Spartina alterniflora*. Wetlands 25: 1-7.

Saltonstall K (2002) Cryptic invasion by a non-native genotype of the common reed, *Phragmites australis*, into North America. Proceedings of the National Academy of Sciences of the United States of America 99: 2445-2449.

Saltonstall K (2011) Remnant native *Phragmites australis* maintains genetic diversity despite multiple threats. Conservation Genetics 12: 1027-1033.

Trebitz AS, Taylor DL (2007) Exotic and invasive aquatic plants in Great Lakes coastal wetlands: Distribution and relation to watershed land use and plant richness and cover. Journal of Great Lakes Research 33: 705-721.

U.S Fish and Wildlife Service (2008) Detroit River International Wildlife Refuge.

Warren RS, Fell PE, Grimsby JL, Buck EL, Rilling GC, Fertik RA (2001) Rates, patterns, and impacts of *Phragmites australis* expansion and effects of experimental *Phragmites* control on vegetation, macroinvertebrates, and fish within tidelands of the lower Connecticut River. Estuaries 24: 90-107.

Weis JS, Weis P (2003) Is the invasion of the common reed, *Phragmites australis*, into tidal marshes of the eastern US an ecological disaster? Marine Pollution Bulletin 46: 816-820.

Windham L, Lathrop RG Jr. (1999) Effects of *Phragmites australis* (common reed) invasion on aboveground biomass and soil properties in brackish tidal marsh of the Mullica River, New Jersey. Estuaries 22: 927-935.

Figures

Figure 1: Map of field locations within Lake Erie Metropark.

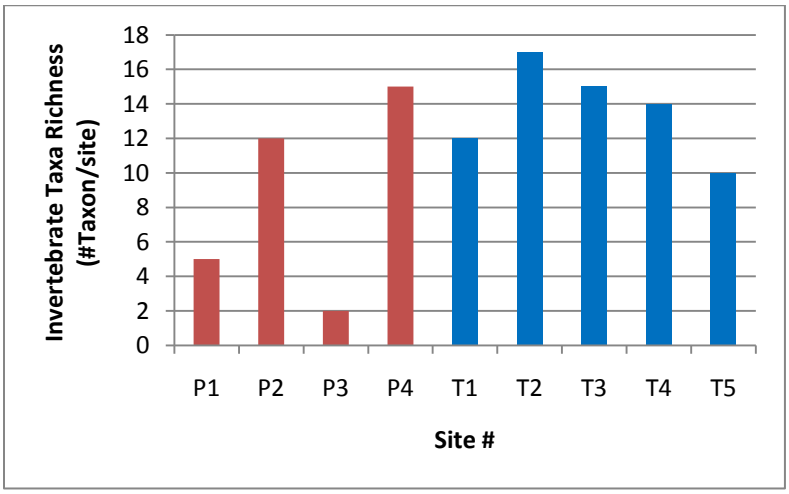

Figure 2: Invertebrate taxonomic richness of each site recorded as the number of invertebrate categories present.

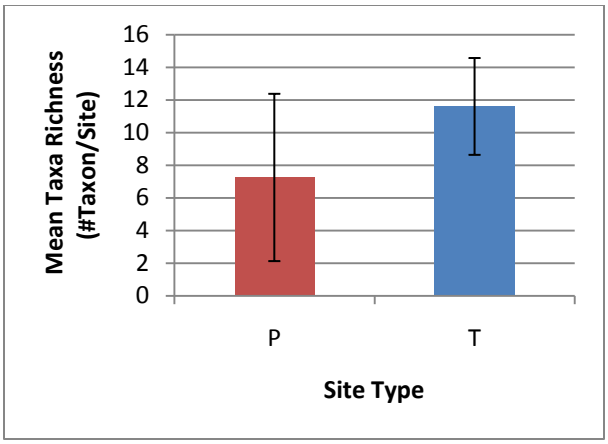

Figure 3: Mean (±1SD) invertebrate taxonomic richness p > 0.05.

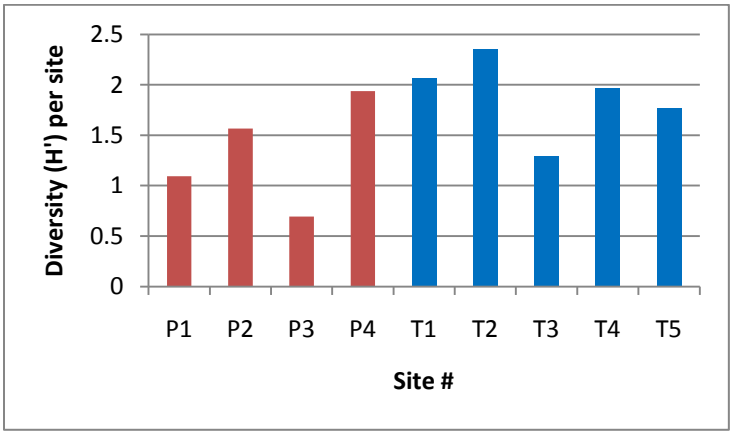

Figure 4: Invertebrate diversity of each individual site sampled.

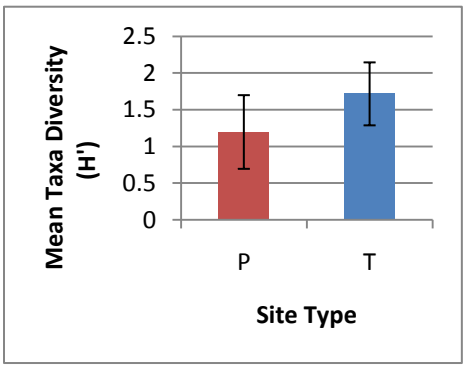

Figure 5: Mean (±1SD) invertebrate diversity H' p > 0.05.

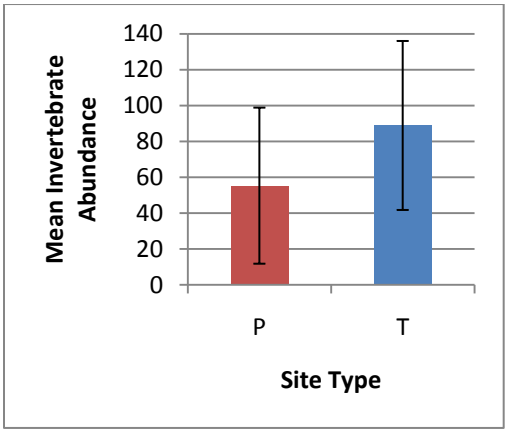

Figure 6: Mean (±1SD) invertebrate abundance p > 0.05.

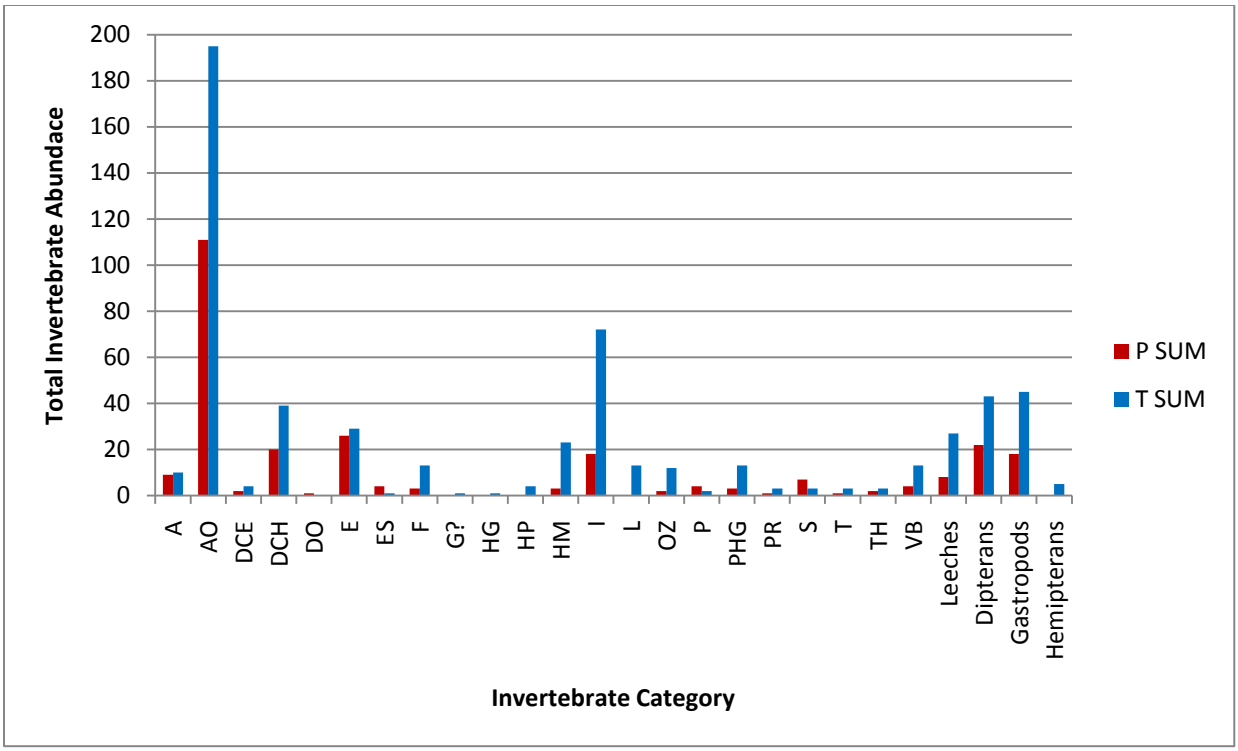

Figure 7: Sum abundance of each taxon for *Phragmites*-dominated sites and *Typha*-dominated sites.

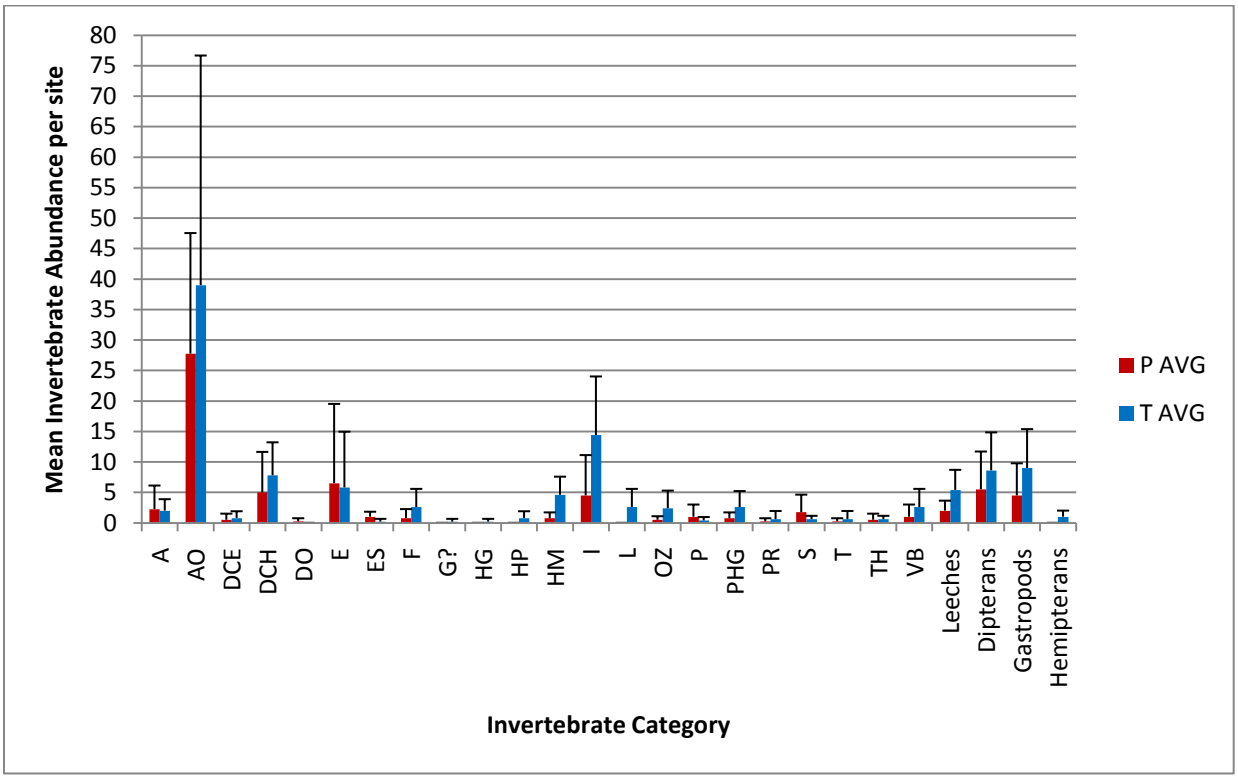

Figure 8: Mean (±1SD) invertebrate abundance of each taxon.

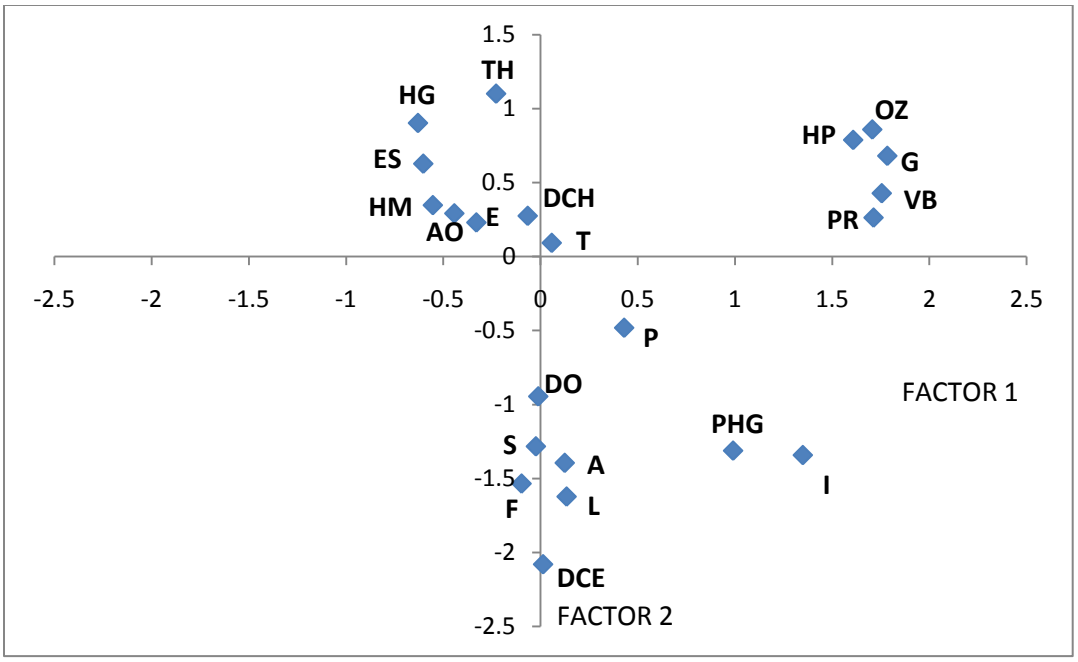

Figure 9: Invertebrate taxonomic categories (abbreviations in Results) plotted as locations of the factor coefficients of the first two factors of PCA.

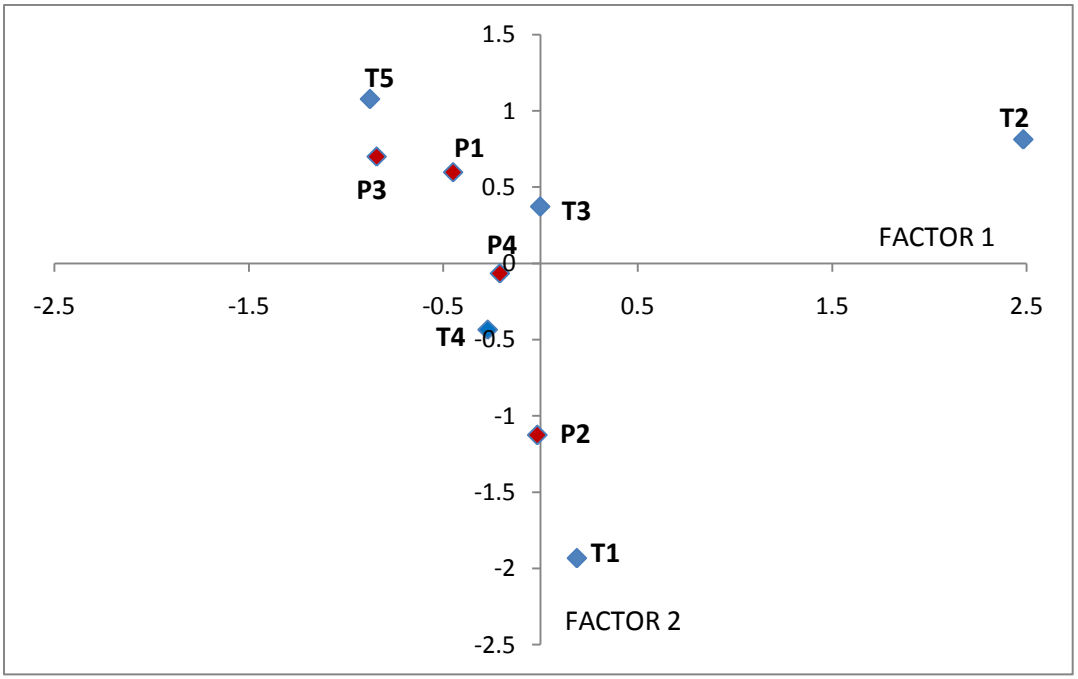

Figure 10: Sites plotted as a position of the factor scores obtained from the first two factors of PCA using invertebrate taxa.

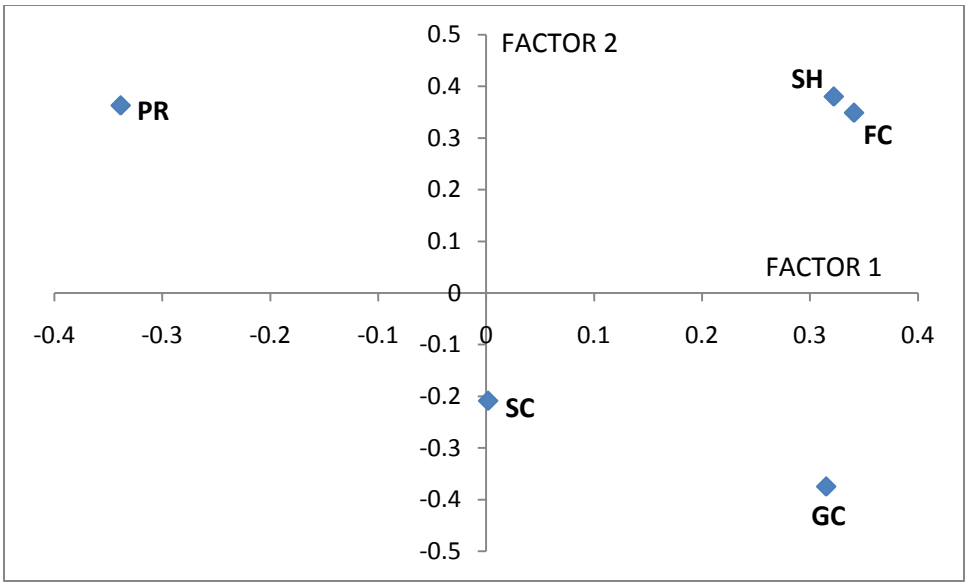

Figure 11: FFG (abbreviations in Results) plotted as locations of the factor coefficients of the first two factors of PCA.

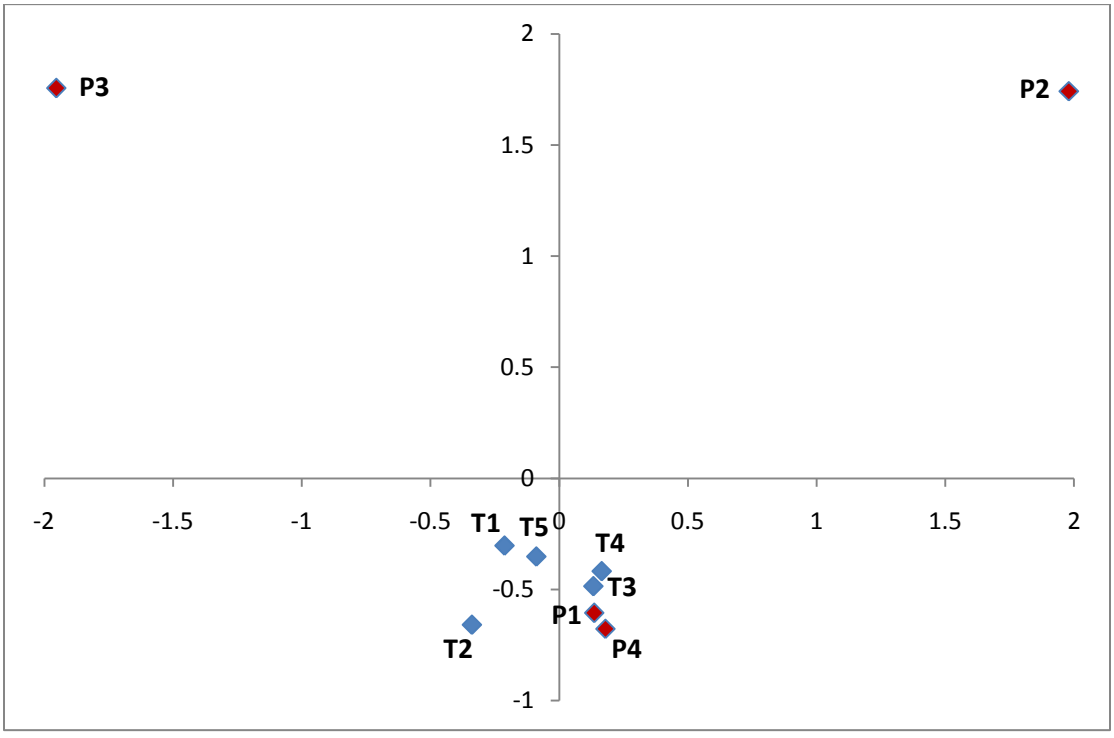

Figure 12: Sites plotted as a position of the factor scores obtained from the first two factors of PCA using FFG.

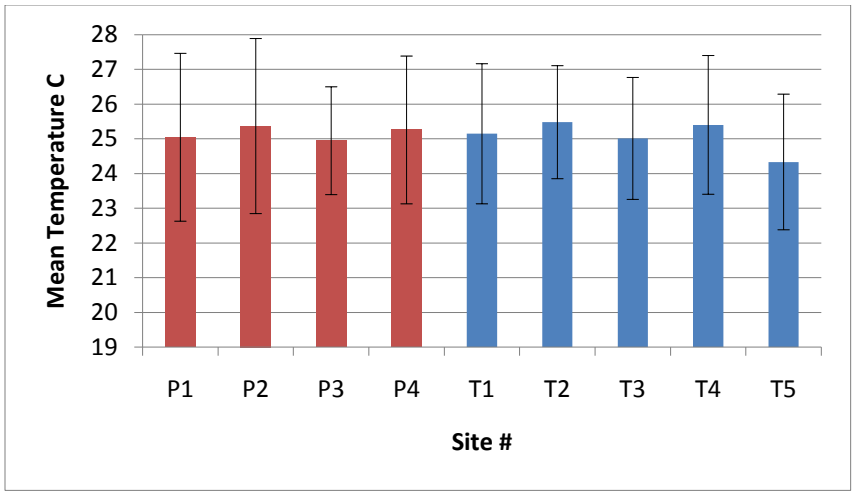

Figure 13: Mean (±1SD) temperature C of each sample site.

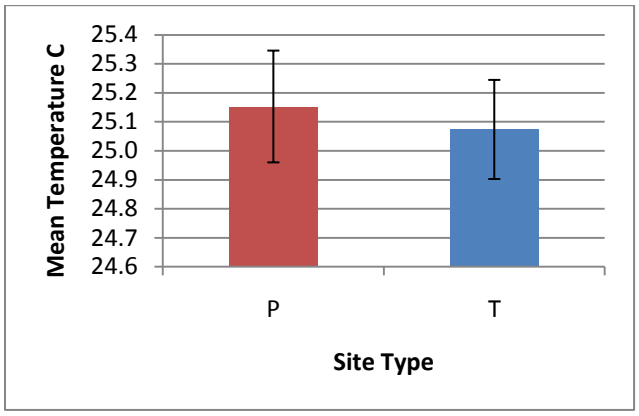

Figure 14: Mean (±1SD) temperature C of the means from Figure 11 p>0.05.

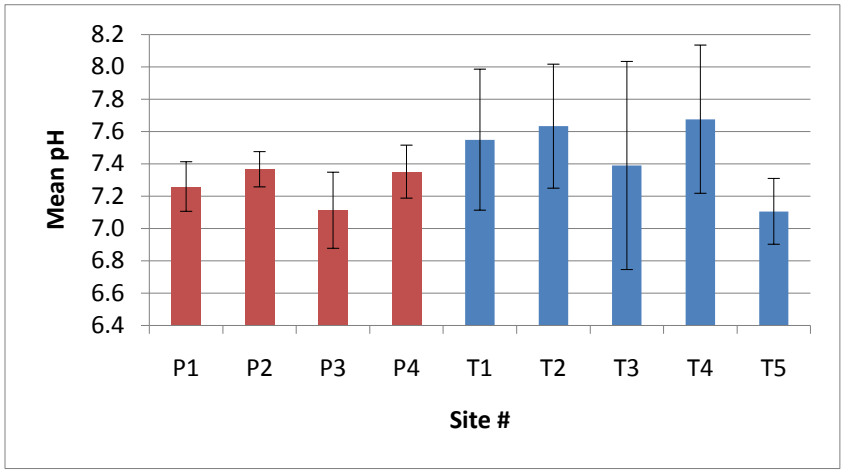

Figure 15: Mean (±1SD) pH of each sample site.

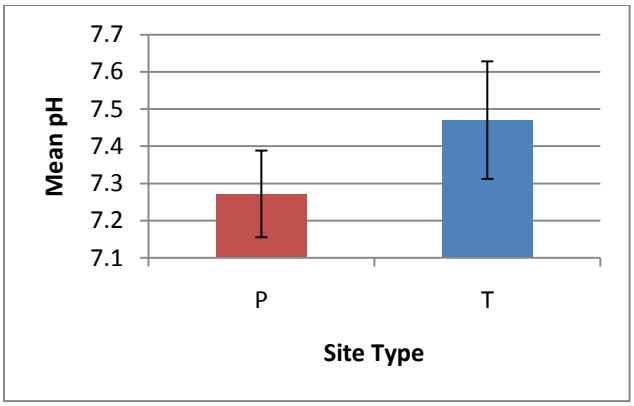

Figure 16: Mean (±1SD) of the mean pH readings from Figure 14 p >0.05.

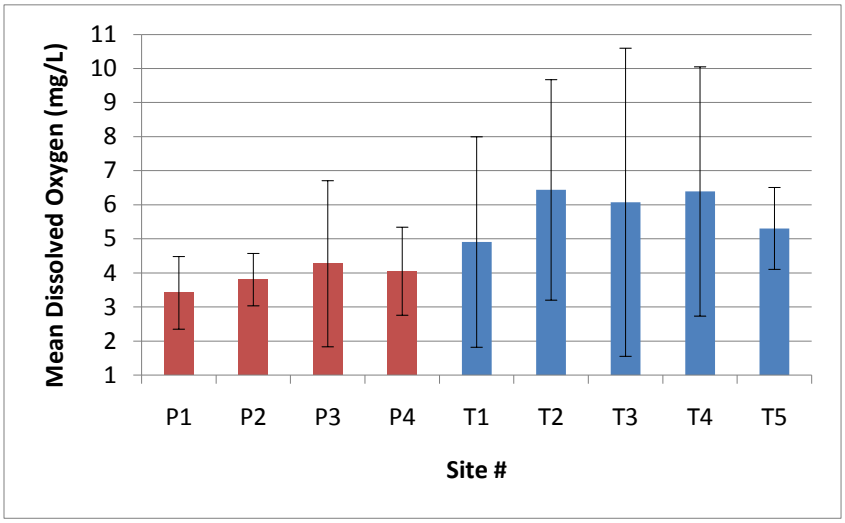

Figure 17: Mean (±1SD) dissolved oxygen concentration mg/L for each site.

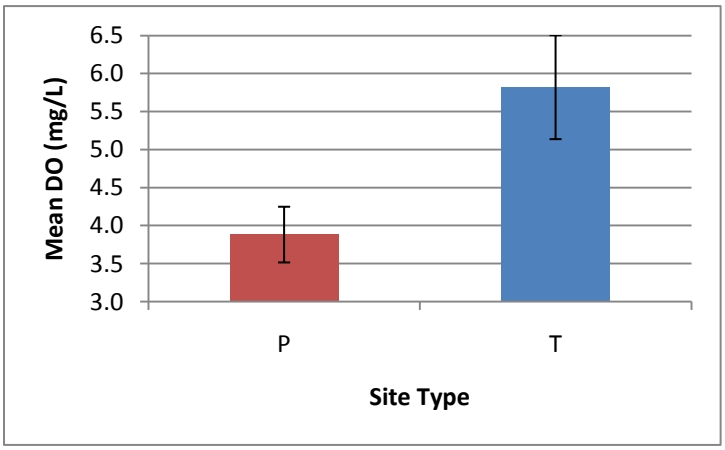

Figure 18: Mean (±1SD) dissolved oxygen concentration mg/L of the means from Figure 17 p<0.05.

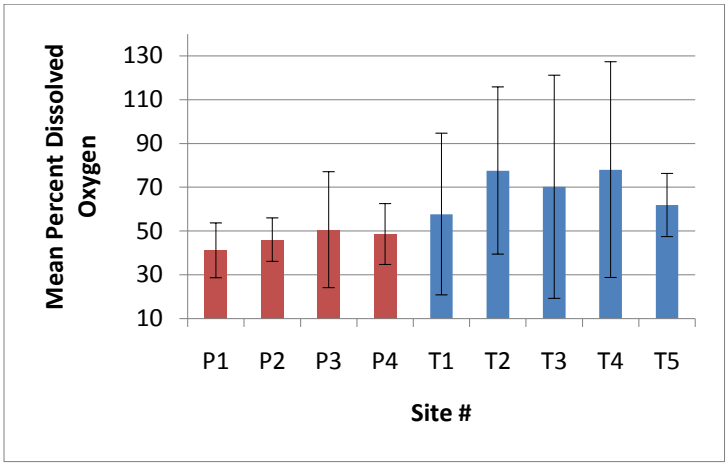

Figure 19: Mean (±1SD) percent dissolved oxygen saturation for each sample site.

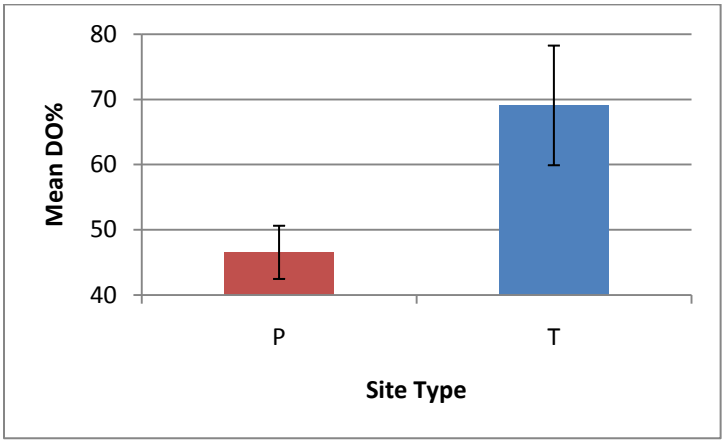

Figure 20: Mean (±1SD) percent dissolved oxygen saturation of the means from Figure 19 $p<0.05$.