Quantifying Waterfowl Use and Habitat Characteristics Following Wetland Restoration in Lake Ontario Coastal Wetlands at Braddock Bay Wildlife Management Area

By

Christopher J. Mitchell

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APPROVED BY:

5/10/24	
Major Advisor	Date
5/10/24	
Committee Member	Date
5/10/24	
Committee Member	Date
5/14/24	
Committee Member	Date
5/14/2024	
Graduate Director	Date
Jacques Rinchard	5/15/2024
Chair, Environmental Science and Ecology	Date

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ABSTRACT

The coastal wetlands of Braddock Bay Wildlife Management Area (WMA) on the southern coast of Lake Ontario provide migratory spring-stopover habitat that is important for a variety of pre-breeding season waterfowl. Cattail (*Typha sp.*) invasion can degrade these spring-stopover wetlands, and directly decrease the amount of open-water interspersion within a wetland. One of the targets of restoration is to increase the amount of open-water area interspersed within marshes to improve waterfowl habitat. This study aimed to assess the effects that dredging various size and shape ponds within a cattail marsh has on the abundance, diversity, and composition of waterfowl within Braddock Bay WMA during late winter into spring. We used a combination of point-count and trail camera surveys to estimate spring waterfowl abundance and composition of dredged ponds, undredged, natural ponds, and large open-water areas occurring throughout the Braddock Bay WMA. We also assessed differences in vegetation, invertebrates, and physical habitat characteristics among the different habitat types of Braddock Bay WMA. We found that waterfowl area-corrected cumulative use days, species, richness, and species diversity were similar or greater in dredged ponds than natural ponds. We also found that vegetation species richness, percentage vegetated, invertebrate order richness, and invertebrate abundance/cubic meter within dredged ponds was similar or greater than natural ponds in Braddock Bay WMA, in each year of sampling. These findings suggest that dredging treatments to increase interspersion within Braddock Bay WMA provide

additional open-water habitat that offers comparable food and cover resources supports similar waterfowl diversity, richness, and use to pre-existing natural ponds. We found that shoreline development index and interspersion were significant explanatory variables for predicting square root transformed waterfowl cumulative use days per hectare for dredged and natural ponds. Using trail cameras proved to be an efficient, non-nonintrusive method for the long-term monitoring of waterfowl in areas that were inaccessible to point-count surveys. Trail camera data provide an estimate of waterfowl use and diversity within the dredged and natural ponds of Braddock Bay WMA. The multivariate relationships we found between wetland habitat characteristics and waterfowl use and diversity can be used to help influence future wetland restoration and waterfowl management efforts within Braddock Bay WMA and other coastal wetlands throughout the Great Lakes region.

INTRODUCTION

Ecologists have established that as the area surveyed increases, species richness also increases (Boecklen 1986, Blake and Karr 1987, Ricklefs, and Lovette 1999). In forest habitats, area has been found to be the most important variable accounting for increased species richness in breeding and migratory habitats of birds (Blake and Karr 1987). Brown and Dinsmore (1986) found that within marsh habitats, bird species richness was greater in wetlands of larger size and that the frequency of a variety of bird species of the family Anatidae (Ducks, Geese, and swans) was positively related to marsh size including Mallards (*Anas platyrhynchos*, Ruddy Ducks (*Oxyura jamaicensis*), Blue-winged Teal *(Anas discors*), and Canada Geese (*Branta canadensis*). Larger wetlands may provide more foraging opportunities and security from predators for waterfowl than smaller wetlands (LaGrange and Dinsmore 1989).

One of the hypotheses used to explain the species-area relationship is the "habitat diversity" hypothesis (Williams 1964 as cited in Connor and McCoy 1979). The "habitat diversity" hypothesis proposes that as the area sampled increases, there is an increase in new habitat niches and their associated species (Williams 1964 as cited in Connor and McCoy 1979). This ultimately results in increased species richness with an increase in area (Williams 1964 as cited in Connor and McCoy 1979). With birds, habitat heterogeneity and patch size can be reliable predictors of species richness and composition (Boecklen 1986, Riffel et al. 2001). In lacustrine habitats of Alaska, waterfowl species richness increased with lake size, potentially due to an increase of resources and space (Roach and Griffith 2015). However, at the landscape scale, the importance of habitat heterogeneity is shown as waterfowl species richness is also correlated with closeness to rivers, the size of other lakes in the area, and the proportion of the lakes perimeter that is considered wetland (Roach and Griffith 2015). Habitat heterogeneity may increase waterfowl species richness due to the high variation in habitat and foraging preferences that can be seen within the great variety of waterfowl species (Roach and Griffith 2015).

Interspersion within a wetland has been correlated with an increase in density and diversity in marsh bird and waterfowl species (Kaminski and Prince 1984, Murkin et al. 1997, Rehm and Baldassarre 2007). Interspersion can be quantified as the amount of interface between emergent vegetation and open-water within a wetland (Rehm and Baldassarre 2007). Decreased interspersion in Great Lakes coastal wetlands can occur because of invasion of open-water habitat by the non-native narrowleaf and hybrid cattail (*T. angustifolia*, *Typha* × *glauca*) and common reed (*Phragmites australis*) (e.g., Peterson and Niemi 2007, Schummer et al. 2012). Invasion in Lake Ontario coastal wetlands by cattail and common reed can be partially attributed to an altered hydrology due to highly regulated lake-levels (Wilcox et al. 2008). A decrease in lake-level variability can increase the competitive advantage of invasive cattail, causing it to expand its range within a wetland (Wilcox et al. 2008). Dredging creates openings in large stands of cattail within Great Lakes wetlands that researchers have found increases interspersion and beneficial habitat for fish and wildlife (Schummer et al. 2012). The dredging of openings within invasive-plant monocultures directly increases the habitat heterogeneity of a wetland.

Spring-Stopover Wetlands and Food Resources

Spring-stopover wetland habitats provide important energy resources for waterfowl to continue northward migration and accumulate nutrients essential to successful reproduction (Anteau and Afton 2004, Beatty et al. 2016). Reserves of nutrients gained in spring-stopover wetlands, such as lipids, proteins, and minerals, can determine the immediate spring condition of migrating Anatidae (Ducks, Geese, and Swans) as they approach the breeding grounds (Anteau and Afton 2004). Waterfowl that have accumulated these nutrients at spring-stopover wetlands may nest earlier which in turn can increase the chance of successful nesting and clutch production (Straub et al. 2012, Tidwell et al. 2013). Waterfowl that have not accumulated these nutrients at spring-stopover wetlands could then have decreased reproductive success at breeding sites which could cause decreases in waterfowl populations (Anteau and Afton 2004). This importance of nutrient accumulation is reflected by a general pattern of increased foraging by waterfowl during spring migration (Arzel et al. 2005).

Cattail invasion can degrade these spring-stopover wetlands, and the food resources they offer, that are important for migrating waterfowl. For example, cattail invasion has been associated with less macroinvertebrate abundance (Lawrence et al. 2016), thus negatively affecting food availability for waterfowl. Temporally, the loss of invertebrate resources is most likely of importance to waterfowl during spring migration. Spring is a time when many species of waterfowl shift from a mostly plant matter diet, to one composed of mostly invertebrates (Arzel et al. 2005, Tidwell et al.

2013, Hitchcock Jr et al. 2021). A spring-diet of mostly invertebrates helps waterfowl to acquire and accumulate the aforementioned reserves of nutrients such as lipids, and proteins that will in turn lead to potential increases in reproduction and ultimately population levels (Anteau and Afton 2004, Tidwell et al. 2013, Beatty et al. 2016). Despite the importance of spring migratory stopover sites to a variety of waterfowl species, there is less information published on the ecology of waterfowl at these sites or the quality of habitat migratory stopover sites offer in the spring (Arzel et al. 2005).

In addition to aquatic invertebrates, aquatic vegetation also provides an important food resource for migratory waterfowl (Dick et al. 2004, Gettys et al. 2014). Species of particular importance to waterfowl include sago pondweed (*Stuckenia pectinata*) and eelgrass (*Vallisneria americana*) (Lovvorn 1989, Dick et al. 2004). The tubers of sago pondweed and eelgrass are of particular importance to the canvasback (*Aythya valisineria*) during the period of late fall to early spring (Lovvorn 1989). A variety of waterfowl species feed on the different parts of sago pondweed, including the seeds, foliage, and tubers (Anderson and Low 1976).

Waterfowl species can preferentially select habitats dominated by native aquatic vegetation over those dominated by invasive aquatic vegetation (Dick et al. 2004). This may be due to the increased invertebrate biomass and diversity that occurs on native aquatic vegetation such as star-leaved duckweed (*Lemna trisulca*) and sago pondweed in comparison to invasive aquatic vegetation such as Eurasian watermilfoil (*Myriophyllum spicatum*) (Krull 1970). Invasive aquatic plant species can outcompete native aquatic plants and decrease habitat heterogeneity required to sustain more abundant and diverse aquatic macroinvertebrate communities (Gettys et al. 2014). Although native vegetation such as common waterweed (*Elodea canadensis*) is not a directly important food source for waterfowl, it can indirectly supply a food source in the form of habitat for a variety of aquatic invertebrates (Krull 1970).

Great Lakes coastal wetland restoration and waterfowl

The deepwater and coastal wetland habitats of the Great Lakes region are important to breeding, migrating, and wintering populations of a great diversity of waterfowl species (Prince et al 1992). More than 30 species of waterfowl from the tribes Cygnini, Anserini, Anatini, Aythyini, Cairinini, Mergini, and Oxyurini use Great Lakes coastal wetlands at some point in the year (Prince et al. 1992). Since 1800, approximately 50% of coastal wetland area in the Great Lakes region have been lost (Albert 2003), and these wetlands are impacted by anthropogenic disturbance (e.g., Wolter et al. 2006). To reverse the loss and improve wetland habitats, specific programs - such as the Great Lakes Restoration Initiative in the United States- provide funding to a variety of agencies and non-profit organizations to implement restoration actions. One of the targets of restoration is to increase the amount of open-water area interspersed within marshes to improve waterfowl habitat. However, less is known about how waterfowl and other wetland characteristics are affected by these small-scale treatments, and how these habitats meet the needs of waterfowl at critical periods, such as spring migration. The coastal wetlands of Braddock Bay Wildlife

Management Area (WMA) on the southern coast of Lake Ontario provide migratory stopover habitat that is potentially very important for pre breeding season waterfowl. The dredging of channels and "ponds" (0.01 ha to 0.5 ha open-water areas in the cattail mat) in Braddock Bay, Buck Pond, and Long Pond have directly increased the amount of available open-water habitat as well as habitat heterogeneity within the Braddock Bay (WMA) and may have increased the amount of preferred habitat for waterfowl. Differences in habitat structure, and food availability in the form of vegetation and invertebrates, may be present within the different wetland habitat types offered within the Braddock Bay WMA. These differences in habitat structure and food availability may directly affect the abundance and diversity of waterfowl species present within wetlands (e.g., Kaminski and Prince 1981, Kaminski and Prince 1984, Arzel et al. 2005), such as those within the Braddock Bay WMA. Despite the impact that interspersion treatments of cattail, occurring in the Lower Great Lakes coastal wetlands as a restoration technique, could have on waterfowl within the Atlantic flyway, there are currently no monitoring protocols in place to determine the extent of waterfowl use corresponding to the variations within this technique as seen in the Braddock Bay WMA.

My goal was to evaluate if there were differences in waterfowl use and habitat characteristics of the various habitat types within the Braddock Bay WMA, during late winter and into spring migration. Ultimately, this will allow for future researchers and managers to assess potential selection of habitat type within Braddock Bay WMA by various species of waterfowl. I assessed the effects that dredging various size and shape ponds within a cattail marsh has on the abundance, diversity, and composition of waterfowl within Braddock Bay WMA during late winter into spring. At the local scale, the results of my research will be useful for determining the importance of habitats at the Braddock Bay WMA for a variety of waterfowl species following wetland restoration. At a larger scale, the results of my research can be used in future interspersion restoration efforts to meet objectives concerning the use and diversity of a variety of waterfowl species,

The first objective of this study was to estimate waterfowl abundance and composition of dredged ponds, undredged, open-water areas within the marsh (hereafter "natural ponds"), and the open-water lacustrine habitats (hereafter "large open-water areas") occurring throughout the Braddock Bay WMA. I hypothesized that overall waterfowl use, richness, and diversity of Braddock Bay WMA will be greater when comparing total available open-water habitat (dredged ponds, natural ponds, and large open-water areas) to available open-water habitat (natural ponds and large open-water areas) prior to restoration. Dredged ponds increase the amount of isolated open-water habitat as well as potentially increasing food resources in the form of submerged aquatic vegetation and aquatic invertebrates. I also hypothesized that area-corrected cumulative use days of dabbling ducks would be greatest in dredged ponds, and natural ponds in comparison to large open-water areas due to an increase in shallow water habitat preferred by dabbling species of Anatidae for foraging. The opposite relationship was expected for diving/sea duck species of Anatidae as they prefer deeper water habitats for foraging.

The second objective of my study was to assess differences in species richness and abundance of emergent and submerged aquatic vegetation among dredged ponds, natural ponds, and large open-water areas occurring throughout the Braddock Bay WMA. I hypothesized that richness and abundance of emergent vegetation would be similar between habitat types due to the prevalence of invasive cattail that dominates floating vegetation mats bordering the open-water habitats of Braddock Bay WMA. I hypothesized that the richness and abundance of submerged aquatic vegetation would be similar between different between habitat types due to my sampling in the spring being early in the growing season. I also hypothesized that vegetation-water interspersion would be positively correlated with waterfowl area-corrected cumulative use-days, species richness, and species diversity.

The third objective of this study was to compare invertebrate order richness and abundance among dredged ponds, natural ponds, and large open-water areas within the Braddock Bay WMA. I hypothesized that invertebrate order richness and abundance would be similar between habitat types due to a likely similar vegetation structure and temporally early sampling period.

The final objective of this study was to quantify differences in open-water area, average depth, distance to roads, and shoreline development index between dredged ponds, natural ponds, and large open-water areas within the Braddock Bay WMA. I hypothesized that waterfowl area-corrected cumulative use-days, species richness, and species diversity would be positively correlated with shoreline development index, open-water area, and distance to disturbance and particular waterfowl species would be found in shallow or deeper water depths depending on foraging preference.

METHODS

Study Site

The study site was a series of coastal wetlands on the southern shore of Lake Ontario near Greece, New York. The study included Braddock Bay, Long Pond, and Buck Pond of the approximately 860-hectare Braddock Bay Wildlife Management Area. The wetlands of Braddock Bay are classified as riverine open embayment coastal wetlands, whereas Long Pond and Buck Pond are barrier beach lagoon coastal wetlands (Albert et al. 2005). These coastal wetlands include large stands of emergent marsh dominated by hybrid cattail (*Typha* x *glauca*) and narrowleaf cattail (*Typha angustifolia*) that contain less open-water habitat and border large open-water areas. In 2014, U.S. Fish and Wildlife Service excavated small open-water ponds and channels within the cattail-dominated emergent marsh to during a restoration project (Wilcox and Bateman 2018). Ducks Unlimited and associated partners conducted similar restoration efforts at nearby coastal wetlands including Buck Pond to the east of Braddock Bay (TNC 2015). Upon excavation of these ponds and channels, small areas of mixed-emergent marsh were seen (Wilcox and Bateman 2018). Dredged pond size varies within Braddock Bay WMA but most dredged ponds are about 0.01 to 0.5 ha. Large open-water areas for Braddock Bay, Buck Pond, and Long Pond range from 10 to 180 ha. Within Braddock Bay, Long Pond, and Buck Pond, there are also a small number of pre-existing natural ponds within the large stands of invasive cattail.

Dredged ponds of different size ranges were randomly selected for sampling in Braddock Bay, Long Pond, and Buck Pond (Table 1). A cutoff was set at 0.25 ha, to evenly split the dredged ponds into two size classes: large ponds and small ponds. Natural ponds were also randomly selected for Braddock Bay and Buck Pond. Natural ponds were not split into size classes due to there being a small quantity of natural ponds in Braddock Bay, and no natural ponds present in Long Pond. The selected dredged and natural ponds of Braddock Bay, Buck Pond, and Long Pond, as well as their corresponding large open-water habitat, were surveyed for waterfowl, vegetation, invertebrates, and physical habitat characteristics in the spring of 2020 and 2021 (Figures 1-3.).

Waterfowl Surveying

I conducted ground survey counts in the late winter and spring of 2020 and 2021 to study use of habitats by waterfowl in the Braddock Bay WMA. The migration chronology of waterfowl within the Braddock Bay WMA dictated the start and end of my surveys for each individual field season. I estimated the length of migration within Braddock Bay WMA using eBird (2019) bar charts of waterfowl observations from January to June of 2010 to 2019. Using this information, I estimated that the migration within Braddock Bay WMA occurs on average from mid-February to May (eBird 2019). This was determined by a major increase in waterfowl numbers at the beginning of migration and a sudden decrease in waterfowl numbers at the end of migration. I conducted my waterfowl surveys within the Braddock Bay WMA resembling this time period. However, due to interannual variation in waterfowl migration, weather, and ice coverage in the Braddock Bay WMA, start and end dates had the potential to vary between my sampling years. Therefore, beginning in mid-January, I monitored ice cover of the ponds and large open-water areas within the Braddock Bay WMA. The official start of my data collection was determined by monitoring when ice coverage began to recede in the dredged and natural ponds, allowing all of the open-water habitats to be used by waterfowl. The end of spring migration, marking the end of my waterfowl surveying, was determined by a steep decline in waterfowl seen during point-count surveys, followed by consistently low waterfowl presence.

Point-count surveys of the large open-water areas of Braddock Bay, Long Pond, and Buck Pond began when ice began to recede within the dredged and natural ponds, allowing them to be used by waterfowl. This ensured that my data collection represented the same temporal scale while surveying for waterfowl amongst the different habitat types. Point-count surveys were conducted every six to ten days, between sunrise and three and a half hours after sunrise, aligning closely to the time period of other published waterfowl surveys (e.g., Murkin and Kadlec 1986, LaGrange and Dinsmore 1989, Grover and Baldassare 1995). Point count surveys were done at two points for Braddock Bay, three points for Long Pond, and one point for Buck Pond to ensure maximum coverage of the large open-water areas of each wetland. A 20-minute waterfowl count was done at each of the six designated points, where we recorded all waterfowl seen, including those flying at tree height or below (Grover and Baldassarre 1995). Point counts were done during clear or cloudy conditions and not during periods of rain or high wind (Mulhern et al. 1985).

Coinciding with the point-count surveys of the large open-water areas, trail cameras set on time-lapse mode were placed in each of the selected dredged and natural ponds similar to the methods of (Cowardin and Ashe 1965, Murkin and Kadlec 1986). Trail cameras were set up in each of the selected ponds when ice cover was still present, to prevent missing the arrival of waterfowl and were taken down aligning with the end of spring-migration as determined by my point-count surveys. The trail cameras were programmed on time lapse mode to take a picture every 15 minutes, during the same time that point-count surveys were being conducted (sunrise to 3.5 hours after sunrise), to ensure we were estimating waterfowl use of the different habitats on the same temporal scale. Time lapse mode was set to a wide enough time range to account for the sampling window of sunrise to 3.5 hours after sunrise, for the entire sampling season. I used Stealth Cam XV4X trail cameras for my study. The Stealth Cam XV4X trail camera was set to take 32-megapixel pictures which allowed for photo quality sufficient enough to identify waterfowl at further distances from the camera. The trail cameras were set up at the vegetation-water interface, on a PVC pipe driven into the bottom sediment, facing toward the ponds. Trail cameras were placed facing north, when possible, to prevent a decrease in photo quality due to the sun. A known area of open-water directly in front of each trail camera was marked with flagging and/or floating markers within each dredged and natural pond to allow for the calculation of a known area that the camera captures. All waterfowl were counted and identified to species within this known area. Using the data collected from the trail camera surveys and point-count estimates, species richness and species diversity (Shannon-Weiner Index, $H^{'}$) were calculated for each pond and large open-water area sampled.

Cumulative waterfowl use days (UDs, Equation 1)) was calculated for each habitat type using a method similar to Stafford et al. (2007; 2010) . Waterfowl use days are an estimate of total waterfowl use extrapolated over a time period of interest, calculated by taking the sum of the moving average for each waterfowl species (Stafford et al. 2010). Cumulative use days were calculated for each dredged pond, natural pond, and large open-water area for each species of waterfowl, giving me an estimate of waterfowl use of each habitat type by species over the entire period of observation.

$$UD_{hs} = \sum_{t=0}^{N}((n_{(t)hs} + n_{(t+1)hs})/2) \times (d_{(t+1)} - d_t)$$ Equation 1.

Specifically, UDs were calculated using number of birds *n* from survey *t* (weekly point-count or day of trail camera survey) at site *h* (individual dredged ponds, natural ponds, or large open-water areas) of species *s* (Mallard (*Anas platyrhynchos*), Northern Pintail (*Anas acuta*), etc.) where the average of $n_{(t)}$ and $n_{(t+1)}$ is multiplied by the number of days between surveys. Days between surveys were calculated by subtracting the Julian date *d* of survey *t* from the Julien date *d* of survey *(t+1).* There was approximately seven days in between point-count surveys and trail camera surveys included for calculation of UDs. The waterfowl surveying methods of point-count and trail cameras used the same equation, with the only difference being the interpretation of "*n*". For the point-count surveys, "*n*" equals the number of individuals of each species seen. For the trail camera surveys, "*n*" equals the maximum number of individuals of the species seen within the 3.5-hour time period trail cameras are in operation for that given day .

Cumulative use days were corrected for area to standardize waterfowl numbers between the different sized open-water habitats, allowing us to make comparisons of waterfowl use by habitat type (LaGrange and Dinsmoore 1989, Stafford et al. 2007). To do this, the area of open-water that each trail camera was capturing in a picture was calculated based on a known marked off area within each open-water habitat. Cumulative use-day values for each species were then multiplied by (1/Known Area) to calculate cumulative use days per hectare of open-water habitat.

These area corrections of cumulative use days, for each habitat type, allowed for comparisons of waterfowl use to be made.

Vegetation Surveying

Surveying for vegetation bordering and within each habitat type (dredged ponds, natural ponds, and large open-water areas) was conducted once in the spring of 2020 and again in the spring of 2021. Within each habitat type, three transects, starting at randomly selected points, were conducted. Each vegetation transect began with the placement of a 1x1 m quadrat in the emergent zone, directly adjacent to the open-water of the dredged pond, natural pond, or large open-water area. The second quadrat of the vegetation transect was placed at the start of the submerged aquatic zone, directly adjacent to the start of emergent vegetation. The third quadrat of the transect was placed within the submerged aquatic zone, 10 m away from the emergent vegetation-water interface.

Data collected within the emergent vegetation quadrats included species presence, percent cover of all species, water depth, standing dead percentage, and organic depth. Submerged aquatic vegetation was sampled by raking and identifying all submerged aquatic vegetation falling within the 1x1 m quadrat. Floating vegetation falling within the submerged aquatic vegetation quadrats was also quantified. Data collected within submerged aquatic vegetation quadrats included species presence, percent cover of all species, and water depth. Depth measurements were taken in each quadrat using a ten-foot-long piece of aluminum conduit that was marked in 10 cm increments. The data we collected during vegetation surveying allowed us to calculate species richness and abundance metrics for the emergent and submerged aquatic vegetation zones for the open-water area within each habitat type.

Aquatic invertebrate sampling

I collected aquatic invertebrates once during my waterfowl observation windows using a standard sweep net following methods similar to Turner and Trexler (1997). The timing of aquatic invertebrate sampling aligned with vegetation sampling that took place in the spring of 2020 and 2021. The sweep net method is a method commonly used for collecting invertebrates in wetland habitats (e.g., Schummer et al. 2012, Straub et al. 2012). Three 1 m sweeps, directly adjacent to the third vegetation quadrat of each transect, were done to collect invertebrate samples within each dredged pond, natural pond, and large-open-water area. Samples were processed in the field using a 500 µm sieve bucket following the method described by Straub et al. (2012). Invertebrates retained by the 500 µm sieve bucket were preserved in vials using a 70% ethanol solution. Invertebrates were then identified to order and enumerated using a microscope in the laboratory. The data collected allowed us to evaluate the composition and availability of aquatic invertebrates, within the open-water area of each habitat type. Invertebrate abundance per cubic meter (IAm^3, Equation 2.) was calculated for each dredged pond, natural pond, and large open-water area where IA=the mean sampled invertebrate abundance, and A= the area sampled by the D-net.

$$IAm^3 = IA\ x\ (\frac{1}{A}) \qquad \text{Equation 2.}$$

Physical Habitat Characteristics

Average depth was calculated for the selected dredged and natural ponds using data collected during submerged aquatic vegetation sampling. Open-water area, length of emergent vegetation-water interface, and minimum distance to disturbance (roads, buildings, etc.) of each selected dredged pond, natural pond, and large open-water area within Braddock Bay, Buck Pond, and Long Pond was quantified using ESRI ArcGIS. Shoreline development index (SDI, Equation 3.) was calculated following the methods of Kaminski and Prince (1984) where L= the length of emergent vegetation-water interface for each dredged or natural pond, and A= the area of open-water within each dredged or natural pond. As shoreline development increases, the cover providing emergent zone surrounding each open-water area is more irregular and vegetation-water interspersion increases (Kaminski and Prince 1984). Shoreline development index values closest to a value of one, have a shoreline development most similar to a circle.

$$SDI = \left(\frac{L}{2\sqrt{\pi \bullet A}}\right) \quad \text{Equation 3}$$

DATA ANALYSES

Chronology and Composition of Braddock Bay WMA Waterfowl

To obtain a better understanding of the spring migration chronology of waterfowl in the Braddock Bay WMA, I plotted date vs. cumulative waterfowl abundance for the entire Braddock Bay WMA. Cumulative waterfowl abundance was split by site and by foraging guild. This was done for the spring of 2020 and 2021 . Migration chronology and composition data was descriptively analyzed through graphs.

Comparison of Waterfowl Use and Diversity among pond types of Braddock Bay Wildlife Management Area

To compare the mean waterfowl cumulative use days/ha (WCUD/ha), waterfowl species richness (S), and waterfowl diversity (H’) of each pond type, I used two, two-factor ANOVAs with replication, per metric, for each year of sampling. The factors for these ANOVAs were pond type and site. The first ANOVA was for total WMA and had two levels: small-dredged and large-dredged. The second ANOVA was for Braddock Bay and Buck Pond only and had three levels: small-dredged pond, large-dredged pond, and natural pond. A significance threshold of ($p<0.05$) was set to determine if there were significant differences in WCUD/ha, waterfowl species richness, or waterfowl diversity between pond types. Waterfowl species richness ANOVAs were completed in Microsoft Excel. WCUD/ha and H' ANOVAs were completed in R-studio.

Comparison of Vegetation species richness and abundance among pond types of Braddock Bay Wildlife Management Area

To compare the mean vegetation species richness and mean vegetated-percent of each pond type, I used two individual, two-factor ANOVAs with replication, per metric, for each year of sampling. The factors for these ANOVAs were pond type and site. The first ANOVA was for total WMA had two levels: small-dredged and large-dredged. The second ANOVA was for Braddock Bay and Buck Pond and had three levels: small-dredged pond, large-dredged pond, and natural pond. A significance threshold of ($p<0.05$) was set to determine if there were significant differences in vegetation species richness and vegetated-percent between the different pond types. A Post-hoc Tukey's HSD test was run for the ANOVAs with three levels to determine which mean species richness or mean vegetated-percent by pond type differed significantly from one another. All ANOVAs were completed in Microsoft Excel.

Comparison of invertebrate order richness and abundance among pond types of Braddock Bay Wildlife Management Area

To compare mean invertebrate order richness and mean invertebrate abundance/m^3 of each pond type, I used two individual, two-factor ANOVAs with replication, per metric, for each year of sampling. The factors for these ANOVAs were pond type and site. The first ANOVA was for total WMA and had two levels: small-dredged and large-dredged. The second ANOVA included only Braddock Bay and Buck Pond and had three levels: small-dredged pond, large-dredged pond, and natural pond. A significance threshold of ($p<0.05$) was set to determine if there were significant differences in mean invertebrate order richness or invertebrate abundance between pond types. If there was significance found, a Post-hoc Tukey's HSD test was run to determine which mean invertebrate order richness or invertebrate abundance by pond type was differed significantly from one another. All ANOVAs were completed in Microsoft Excel.

Waterfowl use and diversity in Braddock Bay Wildlife Management Area

I used multiple linear regression analyses using 'MASS' package of "R" 4.3.3 (R Core Team 2023) to identify which habitat variables (predictor variables) best predicted area-corrected cumulative use days of waterfowl, waterfowl species richness, and Shannon-Weiner diversity index of waterfowl (response variables). To evaluate the degree of correlation among the predictor variables, I used Pearson's correlation coefficient if the habitat variables were normally distributed; otherwise, I used Spearman's rank correlation. To ensure that there was no multicollinearity among predictor variables, I retained invertebrate abundance/m^3, vegetation species richness, area of open-water, average depth, shoreline development index, interspersion, and distance to disturbance.

I used stepwise selection for the multiple regression, where independent variables are added one at a time if they meet statistical criteria, and removed if they no longer contribute significantly. The selection results in the specification of a best fitting model.

Selected candidate models including different explanatory variables were then tested against the final stepwise model to determine which model(s) best explained waterfowl cumulative use days/ha, waterfowl species richness, and Shannon-Weiner diversity of waterfowl (Table 13). This determination was based on the model with the lowest AICc value using the 'AICcmodavg' package in R (Mazzerole 2023), which indicated the most parsimonious model (Burnham and Anderson 2002). Models within 2 units of AICc were not considered to be different from one another.

I interpreted differences among habitat type and waterfowl species composition using non-metric dimensional scaling (NMDS) ordination using the 'vegan' package in "R" (Oksanen et al. 2022). The exploratory NMDS ordination used a distance or dissimilarity matrix to plot my data on species, and habitat type on a multidimensional scale. The NMDS was run multiple times to assess the number of dimensions (2-3) that produced the NMDS configuration with the least associated stress. The NMDS configuration was only interpreted if the stress was between 0 and 0.2 (Quinn and Keough 2002). Points on the multidimensional scale of species and habitat type (small-dredged pond, large-dredged pond natural pond, and large open-water area), that were closer together, were considered to be more similar than points that were further apart. The measured habitat variables were then overlain on the NMDS ordination as vectors to help visualize the habitat variables that were associated with each habitat type as well as waterfowl community composition.

RESULTS

Chronology and Composition of waterfowl using Braddock Bay Wildlife Management Area

The peak abundance of waterfowl in Braddock Bay WMA for each year of sampling occurred during the first point count after most ice had melted, which took place on 3/10 in 2020 and 3/15 in 2021 (Figure 4., Figure 6.). On these dates, waterfowl abundance across all sites and pond types was 1,649 individuals in 2020 and 4,125 individuals in 2021 (Figure 4., Figure 6.). Of the 1,649 individuals on 3/10/2020, 10.92% were dabbling ducks, 58.69% were diving ducks, 25.96% were Geese, and 4.43% were swans (Figure 5.). Out of 4,125 individuals on 3/15/2021, 16.75% were dabbling ducks, 57.19% were diving ducks, 16.68% were Geese, and 9.38% were swans (Figure 6.) After the first point count, each year showed a trend of declining waterfowl abundance as the sampling season progressed (Figure 4., Figure 6.). On 5/7/2020 and 5/9/2021, the final point counts of each sampling season, waterfowl abundance was 195 individuals in 2020 and 358 individuals in 2021 (Figure 4., Figure 5.). On the final point count date of each sampling season, waterfowl composition was dominated collectively by Canada Geese and Mute Swans (*Cyngus olor*), with few dabbling or diving ducks remaining. On 5/7/2020, waterfowl composition consisted of 8.21% dabbling ducks, 11.79% diving ducks, 28.21% Geese, and 51.79% swans (Figure 5.). On 5/9/2021 waterfowl composition consisted of 25.46% dabbling ducks, 20.30% diving ducks, 29.15% Geese, and 25.09% swans (Figure 7.). The average waterfowl composition in the 2020 field season across all sites and pond types was 8.23% dabbling ducks, 58.77% diving ducks, 16.50% Geese, and 16.50% swans. In comparison the average waterfowl composition in the 2021 field season across all sites and pond types was 41.71% dabbling ducks, 30.98% diving ducks, 10.90% Geese, and 16.42% swans (Figure 7.).

Comparison of Waterfowl Use and Diversity among pond types of Braddock Bay Wildlife Management Area

Mean waterfowl cumulative use days/ha (WCUD/ha), calculated from trail camera data, were not significantly different between small-dredged, large-dredged, or natural pond types in the spring of 2020 or 2021 (Figure 8.). The mean WCUD/ha across all Braddock Bay WMA small-dredged ponds was 9,575.7 ± 2,867.8 (SE) WCUD/ha in 2020 and 14,971 ± 4,423.3 WCUD/ha in 2021. Total Braddock Bay WMA large-dredged pond mean WCUD/ha was 19,352 ± 15,166 WCUD/ha in 2020 and 10,848 ± 2,838.5 WCUD/ha in 2021. Natural ponds across Braddock Bay and Buck Pond had a mean WCUD/ha of 12,222 ± 3,628.0 WCUD/ha in 2020 and 13,809

± 4,151.4 WCUD/ha in 2021. Total Braddock Bay WMA Shannon-Weiner diversity index of waterfowl (H') was not significantly different between total WMA small-dredged ponds and large-dredged ponds, or between small-dredged, large-dredged, and natural ponds within Braddock Bay and Buck Pond, in either sampling year (Table 2., Table 3.). Waterfowl species richness was significantly higher ($F = 17.5$, $P \leq 0.01$) in total WMA large-dredged ponds than in total WMA small-dredged ponds in the spring of 2020 (Table 2, Figure 9). Total Braddock Bay WMA species richness (S) in 2020 was 4.3 ± 0.44 (SE) S for small-dredged ponds and 7.0 ± 0.47 S for large-dredged ponds. Total Braddock Bay WMA species richness was not significantly different between small-dredged and large-dredged ponds in 2021 or between small-dredged, large-dredged, and natural ponds within Braddock Bay and Buck Pond in either sampling year (Table 2., Table 3.).

The mean WCUD/ha across all Braddock Bay WMA large open-water areas was 112.35 ± 44.107 WCUD/ha in 2020 and 299.21 ± 142.88 WCUD/ha in 2021 (Table 4). Mean waterfowl species richness for total Braddock Bay WMA large open-water areas was similar between years with values of 20.3 ± 2.0 S in 2020 and 19 ± 1.7 S in 2021 (Table 4.). Shannon-Weiner diversity index of waterfowl for total Braddock Bay WMA large open-water areas was also similar between years with values of 2.1 ± 0.13 (SE) H' for 2020 and 2.1 ± 0.23 H' for 2021 (Table 4.). Total Braddock Bay WMA species richness across sampling years was 26 species of waterfowl (Table 5). Split by pond type, large open-water areas, small-dredged ponds, large-dredged ponds, and natural ponds had species richness of 26, 15, 13, and

15, respectively (Table 5). White-winged scoter (*Melanitta dehlandi*) was the only waterfowl species present in 2020 that were not present in 2021 (Table 5). In comparison, Blue-winged teal, and Eurasian Wigeon (*Anas penelope*) were present in 2021 but were not recorded in 2020 (Table 5). Six waterfowl species including the American Black Duck (*Anas rubripes*), American Wigeon (*Mareca americana*), Canada Goose, Hooded Merganser (*Lophodytes cucullatus*), Mallard, Mute Swan, and Wood Duck (*Aix sponsa*) were seen across all pond types in each sampling year (Table 5).

Comparison of Vegetation species richness and abundance among pond types of Braddock Bay Wildlife Management Area

Mean vegetation species richness (S) was not significantly different between total Braddock Bay WMA small-dredged and large-dredged pond types in spring of 2020 or 2021 (Table 6.). Vegetation species richness for total WMA small-dredged ponds was 6.9 ± 0.59 (SE) S in 2020 and 7.0 ± 0.71 S in 2021 (Table 6). In comparison, mean vegetation species richness for total WMA large-dredged ponds was 7.7 ± 0.44 S in 2020 and 6.0 ± 0.33 S in 2021 (Table 6.). The mean percent vegetated (%) of total WMA small-dredged and large-dredged ponds was not significantly different in 2020 with values of 28.42 ± 6.85 (SE) % in small-dredged ponds, and 35.59 ± 7.31 % in large-dredged ponds (Table 6.). In the spring of 2021, mean percent vegetated was significantly higher ($F = 12.3$, $P \leq 0.01$) in large-dredged ponds (32.95 ± 3.99 %) than in small-dredged ponds (19.95 ± 2.73 %) (Figure 10., Table 6).

There was no significant difference in mean vegetation species richness or mean percent vegetated between large-dredged, small-dredged, and natural pond types within Braddock Bay and Buck Pond in 2020 (Table 7.). In the spring of 2021, mean vegetation species richness within Braddock Bay and Buck Pond was significantly higher in small-dredged ponds than in large-dredged or natural ponds (Figure 11, Table 7: $F = 9.5$, $P \leq 0.01$). Braddock Bay and Buck Pond mean vegetation species richness in 2021 was 8.2 ± 0.60 S for small-dredged ponds, 5.8 ± 0.40 S for large-dredged ponds, and 5.5 ± 0.34 S for natural ponds (Table 7.). Braddock Bay and Buck Pond mean percent vegetated in the spring of 2021, was significantly higher in large-dredged ponds (36.22 ± 5.08 %) than natural ponds (15.17 ± 3.55 %) with small-dredged ponds (23.13 ± 2.98 %) having no significant difference between large-dredged or natural ponds (Figure 12, Table 7: $F = 9.7$, $P \leq 0.01$).

Mean vegetation interspersion (m) for total Braddock Bay WMA was 169.6 ± 23.54 (SE) m for small-dredged ponds, 320.4 ± 37.37 m for large-dredged ponds, 332.6 ± 97.94 m for natural ponds, and 6,260 ± 1,040 m for large open-water areas (Table 8.) Emergent zone standing dead vegetation, dominated by Typha was prevalent in the spring of 2020 and 2021 (Table 8.). In the spring of 2020, emergent zone standing dead percentage was 81.85 ± 4.13 (SE) % in small-dredged ponds, 74.26 ± 4.84 % in large-dredged ponds, 68.33 ± 4.30 % in natural ponds, and 62.78 ± 2.00 % in large open-water areas (Table 8.) Similar to 2020, emergent zone standing dead percentage in 2021 was 61.48 ± 3.86 % for small-dredged ponds, 71.85 ± 5.03

% for large-dredged ponds, 65.83 ± 8.02 % for natural ponds, and 56.67 ± 15.03 % for large open-water areas (Table 8.).

Comparison of invertebrate order richness and abundance among pond types of Braddock Bay Wildlife Management Area

Mean invertebrate order richness (O) and mean invertebrate abundance/cubic meter (IA/m^3) were not significantly different between small-dredged, large-dredged, and natural pond types in the spring of 2020 or 2021 (Table 9, Table 10.). Mean invertebrate order richness was 6.00 ± 0.37 (SE) O in 2020 and 7.1 ± 0.51 O in 2021 for small-dredged ponds, and 6.2 ± 0.57 O in 2020 and 8 ± 0.50 O in 2021 for large-dredged ponds (Table 9.). In comparison, mean invertebrate order richness of natural ponds was 4.8 ± 0.31 O in 2020 and 5.5 ± 0.92 O in 2021 (Table 10). Mean invertebrate abundance was 1,507 ± 535 (SE) IA/m^3 in 2020 and 2,855 ± 1,278 IA/m^3 in 2021 for small-dredged ponds, and 1,459 ± 463.0 IA/m^3 in 2020 and 4,608 ± 3,003 IA/m^3 in 2021 for large-dredged ponds (Table 9.). In comparison, mean invertebrate abundance of natural ponds was 1,029 ± 234 IA/m^3 in 2020 and 560 ± 108 IA/m^3 in 2021 (Table 10).

Comparison of physical habitat characteristics among pond types of Braddock Bay Wildlife Management Area

The mean distance to disturbance (m) for total Braddock Bay WMA was 175.07 ± 60.42 (SE) m for small-dredged ponds, 204.72 ± 20.53 m for large-dredged ponds, 135.85 ± 28.60 m for natural ponds, and 0 m for large open-water areas (Table

11.). The mean open-water area for total Braddock Bay WMA was 0.11 ± 0.02 (SE) ha for small-dredged ponds, 0.35 ± 0.03 ha for large-dredged ponds, 0.54 ± 0.23 ha for natural ponds, and 127.00 ± 35.09 ha for large open-water areas (Table 11.). The mean shoreline development index (SD) for total Braddock Bay WMA was 5.47 ± 0.62 (SE) SD for small-dredged ponds, 7.67 ± 1.20 for large-dredged ponds, 9.51 ± 2.55 SD for natural ponds, and 1.59 ± 0.06 SD for large open-water areas (Table 11.). Mean water depth (cm) within Braddock Bay WMA was least across all pond types in the spring of 2021 in comparison to the spring of 2020 (Figure 13, Table 12). Mean water depth of the sampled submerged aquatic zone in the spring of 2020 was 90.74 ± 5.52 (SE) cm in small-dredged ponds, 105.19 ± 7.01 cm in large-dredged ponds, 89.44 ± 7.77 cm in natural ponds, and 116.67 ± 12.62 cm in large open-water areas (Figure 13., Table 12.). In the spring of 2021, mean water depth of the sampled submerged aquatic zone was 54.44 ± 11.89 cm in small-dredged ponds, 66.67 ± 6.41 cm in large-dredged ponds, 67.50 ± 14.75 cm in natural ponds, and 73.33 ± 10.18 cm in large open-water areas (Figure 13., Table 12.). Mean Lake Ontario water level was 75.29 m in the spring of 2020 and 74.62 m in the spring of 2021 (Table 12). Lake Ontario water level was above the long-term historical average in the spring of 2020 and below the long-term historical average in 2021 (Figure 14: International Joint Commission, 2021).

Waterfowl use and diversity in Braddock Bay Wildlife Management Area

Stepwise multiple regression for each waterfowl metric began with a model including the explanatory variables shoreline development index, open-water area (ha), average water depth (cm), plant species richness, interspersion (m), invertebrate abundance/m^3, distance to disturbance (m), site, and pond type with year as a factor. In the stepwise model for square root transformed waterfowl cumulative use days/hectare, shoreline development index, interspersion, and emergent zone standing dead percentage were all retained as explanatory variables. Shoreline development index and interspersion were significant explanatory variables within the model ($P \leq 0.05$). Emergent zone standing dead percentage and interspersion had a negative effect on WCUD/ha, and shoreline development index had a positive effect on WCUD/ha within the stepwise model. In the stepwise model for waterfowl species richness, shoreline development index, the site Buck Pond, year, average water depth, and interspersion, were all significant explanatory variables that were retained within the model. The site Long Pond and emergent zone standing dead percentage were also retained in the stepwise model for waterfowl species richness but were insignificant ($P \leq 0.05$) All retained explanatory variables for this model had a positive effect on waterfowl species richness except for shoreline development index. In the stepwise model for Shannon-Weiner index of waterfowl diversity, year was the only significant explanatory variables retained ($P \leq 0.05$). Plant species richness was also retained within the model for waterfowl diversity, although insignificant ($P = 0.12$). Year and plant species richness have a positive effect on waterfowl species diversity within the stepwise model.

Selected candidate models were tested against the stepwise models for square root transformed waterfowl cumulative use days/ha, waterfowl species richness, and waterfowl diversity (Table 13.) . Model selection for waterfowl cumulative use days/ha found that the stepwise model and candidate model four were significant models (AIC ≤ 2) in predicting square root transformed WCUD/ha with the stepwise model being the best model found (Table 14.). The stepwise model includes shoreline development index, interspersion, and the emergent zone standing dead percentage as explanatory variables for predicting WCUD/ha. Candidate model four, the second-best model, contains the explanatory variables area (ha), shoreline development index, and interspersion. Model selection for waterfowl species richness found that the stepwise model was the only significant model in predicting waterfowl species richness within Braddock Bay WMA (Table 15: AIC ≤ 2). The stepwise model for waterfowl species richness retained the explanatory variables, shoreline development index, site, year, average water depth (cm), interspersion, and emergent zone standing dead percentage. Model selection for Shannon-Weiner diversity of waterfowl found that the stepwise model was the only significant model in predicting waterfowl diversity within Braddock Bay WMA (Table 16: AIC ≤ 2). The stepwise model for predicting waterfowl diversity retained the explanatory variables plant species richness and year.

In the NMDS, a two-dimensional solution was found using a matrix of data that was formed using a square root transformation and Bray-Curtis distance measure (Figure 15.). The best model was found on try zero with a stress of 0.1626 and was not repeated after 20 tries. Six environmental vectors were significant with the axis of the ordination (Figure 15.). These environmental vectors included shoreline development index, distance to disturbance (m), total area (ha), vegetation interspersion, submerged aquatic vegetation zone organic depth (cm), and submerged aquatic vegetation zone water depth (cm). Large open-water areas grouped closely on the ordination with mostly waterfowl species that are diving or sea ducks including White-winged Scoter (*Melanitta deglandi*), Redhead (*Aythya americana*), Common Goldeneye (*Bucephala clangula*), Lesser Scaup (*Aythya affinis*), and Red-breasted Merganser (*Mergus serrator*) (Figure 15.). Dabbling waterfowl species that grouped with Braddock Bay large open-water areas included American Black Duck (*Anas rubripes*), American Wigeon (*Mareca americana*), Blue-winged Teal, and Northern Shoveler (*Spatula clypeata*) (Figure 15.). The large open-water areas and previously mentioned waterfowl species, were correlated with an increase in the environmental variables total area (ha), vegetation interspersion (m), and submerged aquatic vegetation zone organic depth (cm). Waterfowl species that grouped closely with an increase in the environmental vector submerged aquatic zone water depth included Mute Swan, and Common Merganser (Figure 15.). In contrast, species including Mallards, Green-winged Teal, Gadwall, and Wood Duck grouped on the ordination further from the environmental vector showing increasing submerged aquatic vegetation zone water depth (Figure 15.). Waterfowl species that that grouped closely with the distance to disturbance (m) and shoreline development index environmental vectors were Canada Geese, and Hooded Mergansers (Figure 15.). Most of Braddock Bay WMA small-dredged, large-dredged, and natural pond types grouped closely together near the center of the ordination with outliers from each pond type being present within the ordination (Figure 15.).

DISCUSSION

Chronology and Composition of waterfowl Braddock Bay Wildlife Management Area

The coastal wetland habitats of the Great Lakes region, such as those in Braddock Bay WMA, are significant for a great diversity of waterfowl species in various stages of their migration and life cycle (Prince et al 1992). The aforementioned, Great Lakes coastal waterfowl diversity, was displayed during our sampling within Braddock Bay WMA, with 26 different waterfowl species being recorded in the springs of 2020 and 2021. Waterfowl abundance in Braddock Bay WMA for each year of sampling peaked immediately after ice melt, and then declined by approximately 90% by the second week of May. The waterfowl abundance peaks seen in both years of sampling at ice-out consisted of mostly diving duck species using the large open-water areas of Braddock Bay WMA. The waterfowl seen early in our sampling period during these peaks were likely wintering in the area or early northward migrants. Kaminski and Prince (1981) observed a similar pattern where mean waterfowl pair numbers declined to approximately 40% of peak numbers by late May. Within Braddock Bay WMA, diving ducks were more abundant in the spring of 2020 compared to 2021, with dabbling ducks showing the opposite relationship. Major water level fluctuations between years likely drove the waterfowl composition differences seen in this study. The high-water conditions of 2020 and low-water conditions of 2021 offered habitat and foraging conditions that were optimal for diving ducks and dabbling ducks respectively. Geese and swans accounted for similar waterfowl composition percentages in each year of sampling in Braddock Bay WMA. Although not statistically compared, differences in Braddock Bay WMA waterfowl abundance and diversity between years may be directly attributed to differences in habitat structure and food availability (Kaminski and Prince 1981, Kaminski and Prince 1984, Arzel et al. 2005).

Waterfowl use and diversity in Braddock Bay Wildlife Management Area

We found that shoreline development index and interspersion were the most significant explanatory variables when predicting waterfowl cumulative use days/hectare for dredged and natural ponds. As hypothesized, shoreline development index had a significant positive effect on waterfowl cumulative use days/ha. Supporting my research that waterfowl use is positively affected by shoreline development, Kaminski and Prince (1984) found that spring pair densities of various waterfowl species were directly related to increased shoreline development. Prior research has predominantly found that interspersion within a wetland has been correlated with an increase in density and diversity in marsh bird and waterfowl species (Kaminski and Prince 1984, Murkin et al. 1997, Rehm and Baldassarre 2007). In contrast, we found that interspersion in Braddock Bay WMA had a negative effect on waterfowl cumulative use days/hectare. A potential explanation for this is that waterfowl are potentially less likely to be captured by trail camera in a pond that has higher interspersion. Increased interspersion offers waterfowl additional emergent to open-water cover that likely decreases their susceptibility to being detected during trail camera photo processing. Also, waterfowl cumulative use days/ha was calculated using only waterfowl observations within a "known area" which could have negatively skewed waterfowl observations than what was actually present on the pond.

In comparison to waterfowl cumulative use days, shoreline development index had a significant negative relationship on predicting waterfowl species richness. Site, average water depth, and interspersion all had a positive relationship with waterfowl species richness. In contrast to waterfowl cumulative use days/ha, increased interspersion was positively related to waterfowl species richness. In alignment with our hypothesis and prior research (Kaminski and Prince 1984), interspersion was expected to have a positive effect on all of our waterfowl metrics rather than waterfowl species richness alone. Dredging treatments to increase open-water interspersion, such as those within the cattail dominated wetlands of Braddock Bay WMA, have been found to increase beneficial habitat for a variety of wetland animals, including but not limited to waterfowl (Kaminski and Prince 1984, Schummer et al. 2012). A positive effect of water depth on waterfowl species richness could be explained by the assumption that ponds with an increased water depth may have allowed for an increase in foraging opportunity for varying waterfowl species across foraging guilds. Stepwise regression to predict waterfowl Shannon-Weiner diversity was only significantly predicted by year whereby waterfowl species diversity was greater across all ponds in 2021 compared to 2020. Although an insignificant explanatory variable, plant species richness was also positively related to waterfowl species diversity. This could be explained by the assumption that as plant species richness increases, the food and cover provided by a dredged or natural pond can support a wider variety of waterfowl species. When candidate models were selected and tested against the stepwise models, the stepwise models were the best models at predicting their respective waterfowl metric. Waterfowl cumulative use days/ha was the only response variable that had a second significant model. In addition to interspersion, and shoreline development index, candidate model four also contained open-water area as a significant explanatory variable. We hypothesized that as open-water increases, waterfowl area-corrected cumulative use days, species richness, and species diversity would increase. This hypothesis was supported by prior research finding that the frequency and diversity of waterfowl is positively related to marsh size (Brown and Dinsmoore 1986).

Through non-metric dimensional scaling, we found that certain waterfowl species grouped closely with particular pond types within Braddock Bay WMA. We hypothesized that species of diving/sea ducks would preferentially use large-open-water areas in comparison to dredged or natural ponds due to a preference in foraging in deeper water habitats than dabbling species. The most obvious separation of waterfowl species and pond type upon analysis was the grouping of diving/sea duck dominated waterfowl species and large open-water area sites. The diving/sea duck species that grouped with large open-water sites were species that were almost exclusively found within large open-water sites and rarely found within dredged or natural ponds. However, several dabbling duck species that were not prevalent in dredged or natural ponds also grouped with large open-water area sites. This grouping of waterfowl species with large open-water area sites occurred along significant environmental vectors of increasing open-water area and perimeter. Each of these environmental variables were defining of large open-water area sites that were much larger than dredged ponds and natural ponds within Braddock Bay WMA. Waterfowl species that were commonly seen across pond types such as Mallards, Wood Ducks, Green-winged Teal (*Anas carolinensis*), and Ring-necked Duck (*Aythya collaris*) grouped together in the center of the ordination. These waterfowl species grouped with most of sampled natural and dredged ponds across sites as the species composition of these sites was often overlapping. This grouping of predominantly dabbling waterfowl species and sites grouped further away from the environmental vectors demonstrating increasing submerged aquatic zone depth, open-water area, and interspersion. We expected dabbling waterfowl species to be more prevalent in natural and dredged ponds that offer increased shallow water foraging opportunity than large open-water areas. Canada Geese and Hooded Merganser grouped with the increasing physical habitat characteristics, shoreline development index and distance to disturbance. This suggests that these waterfowl species preferentially selected wetlands that had a higher distance to disturbance and shoreline development index.

Waterfowl Use and Diversity among pond types at Braddock Bay Wildlife Management Area

Recent restoration efforts in Braddock Bay WMA and other Great Lakes coastal wetlands have been directed at increasing interspersion (Schummer et al. 2012). There was limited research on how waterfowl and other wetland characteristics are affected by dredging treatments, used to increase interspersion, within Braddock Bay WMA. Our findings demonstrate that dredged ponds in Braddock Bay WMA had similar waterfowl species diversity, richness and overall cumulative use days/ha compared to natural ponds during each spring of the study. This suggests dredging treatments to increase interspersion within Braddock Bay WMA provide additional open-water habitat that supports similar waterfowl diversity, richness, and use to pre-existing natural ponds. Dredged pond treatments to increase interspersion within Long Point, Lake Erie, had comparable or greater marsh bird species richness and diversity than natural wetland habitats (Schummer et al. 2012). Although Schummer et al. (2012) found this relationship in non-Anatidae marsh birds, it supports our similar results that dredged ponds had similar waterfowl diversity, richness, and use to natural ponds. Increased interspersion, such as through the dredging treatments in Braddock Bay WMA and Long Point, has been positively correlated with an increase in density and diversity in marsh bird and waterfowl species (Kaminski and Prince 1984, Murkin et al. 1997, Rehm and Baldassarre 2007).

Although waterfowl species richness was similar between dredged ponds and natural ponds in each year, waterfowl species richness of large-dredged ponds was greater than in small-dredged ponds in the spring of 2020. This finding aligns with bird species richness being positively related to size (Brown and Dinsmore 1986, Riffel et al. 2001). Specifically, the frequency and variety of Anatidae species such as Mallards and Canada Geese often have a positive relationship to marsh size (Brown and Dinsmoore 1986). While we were unable to statistically compare dredged/natural ponds to large open-water areas due to the differences in open-water area between the different habitats, raw counts of species richness and diversity of large open-water areas for each sampling year was greater in large-open-water areas than the different dredged and natural pond types within Braddock Bay WMA. The waterfowl species richness and diversity of Braddock Bay large open-water areas, further supports research that these waterfowl metrics are positively associated with larger wetlands. A potential explanation for this is that larger wetlands or larger open-water areas may provide more foraging opportunity and security from predators for waterfowl than smaller wetlands (LaGrange and Dinsmore 1989, Roach and Griffith 2015). An alternative or coinciding hypothesis for this would be the aforementioned "habitat diversity" hypothesis (Williams 1964 as cited in Connor and McCoy 1979). Increased habitat heterogeneity which can result from an increase in sampled area, has been shown to be positively correlated with bird species richness and composition (Boecklen 1986, Riffel et al. 2001). The larger open-water habitats Braddock Bay WMA large-dredged ponds, larger natural ponds and large open-water areas may exhibit greater habitat heterogeneity than small-dredged ponds.

Vegetation species richness and abundance among pond types of Braddock Bay Wildlife Management Area

Spring-stopover wetland habitats, such as the wetlands of Braddock Bay WMA provide important energy resources for waterfowl to continue migration and accumulate nutrients essential to successful reproduction (Anteau and Afton 2004, Beatty et al. 2016). Within spring-stopover wetland habitats, vegetation provides an important food resource for migratory waterfowl (Dick et al. 2004, Gettys et al. 2014) as well as cover from disturbance and predators. The second objective of our research was to assess differences in species richness and abundance of emergent and submerged aquatic vegetation among dredged ponds, natural ponds, and large open-water areas occurring throughout the Braddock Bay WMA, that may be associated with waterfowl habitat use. In support of our hypothesis, we found that vegetation species richness and percent vegetated within dredged ponds was as great or greater than natural ponds in Braddock Bay WMA in each year of sampling. This finding suggests dredging treatments to increase interspersion within Braddock Bay WMA provide additional open-water habitat that supports similar vegetation species richness, and percent vegetated to pre-existing natural ponds. Schummer et al. (2012) found similar vegetation results including that relative abundance, species richness, and species diversity of vegetation were as great, and in several cases even greater, at dredged ponds relative to other natural wetland habitats. Regardless of if the pond was natural or dredged, the emergent zone bordering all open-water areas within Braddock Bay WMA were consistently dominated by standing dead cattail. Lake Ontario coastal wetlands, such as Braddock Bay WMA, are dominated by cattail in part due to a decrease in water level ability that allows invasive cattail to outcompete native vegetation (Wilcox et al. 2008).

Invertebrate order richness and abundance among pond types of Braddock Bay Wildlife Management Area

Along with vegetation, aquatic invertebrates are an important food source for migratory waterfowl within spring-stopover wetlands, such as Braddock Bay WMA (Arzel et al. 2005, Anteau and Afton 2004 Tidwell et al. 2013, Beatty et al. 2016). Waterfowl using spring-stopover wetlands have a diet dominated by invertebrates to help acquire and accumulate nutrient reserves that will help lead to increases in reproduction success (Anteau and Afton 2004, Beatty et al. 2016, Tidwell et al. 2013). The third objective of our study was to compare invertebrate order richness and abundance among dredged ponds, natural ponds, and large open-water areas within the Braddock Bay WMA that may be associated with waterfowl habitat use. The findings of our research show that invertebrate order richness and density were similar among small-dredged, large-dredged, and natural ponds within Braddock Bay WMA. This finding supports our hypothesis and suggests dredged ponds and pre-existing natural ponds within Braddock Bay WMA offer similar invertebrate resources to waterfowl during this time period. This implies that if waterfowl are foraging for invertebrates within natural ponds and dredged ponds, they will have equal opportunity to acquire and accumulate nutrients. Schummer et al. (2012) found a similar pattern as us whereby relative abundance, species richness, and species diversity of aquatic invertebrates in Long Point, Lake Erie were as great, and in several cases even greater, at dredged ponds relative to other natural wetland habitats. The availability of spring food resources of vegetation and invertebrates, has been directly increased through the dredging treatments to increase interspersion with Braddock Bay WMA.

Physical habitat characteristics among pond types of Braddock Bay Wildlife Management Area

The fourth objective of our research was to quantify differences in physical habitat characteristics between dredged ponds, natural ponds, and large open-water areas within the Braddock Bay WMA that may be associated with waterfowl habitat use. The most influential physical habitat characteristic that made a distinction between pond types was open-water area. The average open-water area of our sampled small-dredged ponds was approximately 1/3 the size of the average large-dredged pond, and 1/5 the size of the average natural pond. The difference in average open-water area stressed the importance of adjusting waterfowl cumulative use days for area to allow for appropriate statistical comparisons. Distance to disturbance was consistent among the randomly selected ponds of each pond type within Braddock Bay WMA and did not appear to influence overall waterfowl cumulative use days. Large open-water area had a distance to disturbance of zero as they directly bordered major roads, houses, and marinas. Often times through sampling, the greatest concentrations of waterfowl within large open-water areas in Braddock Bay WMA were directly along human influenced shoreline.

In Braddock Bay WMA, shoreline development index was greatest in natural ponds, followed by large-dredged ponds, small-dredged ponds, and large open-water areas. Shoreline development index was expected to be greater in natural ponds due to lack of irregularity and circular shape that is commonly shown by dredged ponds within Braddock Bay WMA. The average water depth of the sampled submerged aquatic vegetation zone was an average of 1/3 less in the spring of 2021 than the spring of 2020 across all natural and dredged ponds in Braddock Bay WMA. Water level fluctuations can highly influence waterfowl abundance and diversity within a wetland, including spring waterfowl pair densities (Kaminski and Prince 1984). As a Great Lakes coastal wetland, the habitats of Braddock Bay WMA are directly influenced by the water levels of Lake Ontario which experienced extreme highs in the spring of 2020 and extreme lows in the spring of 2021 (NOAA 2024). Major water level variation between springs of sampling was likely the most important driver for differences in waterfowl use and composition.

Trail cameras as a method for surveying waterfowl

Using trail cameras in Braddock Bay WMA proved to be an efficient and non-intrusive method to estimate waterfowl use and diversity within dredged and natural ponds. Previous research efforts have used older camera technology to monitor waterfowl use and behavior (Cowardin and Ashe 1965, Murkin and Kadlec 1986) in a similar way to how we used time lapse mode on modern trail cameras. While our cameras were recording data for months at a time, the cameras used by Murkin and Kadlec (1986) were only able to record data one day at a time. Recently, McClinton

(2021) used similar trail camera methods to estimate fall waterfowl use within Michigan hunting areas. Through the use of time-lapse trail cameras, the research of McClinton (2021) in Michigan, and ours in Braddock Bay WMA collectively allowed for long-term waterfowl monitoring without disturbance of waterfowl. Along with decreasing waterfowl disturbance, trail cameras allowed us to survey areas of Braddock Bay WMA that were inconvenient or inaccessible to point count surveys that we conducted on large open-water areas.

While trail cameras offered us an alternative way of sampling dredged and natural ponds of Braddock Bay WMA, they also came with certain limitations Trail cameras required check ins of battery life and SD card space throughout the sampling season to limit potential data loss. Even with trail camera maintenance, data was still lost from batteries dying quickly, SD cards filling up faster than expected, and curious Canada Geese moving the camera to where it is facing away from the open-water of the dredged and natural ponds. In addition to issues discovered in the field, processing of trail camera data was very time consuming and involved analyzing over 50,000 images/sampling year. Although using cameras came with disadvantages, the ability to monitor long-term spring waterfowl use and diversity within hard to access, interior open-water areas is valuable to waterfowl monitoring and management of Braddock Bay WMA.

Conclusions and Implications

The goal of my research was to determine differences in waterfowl use and habitat characteristics between large open-water areas, natural ponds, and dredged ponds that were incorporated into the wetland to increase open-water interspersion within Braddock Bay WMA. Our comparison of dredged ponds and natural ponds within Braddock Bay WMA illustrates that dredging treatments to increase wetland interspersion can provide habitat that is of similar quality to the pre-existing natural ponds. This augmentation of usable open-water habitat within Braddock Bay WMA has the potential to directly increase the abundance and diversity of waterfowl during late winter into spring-migration and likely other times of the year. My research allowed for the assessment of variations of the ‘potholing” technique on vegetation, aquatic invertebrates, physical habitat characteristics, and waterfowl use and composition within Braddock Bay WMA, following restoration. On a local scale, my research helped us understand the importance and quality of Braddock Bay WMA as a spring-stopover for a diverse composition of waterfowl. The multivariate relationships we found between wetland habitat characteristics and waterfowl use and diversity can be used to help influence future wetland restoration and waterfowl management efforts within Braddock Bay WMA. I would recommend that managers of Braddock Bay WMA conduct further research with increased repetition and sample size to improve models and better understand the habitat needs and selection of waterfowl. At a regional or national scale, our results and methods can be used in future wetland restoration to meet objectives concerning use and selection of a variety of waterfowl within Braddock Bay WMA and other wetlands. I would recommend further research or replication of our methods to wetland managers that are working outside of Great Lakes coastal wetlands similar to Braddock Bay WMA.

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TABLES AND FIGURES

Tables

Table 1. Quantity of each habitat type sampled, within Braddock Bay, Long Pond, and Buck Pond.

Site	Large open-water area	Large-dredged pond	Small-dredged pond	Natural Pond
Braddock Bay	1	3	3	3
Buck Pond	1	3	3	3
Long Pond	2	3	3	0

Table 2. Mean waterfowl Shannon-Weiner diversity (H'), and waterfowl species richness for total WMA small-dredged (SD) and large-dredged (LD) pond types. Values are split by response variable and year.

Response Variable	Year	SD	LD	F	P-value
H'	2020	0.39 ± 0.14	0.56 ± 0.16	0.601	0.45
H'	2021	0.78 ± 0.15	0.75 ± 0.13	0.246	0.63
Species Richness	2020	4.3 ± 0.44	7.0 ± 0.47	17.5	<0.01
Species Richness	2021	4.8 ± 1.0	5.8 ± 0.57	0.692	0.42

Table 3. Mean waterfowl Shannon-Weiner diversity (H’), and waterfowl species richness for Buck Pond, and Braddock Bay, small-dredged (SD), large-dredged (LD), and natural (NT) pond types. Values are split by response variable and year.

Response Variable	Year	SD	LD	NT	F	P-value
H'	2020	0.58 ± 0.16	0.69 ± 0.20	0.55 ± 0.16	0.161	0.85
H'	2021	0.80 ± 0.17	0.67 ± 0.19	0.65 ± 0.17	0.245	0.79
Species Richness	2020	5.5 ± 1.4	6.4 ± 0.87	5.4 ± 1.2	3.02	0.09
Species Richness	2021	4.3 ± 0.56	6.6 ± 0.68	6.7 ± 1.0	0.069	0.93

Table 4. Waterfowl cumulative use days/ha, waterfowl species richness, and waterfowl species diversity (H’) of Braddock Bay WMA, split by site, treatment, and year. The shown values are averages of large open-water areas (LOWA) small-dredged (SD), large-dredged (LD), and natural (NT) treatments/site ± 1 SE. Values without a standard error value are single observations.

Treatment	Site	Year	CUDs/ha	Species Richness	H'
SD	BB	2020	9,638.5 ± 2,674.0	6.3 ± 1.9	0.74 ± 0.28
SD	BB	2021	19,789 ± 10,321	4.3 ± 0.88	0.95 ± 0.13
SD	BP	2020	17,079 ± 5,877.3	4.7 ± 2.3	0.42 ± 0.12
SD	BP	2021	10,379 ± 2,863.1	4.3 ± 0.88	0.66 ± 0.33
SD	LP	2020	2,009.4 ± 470.14	3.3 ± 0.88	0
SD	LP	2021	14,288	4	0.67
LD	BB	2020	5,292.8 ± 3,011.2	5.3 ± 0.88	0.67± 0.35
LD	BB	2021	17,094 ± 5,499.0	5.7 ± 0.67	0.93 ± 0.05
LD	BP	2020	3,483.0 ± 1,564.0	8.0 ± 1.0	0.72 ± 0.15
LD	BP	2021	7,058.4 ± 4,825.3	8.0 ± 0.0	0.27 ± 0.27
LD	LP	2020	56,158 ± 53,867	5.0 ± 1.0	0.24 ± 0.13
LD	LP	2021	7,128.0 ± 2,446.3	7.7 ± 0.88	0.90 ± 0.15
NT	BB	2020	12,298 ± 6,370.8	4.0 ± 1.5	0.37 ± 0.20
NT	BB	2021	6,241.9 ± 3,090.9	6.3 ± 2.0	0.46 ± 0.31

NT	BP	2020	12,108 ± 3,137.4	7.5 ± 0.50	0.82 ± 0.14
NT	BP	2021	21,376 ± 4,399.5	7.0 ± 1.0	0.83 ± 0.14
LOWA	BB	2020	125.82	24	2.3
LOWA	BB	2021	308.2	22	2.6
LOWA	BP	2020	181.12	20	2.20
LOWA	BP	2021	541.89	19	1.9
LOWA	LP	2020	30.116	17	1.9
LOWA	LP	2021	47.540	16	1.9
SD TOTAL	BBWMA	2020	9,575.7 ± 2,867.8	4.3 ± 0.44	0.39 ± 0.14
SD TOTAL	BBWMA	2021	14,971 ± 4,423.3	4.8 ± 1.00	0.78 ± 0.15
LD TOTAL	BBWMA	2020	19,352 ± 15,166	7 ± 0.47	0.56 ± 0.16
LD TOTAL	BBWMA	2021	10,848 ± 2,838.5	5.8 ± 0.57	0.75 ± 0.13
NT TOTAL	BBWMA	2020	12,222 ± 3,628.0	5.4 ± 1.2	0.55 ± 0.16
NT TOTAL	BBWMA	2021	13,809 ± 4,151.4	6.7 ± 1.02	0.65 ± 0.17
LOWA TOTAL	BBWMA	2020	112.35 ± 44.107	20.3 ± 2.0	2.1 ± 0.13
LOWA TOTAL	BBWMA	2021	299.21 ± 142.88	19 ± 1.7	2.1 ± 0.23

Table 5. Spring presence of waterfowl species within Braddock Bay WMA, separated by large open-water area (LOWA), small-dredged pond (SD), large-dredged pond (LD) and natural pond (NT). Presence in spring of 2020 and 2021 marked by "20" or "21".

Species	LOWA	LDP	NP	SDP
American Black Duck	20, 21	20, 21	20, 21	20, 21
American Wigeon	20, 21	20, 21	20, 21	20, 21
Blue-winged Teal	21		21	21
Bufflehead	20, 21	20	20, 21	20
Canada Goose	20, 21	20, 21	20, 21	20, 21
Canvasback	20, 21			
Common Goldeneye	20, 21			
Common Merganser	20, 21	20	20, 21	20, 21
Eurasian Wigeon	21			
Gadwall	20, 21	20, 21		20, 21
Greater Scaup	20, 21			
Green-winged Teal	20, 21	20, 21	20, 21	21
Hooded Merganser	20, 21	20, 21	20, 21	20, 21
Lesser Scaup	20, 21			
Long-tailed Duck	20, 21			
Mallard	20, 21	20, 21	20, 21	20, 21
Mute Swan	20, 21	20, 21	20, 21	20, 21

Northern Pintail	20, 21			21
Northern Shoveler	20, 21	20	20	20
Red-breasted Merganser	20, 21			
Redhead	20, 21		21	
Ring-necked Duck	20, 21	20	20, 21	20, 21
Ruddy Duck	20, 21		21	
Tundra Swan	20, 21			
White-winged Scoter	20			
Wood Duck	20, 21	20, 21	20, 21	20, 21

Table 6. Mean vegetation species richness and percent vegetated for total WMA small-dredged (SD) and large-dredged (LD) pond types. Percent vegetated is quantified as an average of all vegetation quadrats across a specific pond type. Values are split by response variable and year. Error bars set to ± 1 SE of the mean vegetation species richness and percent vegetated.

Response Variable	Year	SD	LD	F	P-value
Species Richness	2020	6.9 ± 0.59	7.7 ± 0.44	1.44	0.25
Species Richness	2021	7.0 ± 0.71	6.0 ± 0.33	3.12	0.10
Percent Vegetated	2020	28.42 ± 6.85	35.59 ± 7.31	1.66	0.22
Percent Vegetated	2021	19.95 ± 2.73	32.95 ± 3.99	12.3	<0.01

Table 7. Mean vegetation species richness and percent vegetated for Buck Pond, and Braddock Bay, small-dredged (SD), large-dredged (LD), and natural (NT) pond types. Values are split by response variable and year. Error bars set to ± 1 SE of the mean vegetation species richness and percent vegetated.

Response Variable	Year	SD	LD	NT	F	P-value
Species Richness	2020	7.5 ± 0.76	8.0 ± 0.58	7 ± 0.82	0.46	0.64
Species Richness	2021	8.2 ± 0.60_{A}	5.8 ± 0.40_{B}	5.5 ± 0.34_{B}	9.5	<0.01
Percent Vegetated	2020	39.24 ± 6.49	48.59 ± 5.05	42.26 ± 8.85	0.42	0.67
Percent Vegetated	2021	23.13 ± 2.98_{AB}	36.22 ± 5.08_{A}	15.17 ± 3.55_{B}	9.7	<0.01

Table 8. Braddock Bay WMA interspersion (m) and emergent zone standing dead (%), grouped by treatment and site. Treatments split by large open-water area (LOWA), small-dredged pond (SD), large-dredged pond (LD), and natural pond (NT). Values for interspersion and standing dead shown as the mean ± 1SE of each subset of treatment and site.

TREATMENT	SITE	Interspersion (m)	2020 Standing Dead	2021 Standing Dead
SD	BB	140.2 ± 12.00	81.67 ± 4.81	58.33 ± 1.67
SD	BP	198.5 ± 61.04	79.44 ± 4.75	70 ± 8.82
SD	LP	170.0 ± 43.90	84.44 ± 12.4	56.11 ± 6.55
LD	BB	311.8 ± 32.37	73.89 ± 5.80	73.33 ± 11.7
LD	BP	230.9 ± 24.48	81.67 ± 4.41	62.22 ± 8.68
LD	LP	418.6 ± 79.04	67.22 ± 13.3	80 ± 3.33
NT	BB	452.1 ± 180.17	67.22 ± 6.26	67.22 ± 16.1
NT	BP	213.1 ± 34.86	69.44 ± 7.22	64.44 ± 7.72
LOWA	BB	6,437	66.67	43.33
LOWA	BP	4,377	61.67	86.67
LOWA	LP	7,966	60	40
SD TOTAL	BBWMA	169.6 ± 23.54	81.85 ± 4.13	61.48 ± 3.86
LD TOTAL	BBWMA	320.4 ± 37.37	74.26 ± 4.84	71.85 ± 5.03
NT TOTAL	BBWMA	332.6 ± 97.94	68.33 ± 4.30	65.83 ± 8.02
LOWA TOTAL	BBWMA	6,260 ± 1,040	62.78 ± 2.00	56.67 ± 15.03

Table 9. Mean invertebrate order richness and invertebrate abundance/m^3 for total WMA small-dredged (SD) and large-dredged (LD) pond types. Values are split by response variable and year. Error bars set to ± 1 SE of the mean invertebrate order richness and invertebrate abundance/ m^3.

Response Variable	Year	SD	LD	F	P-value
Order Richness	2020	6.00 ± 0.37	6.2 ± 0.57	0.111	0.74
Order Richness	2021	7.1 ± 0.51	8 ± 0.50	2.29	0.16
Abundance/m^3	2020	1,507 ± 535	1,459 ± 463.0	0.004	0.95
Abundance/m^3	2021	2,855 ± 1,278	4,608 ± 3,003	0.337	0.57

Table 10. Mean invertebrate order richness and invertebrate abundance/m^3 for Buck Pond, and Braddock Bay, small-dredged (SD), large-dredged (LD), and natural (NT) pond types. Values are split by response variable and year. Error bars set to ± 1 SE of the mean invertebrate order richness and invertebrate abundance/ m^3.

Response Variable	Year	SD	LD	NT	F	P-value
Order Richness	2020	5.5 ± 0.34	6.0 ± 0.77	4.8 ± 0.31	1.23	0.33
Order Richness	2021	6.7 ± 0.71	7.6 ± 0.62	5.5 ± 0.92	1.73	0.22
Abundance/m^3	2020	1,705 ± 686	1,077 ± 155	1,029 ± 234	0.756	0.57
Abundance/m^3	2021	656 ± 128	1,438 ± 930	560 ± 108	0.844	0.45

Table 11. Braddock Bay WMA physical habitat characteristics, grouped by site and treatment. Treatments split by large open-water area (LOWA), small-dredged pond (SD), large-dredged pond (LD), and natural pond (NT). Values for physical habitat characteristics shown as the mean ± 1SE of each subset of treatment and site.

Treatment	Site	Distance to Disturbance (m)	Area (ha)	SDI
SD	BB	137.77 ± 48.23	0.08 ± 0.02	4.29 ± 0.08
SD	BP	253.23 ± 183.87	0.13 ± 0.06	6.44 ± 1.70
SD	LP	134.20 ± 55.58	0.11 ± 0.05	5.69 ± 0.79
LD	BB	194.77 ± 42.56	0.36 ± 0.05	8.12 ± 1.03
LD	BP	249.08 ± 30.20	0.32 ± 0.04	4.07 ± 1.56
LD	LP	170.32 ± 26.60	0.38 ± 0.09	10.81 ± 1.51
NT	BB	148.35 ± 47.26	0.85 ± 0.42	12.54 ± 4.72
NT	BP	123.35 ± 41.22	0.23 ± 0.05	6.48 ± 1.01
LOWA	BB	0	118.24	1.67
LOWA	BP	0	71.08	1.47
LOWA	LP	0	191.69	1.62
SD TOTAL	BBWMA	175.07 ± 60.42	0.11 ± 0.02	5.47 ± 0.62
LD TOTAL	BBWMA	204.72 ± 20.53	0.35 ± 0.03	7.67 ± 1.20
NT TOTAL	BBWMA	135.85 ± 28.60	0.54 ± 0.23	9.51 ± 2.55
LOWA TOTAL	BBWMA	0	127.00 ± 35.09	1.59 ± 0.06

Table 12. Comparison of mean Lake Ontario water level (m) and Braddock Bay WMA mean water depth (cm) for spring of 2020 and 2021. Mean Lake Ontario water level was quantified by taking the average of the March, April, and May water levels (NOAA 2024). Mean water depth for Braddock Bay WMA treatments/sites was quantified using the water depth of the submerged aquatic zone. Treatments split by large open-water area (LOWA), small-dredged pond (SD), large-dredged pond (LD), and natural pond (NT). Error bars set to ± 1 SE of the mean water depth.

Treatment	Site	2020	2021
SD	BB	107.78 ± 11.28	98.33 ± 7.52
SD	BP	84.44 ± 1.11	42.78 ± 9.35
SD	LP	80.00 ± 3.85	22.22 ± 1.11
LD	BB	96.67 ± 15.03	58.33 ± 4.19
LD	BP	124.44 ± 8.68	90.00 ± 3.33
LD	LP	94.44 ± 2.94	51.67 ± 6.74
NT	BB	76.67 ± 5.09	60.00 ± 22.42
NT	BP	102.22 ± 10.60	75.00 ± 22.99
LOWA	BP	136.67	80
LOWA	BB	120	86.67
LOWA	LP	93.33	53.33
SD TOTAL	BBWMA	90.74 ± 5.52	54.44 ± 11.89

LD TOTAL	BBWMA	105.19 ± 7.01	66.67 ± 6.41
NT TOTAL	BBWMA	89.44 ± 7.77	67.50 ± 14.75
LOWA TOTAL	BBWMA	116.67 ± 12.62	73.33 ± 10.18
N/A	LAKE ONTARIO	75.29	74.61

Table 13. Candidate models created to test which variables have the most influence in predicting waterfowl cumulative use days/ha, Shannon Weiner diversity of waterfowl, and waterfowl species richness with Braddock Bay Wildlife Management Area. Models 1-14 include environmental variables only, models 15-28 have site added as a variable and year as a factor, and models 29-42 have site and pond type added as variables and year as a factor.

Candidate Model	Variables Included
1	Interspersion (m) + Emergent zone vegetated (%) + Emergent zone standing dead (%)
2	Emergent zone vegetated (%) + Emergent zone standing dead (%)
3	Area (ha) + Shoreline Development Index
4	Area (ha) + Shoreline Development Index + Interspersion (m)
5	Invertebrate Abundance/m^3 + SAV zone vegetated (%) + Average water depth (cm)
6	Invertebrate Abundance/m^3 + Invertebrate order richness + Average water depth (cm)
7	Invertebrate Abundance/m^3 + Invertebrate order richness + SAV zone vegetated (%) +Average water depth (cm)
8	Emergent zone vegetated (%) + SAV zone vegetated (%)
9	SAV zone vegetated (%) + Average SAV zone water depth (cm)
10	Total vegetated (%) + Average water depth (cm) + Vegetation species richness
11	Distance to disturbance (m) + Area (ha)

12	Distance to disturbance (m) + Area (ha) + Interspersion (m)
13	Average water depth (cm) + Area (ha) + Interspersion (m)
14	Average SAV zone water depth + Area (ha) + Interspersion (m)
15	Year (As factor) + Site + Interspersion (m) + Emergent zone vegetated (%) + Emergent zone standing dead (%)
16	Year (As factor) + Site + Emergent zone vegetated (%) + Emergent zone standing dead (%)
17	Year (As factor) + Site + Area (ha) + Shoreline Development Index
18	Year (As factor) + Site + Area (ha) + Shoreline Development Index + Interspersion (m)
19	Year (As factor) + Site + Invertebrate Abundance/m^3 + SAV zone vegetated (%) + Average water depth (cm)
20	Year (As factor) + Site + Invertebrate Abundance/m^3 + Invertebrate order richness + Average water depth (cm)
21	Year (As factor) + Site + Invertebrate Abundance/m^3 + Invertebrate order richness + SAV zone vegetated (%) +Average water depth (cm)
22	Year (As factor) + Site + Emergent zone vegetated (%) + SAV zone vegetated (%)
23	Year (As factor) + Site + SAV zone vegetated (%) + Average SAV zone water depth (cm)
24	Year (As factor) + Site + Total vegetated (%) + Average water depth (cm) + Vegetation species richness
25	Year (As factor) + Site + Distance to disturbance (m) + Area (ha)
26	Year (As factor) + Site + Distance to disturbance (m) + Area (ha) + Interspersion (m)
27	Year (As factor) + Site + Average water depth (cm) + Area (ha) + Interspersion (m)
28	Year (As factor) + Site + Average SAV zone water depth + Area (ha) + Interspersion (m)

29	Pond Type + Year (As factor) + Site + Interspersion (m) + Emergent zone vegetated (%) + Emergent zone standing dead (%)
30	Pond Type + Year (As factor) + Site + Emergent zone vegetated (%) + Emergent zone standing dead (%)
31	Pond Type + Year (As factor) + Site + Area (ha) + Shoreline Development Index
32	Pond Type + Year (As factor) + Site + Area (ha) + Shoreline Development Index + Interspersion (m)
33	Pond Type + Year (As factor) + Site + Invertebrate Abundance/m^3 + SAV zone vegetated (%) + Average water depth (cm)
34	Pond Type + Year (As factor) + Site + Invertebrate Abundance/m^3 + Invertebrate order richness + Average water depth (cm)
35	Pond Type + Year (As factor) + Site + Invertebrate Abundance/m^3 + Invertebrate order richness + SAV zone vegetated (%) +Average water depth (cm)
36	Pond Type + Year (As factor) + Site + Emergent zone vegetated (%) + SAV zone vegetated (%)
37	Pond Type + Year (As factor) + Site + SAV zone vegetated (%) + Average SAV zone water depth (cm)
38	Pond Type + Year (As factor) + Site + Total vegetated (%) + Average water depth (cm) + Vegetation species richness
39	Pond Type + Year (As factor) + Site + Distance to disturbance (m) + Area (ha)
40	Pond Type + Year (As factor) + Site + Distance to disturbance (m) + Area (ha) + Interspersion (m)
41	Pond Type + Year (As factor) + Site + Average water depth (cm) + Area (ha) + Interspersion (m)
42	Pond Type + Year (As factor) + Site + Average SAV zone water depth + Area (ha) + Interspersion (m)

Table 14. Akaike Information Criterion table for selection of a model that has the most influence on predicting waterfowl cumulative use days/ha within Braddock Bay Wildlife Management Area. The top model was the stepwise multiple linear regression with shoreline development index, interspersion, and emergent zone standing dead percentage as explanatory variables.

Model	AIC_C	dAIC	df	Weight
SDI + INT. + EMGSTD	459.4	0.0	5	0.3747
4	461.4	2	5	0.1387
2	462.2	2.8	4	0.0918
6	462.7	3.3	5	0.0731
3	462.9	3.5	4	0.0644
11	463.7	4.3	4	0.0441
9	463.7	4.4	4	0.0424
8	463.7	4.4	4	0.0424
1	464.4	5.0	5	0.0305
7	465.3	5.9	6	0.0196

Table 15. Akaike Information Criterion table for selection of a model that has the most influence on predicting waterfowl species richness within Braddock Bay Wildlife Management Area. The top model was the stepwise multiple linear regression including shoreline development index, site, average water depth, interspersion, and emergent zone standing dead percentage as explanatory variables.

Model	AIC_C	dAIC	df	Weight
SDI + SITE + AVGWD + INT. + EMGSTD	182.6	0.0	9	0.6401
4	184.7	2.1	5	0.2221
18	186.4	3.8	8	0.0945
28	190.5	8	8	0.0120
14	192.3	9.7	5	0.0050
31	192.5	10.0	10	0.0044
27	192.9	10.3	8	0.0037
3	193.1	10.5	4	0.0034
11	193.1	10.5	4	0.0033
13	194.4	11.8	5	0.0018

Table 16. Akaike Information Criterion table for selection of a model that has the most influence on predicting Shannon-Weiner diversity of waterfowl within Braddock Bay Wildlife Management Area. The top model was the stepwise multiple linear regression including plant species richness and year as explanatory variables.

Model	AIC_C	dAIC	df	Weight
Plant SR + Year	43.4	0.0	4	0.5981
2	48.1	4.7	4	0.0569
8	48.9	5.5	4	0.0387
3	49.0	5.5	4	0.0375
11	49.4	5.9	4	0.0311
4	49.5	6.1	5	0.0288
1	50.0	6.6	5	0.0221
9	50.1	6.7	4	0.0214
16	50.9	7.5	7	0.0140
10	51.0	7.6	5	0.0136

Figures

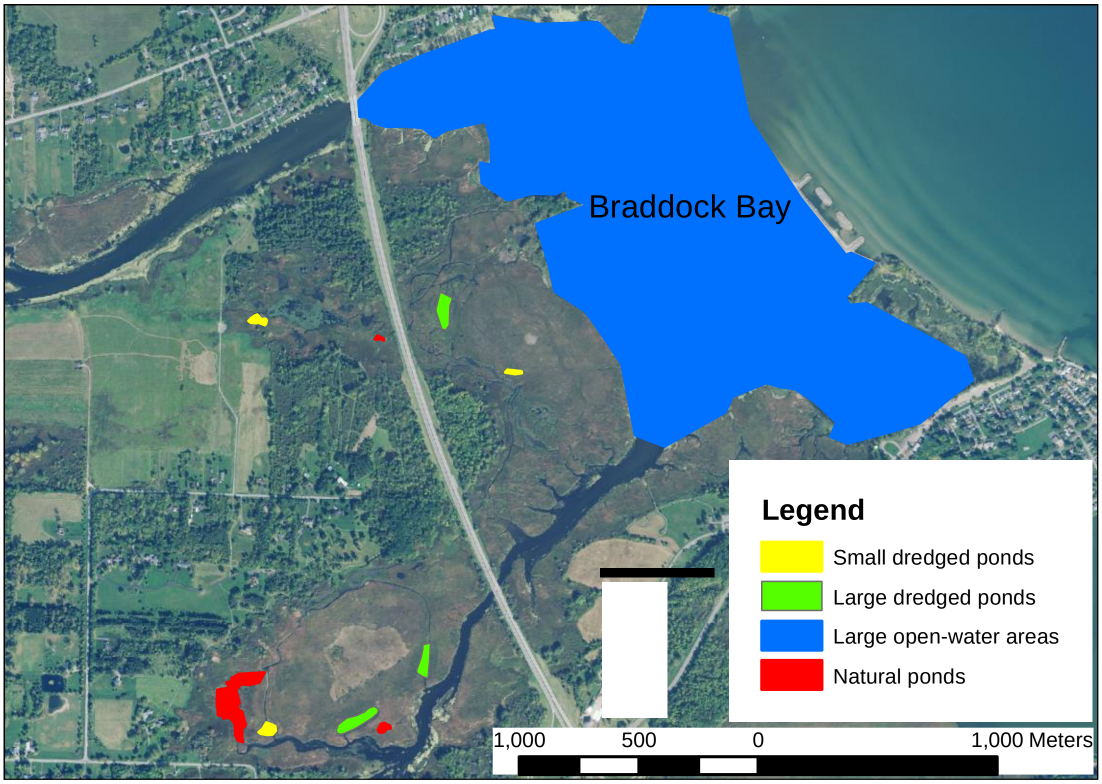


Figure 1. Location of sampling sites for Braddock Bay of the Braddock Bay Wildlife Management Area, Greece, NY.

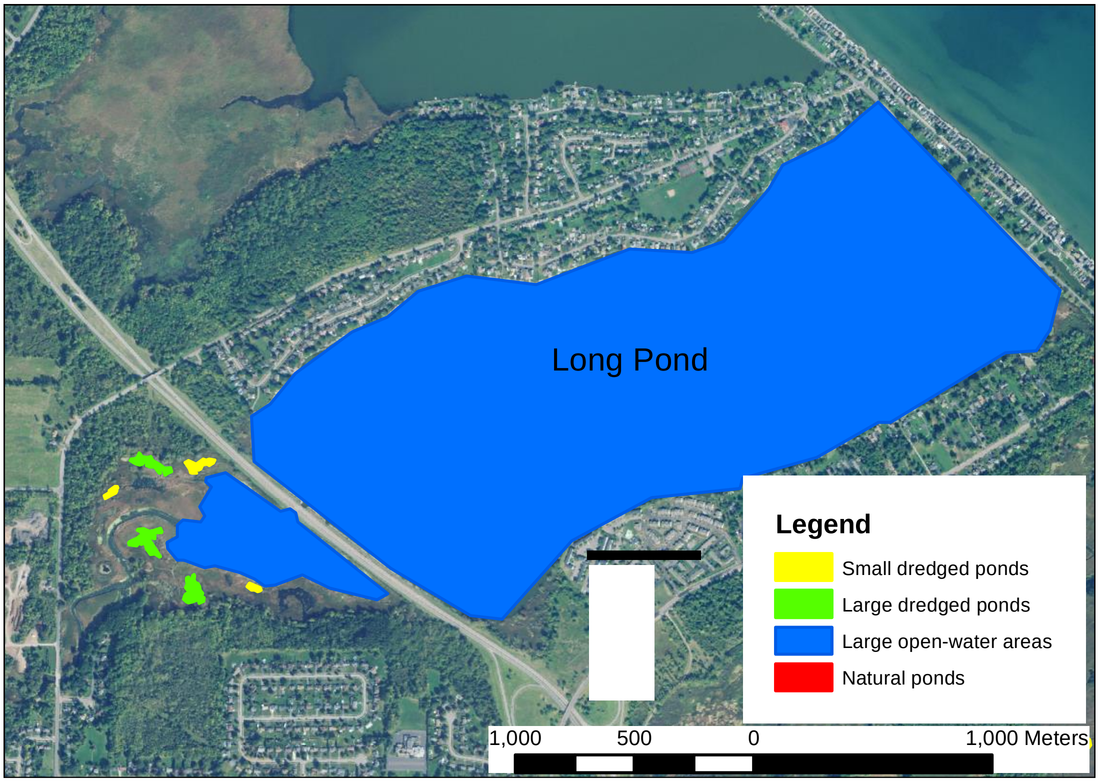


Figure 2. Location of sampling sites for Long Pond of the Braddock Bay Wildlife Management Area, Greece, NY.

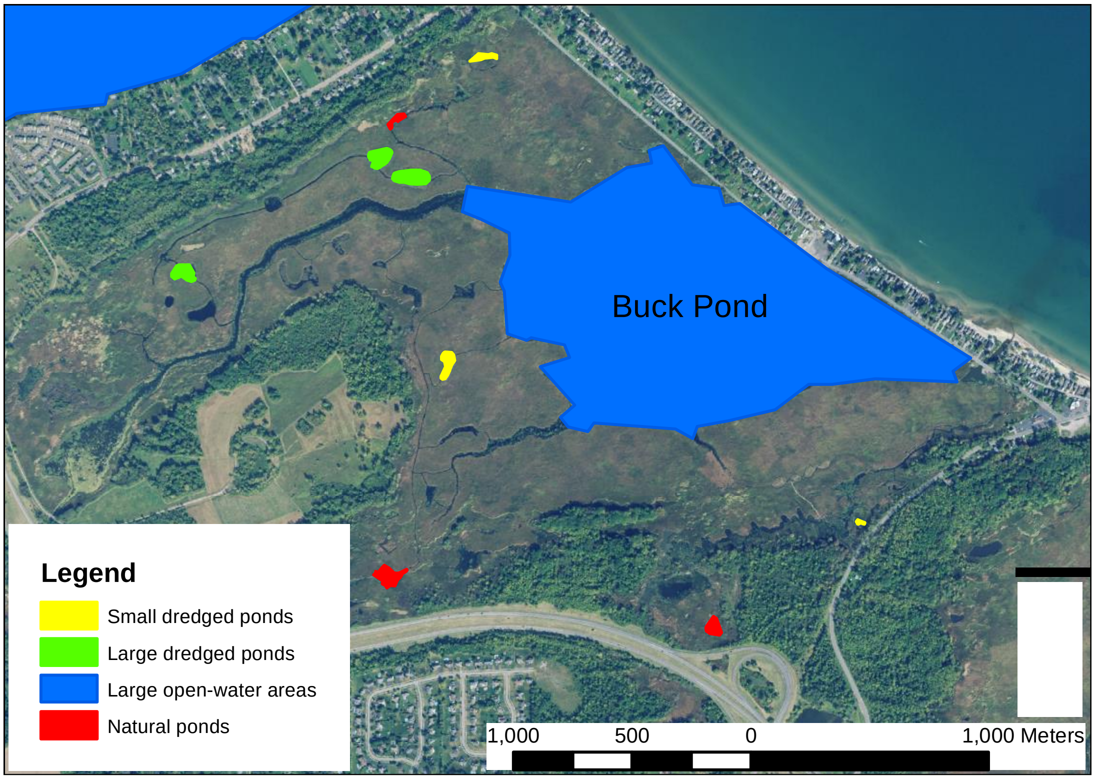


Figure 3. Location of sampling sites for Buck Pond of the Braddock Bay Wildlife Management Area, Greece, NY.

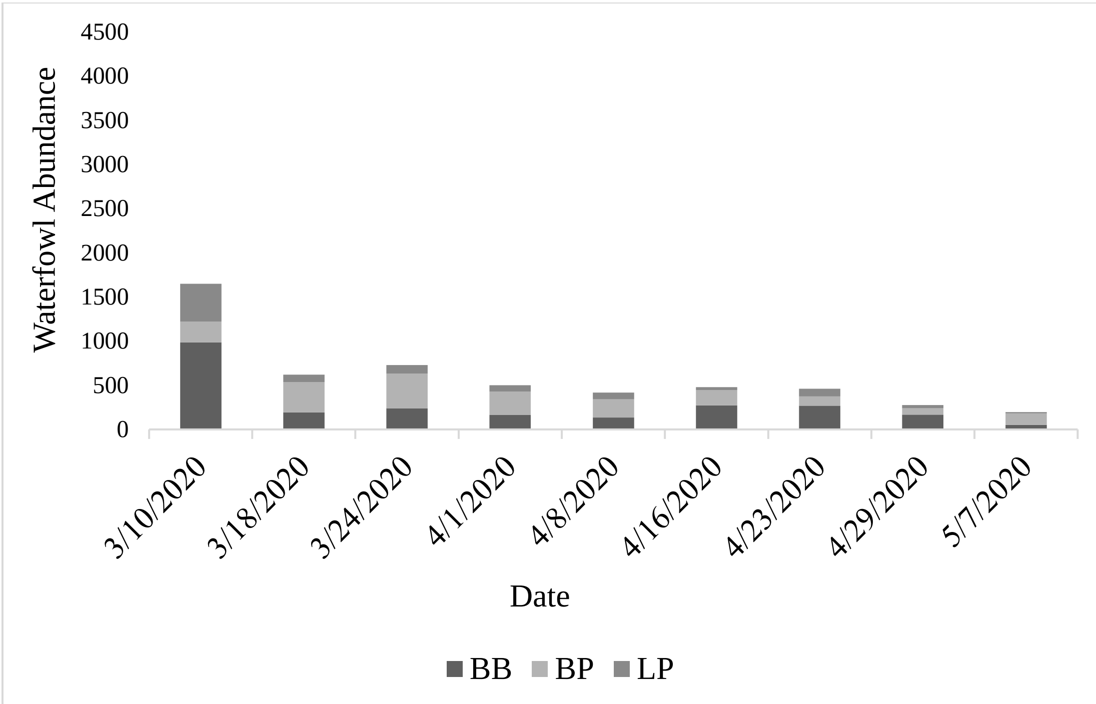


Figure 4. Chronology of total waterfowl use of Braddock Bay WMA for spring of 2020, separated by site.

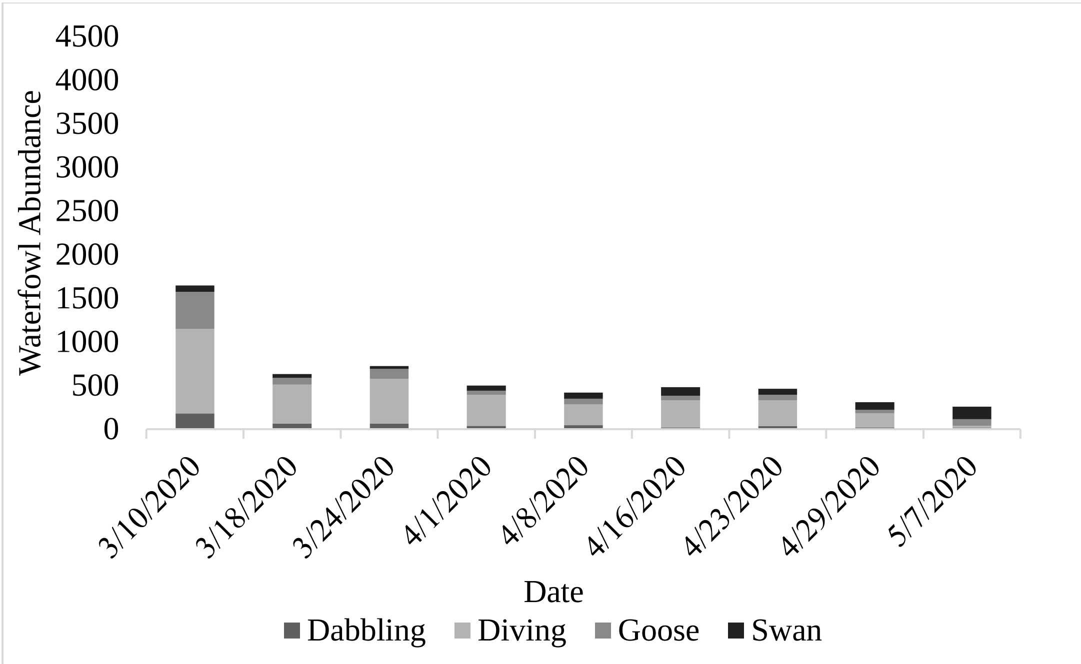


Figure 5. Composition of Braddock Bay WMA waterfowl for spring of 2020 for each point count date. Swan composition was entirely made up of Mute Swans (*Cygnus olor*) except for 3/10/2021 where 1.17% (1/85) of recorded swans were Tundra Swans (*Cygnus columbianus*).

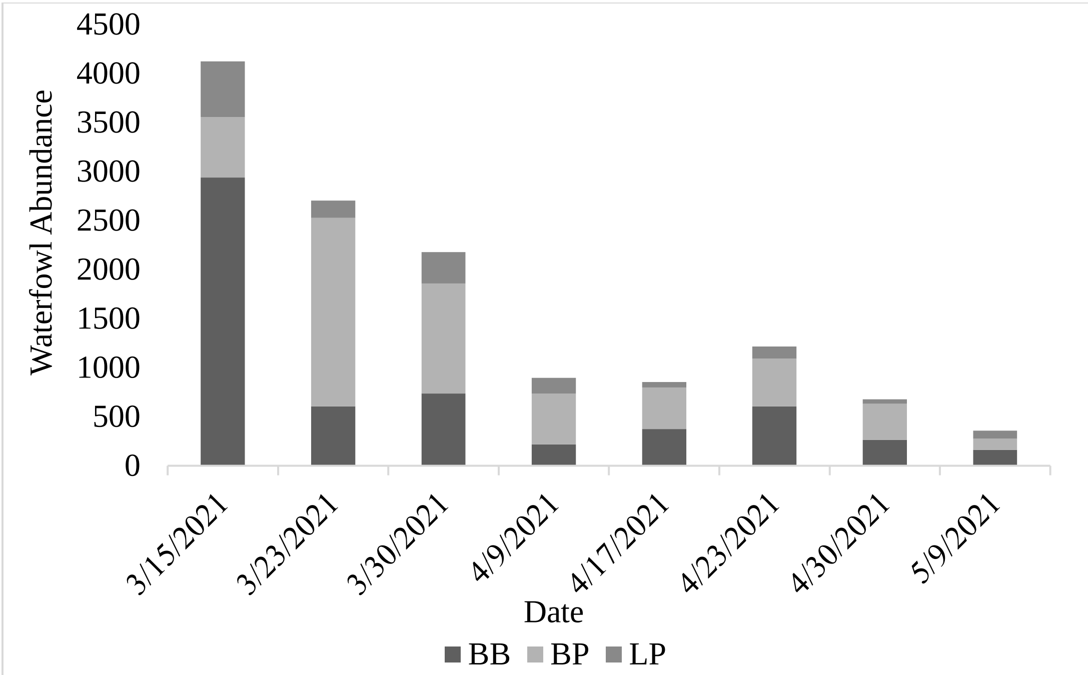


Figure 6. Chronology of total waterfowl use of Braddock Bay WMA for spring of 2021, separated by site.

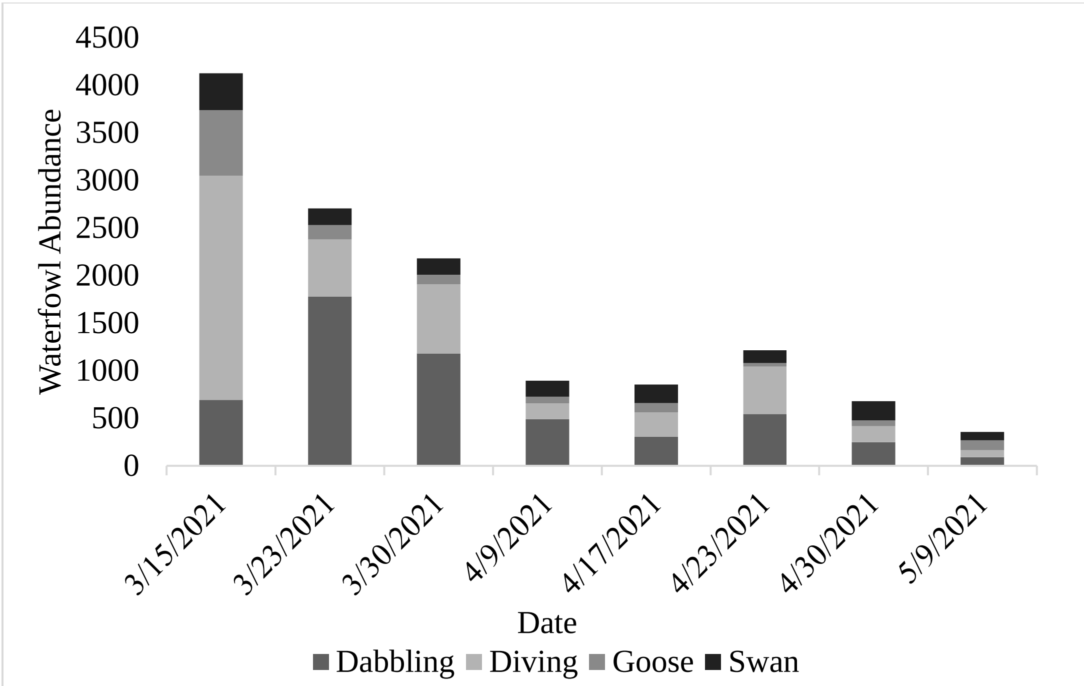


Figure 7. Composition of Braddock Bay WMA waterfowl for spring of 2021 for each point count date. Swan composition was entirely made up of Mute Swans except for 3/15/2021 where 40.71% (160/393) of recorded swans were Tundra Swans.

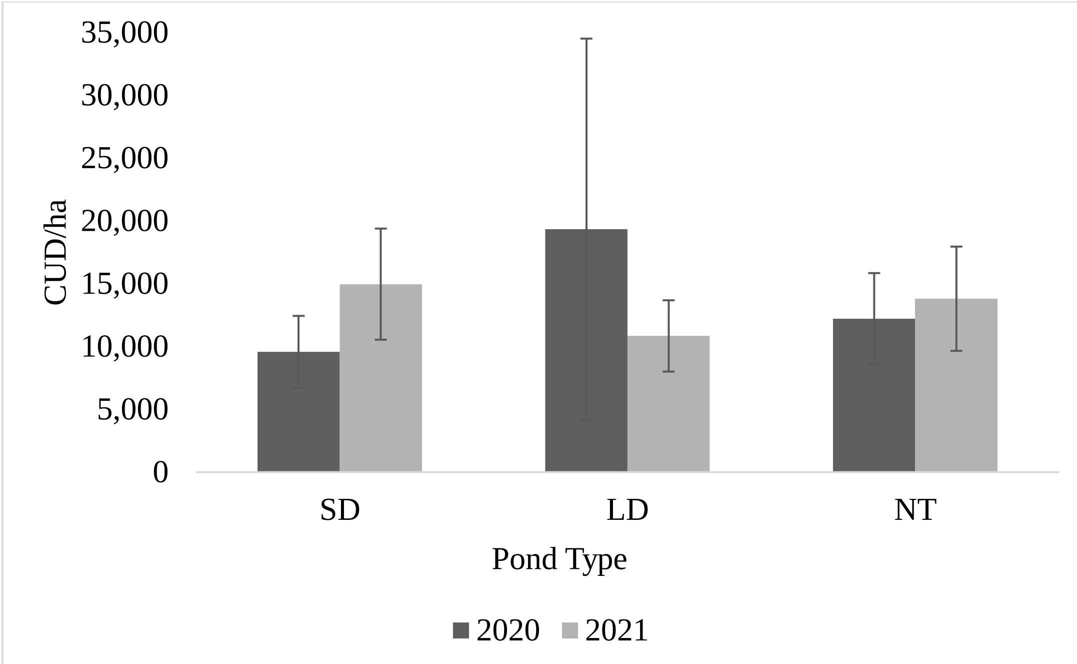


Figure 8. Mean waterfowl cumulative use days/hectare (CUD/ha). Values are split by year and pond type. Pond type is separated into SD for small-dredged pond, LD for large-dredged pond, and NT for natural pond. Mean values for small-dredged and large-dredged pond types are for total WMA. Mean values for natural pond type are for Braddock Bay and Buck Pond only. Error bars are set to ± 1 SE of the mean.

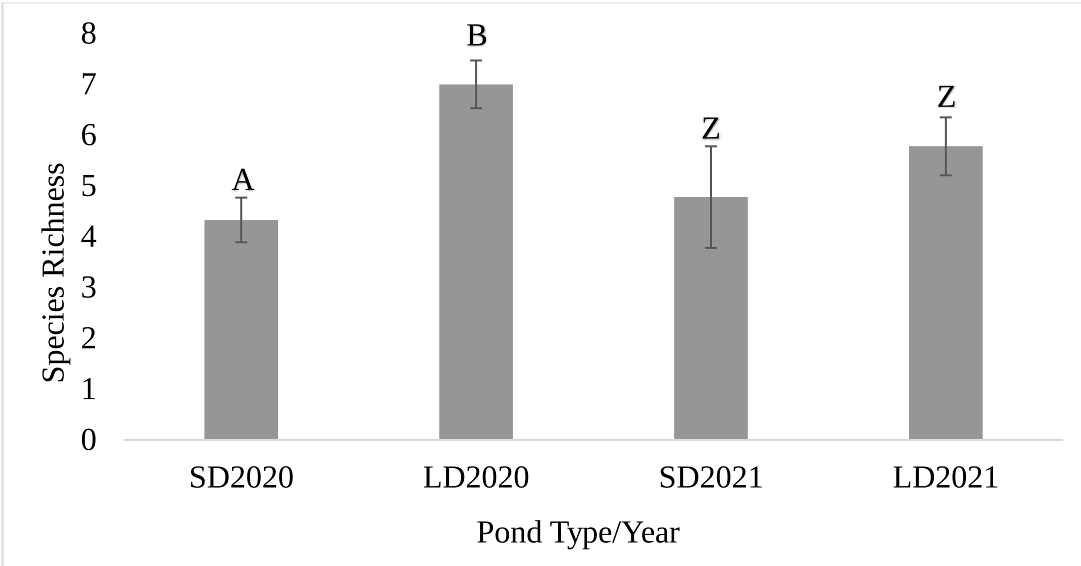


Figure 9. Mean waterfowl species richness for total WMA small-dredged (SD) and large-dredged (LD) pond types. Error bars set to ± 1 SE of the mean waterfowl species richness. Within year waterfowl species richness by pond type with similar letters are not significantly ($P \leq 0.01$).

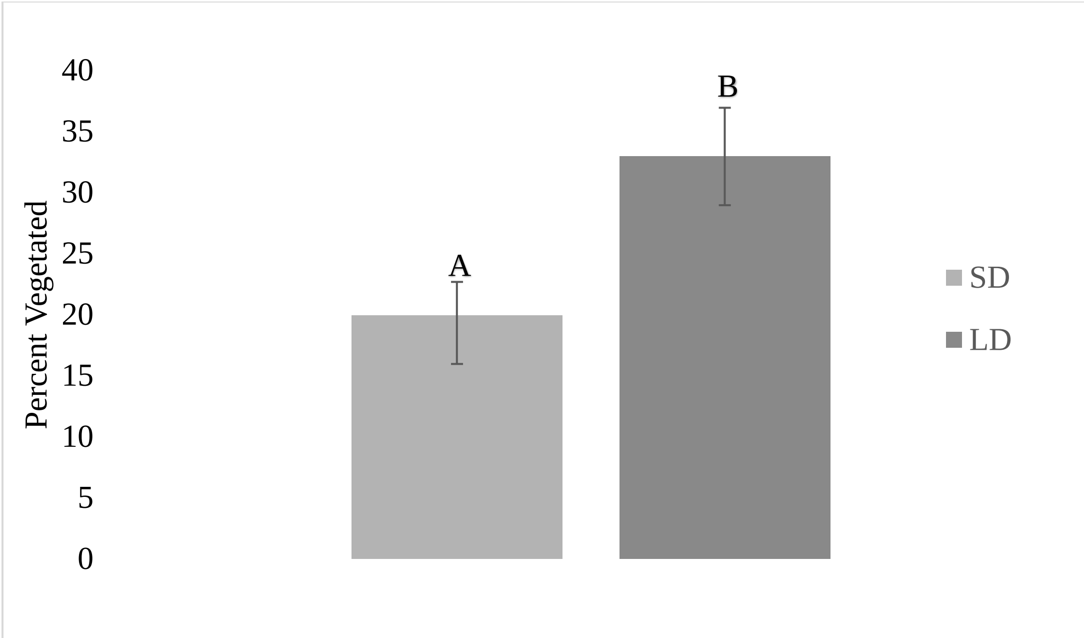


Figure 10. Mean percent vegetated for total WMA large-dredged (LD) and small-dredged (SD) pond types in 2021. Error bars set to ± 1 SE of the mean percent vegetated. Significance ($P \leq 0.01$) shown by a difference in letter between the mean percent vegetated of the two pond types.

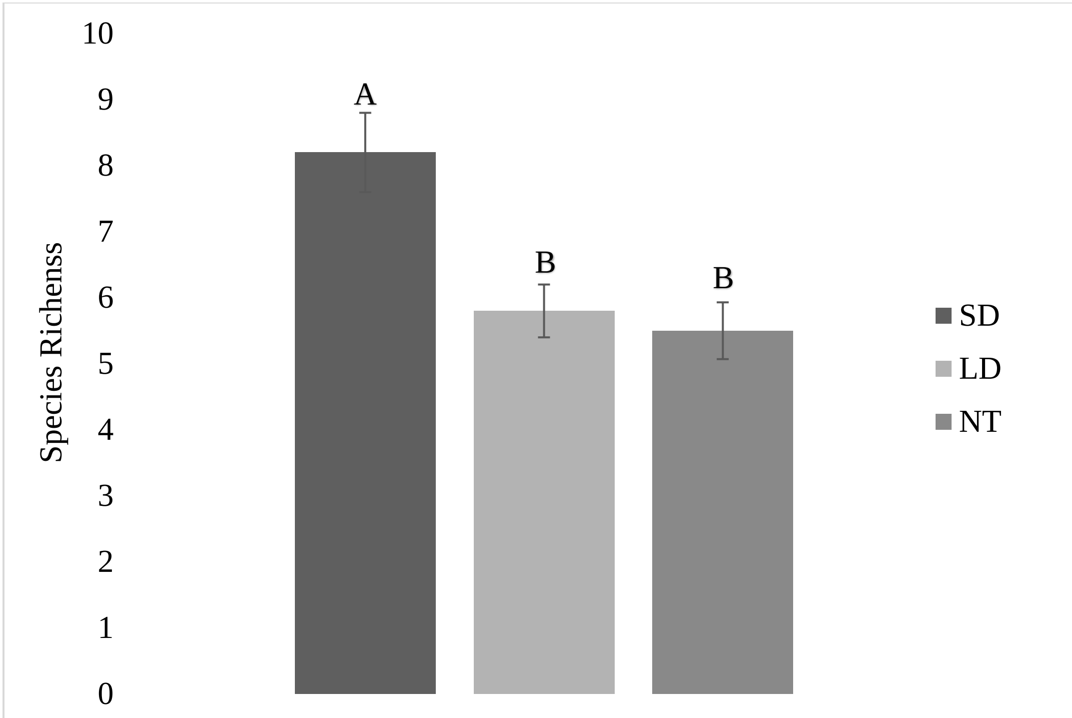


Figure 11. Mean vegetation species richness for Braddock Bay, and Buck Pond, large-dredged (LD), small-dredged (SD), and natural (NT) pond types in 2021. Error bars set to ± 1 SE. Significance shown by a difference in letter between the mean vegetation species richness of the three pond types.

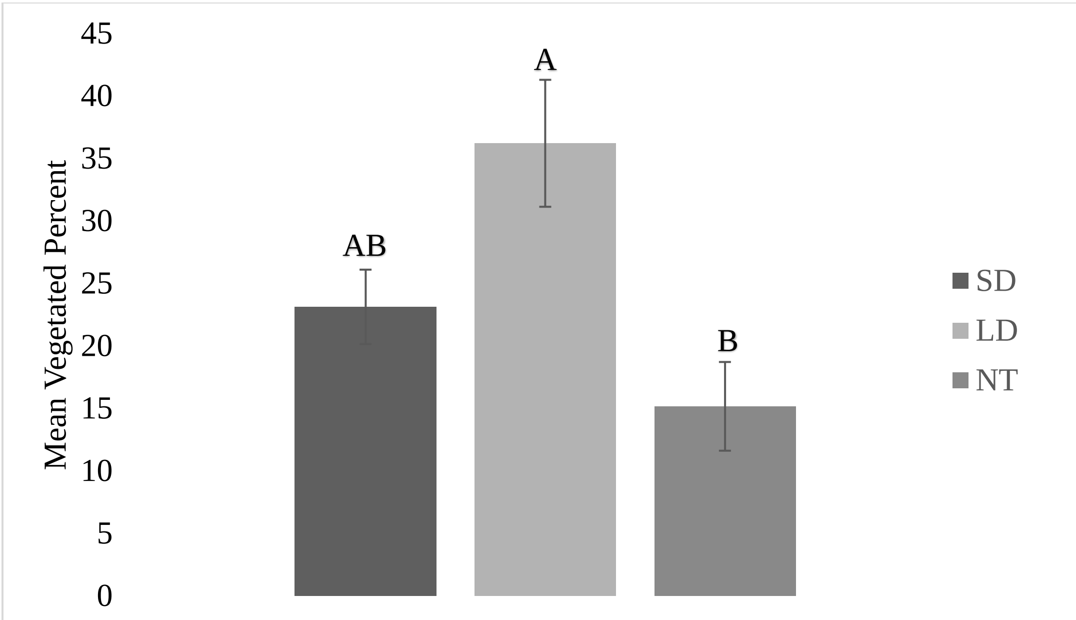


Figure 12. Mean vegetated percent for Braddock Bay, and Buck Pond, large-dredged (LD), small-dredged (SD), and natural (NT) pond types in 2021. Error bars set to ± 1 SE. Significance shown by a difference in letter between the mean vegetated percent of the three pond types.

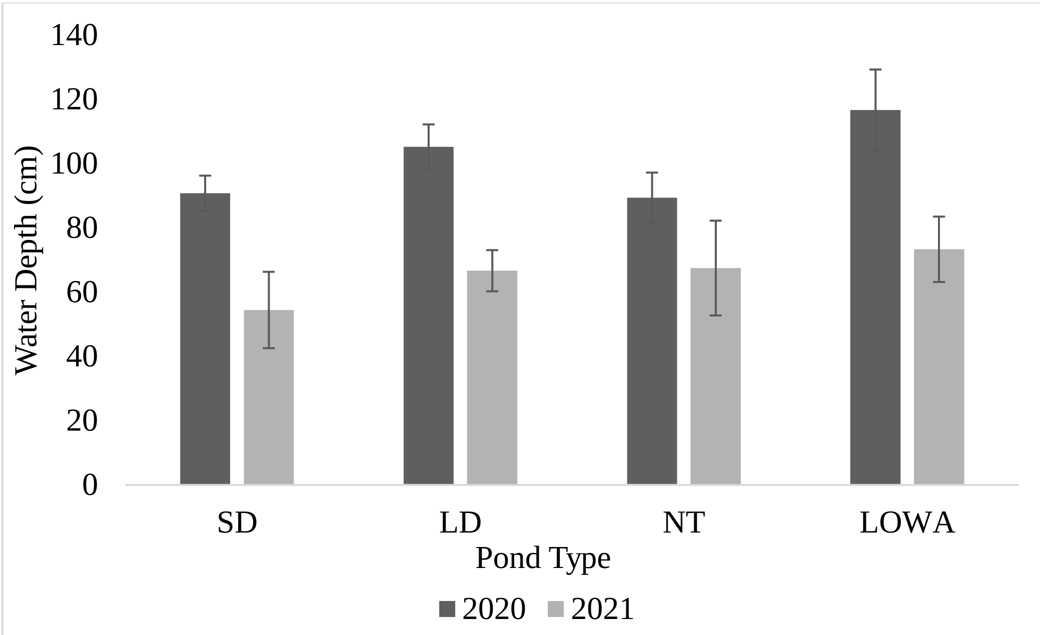


Figure 13. Comparison of Braddock Bay WMA mean water depth in spring of 2020 and 2021, separated by small-dredged ponds (SD), large-dredged ponds (LD), natural ponds (NT), and large open-water areas (LOWA). Average water depth for each habitat type was quantified using the water depth of the submerged aquatic zone. Error bars set to ± 1 SE of the mean water depth for each pond type.

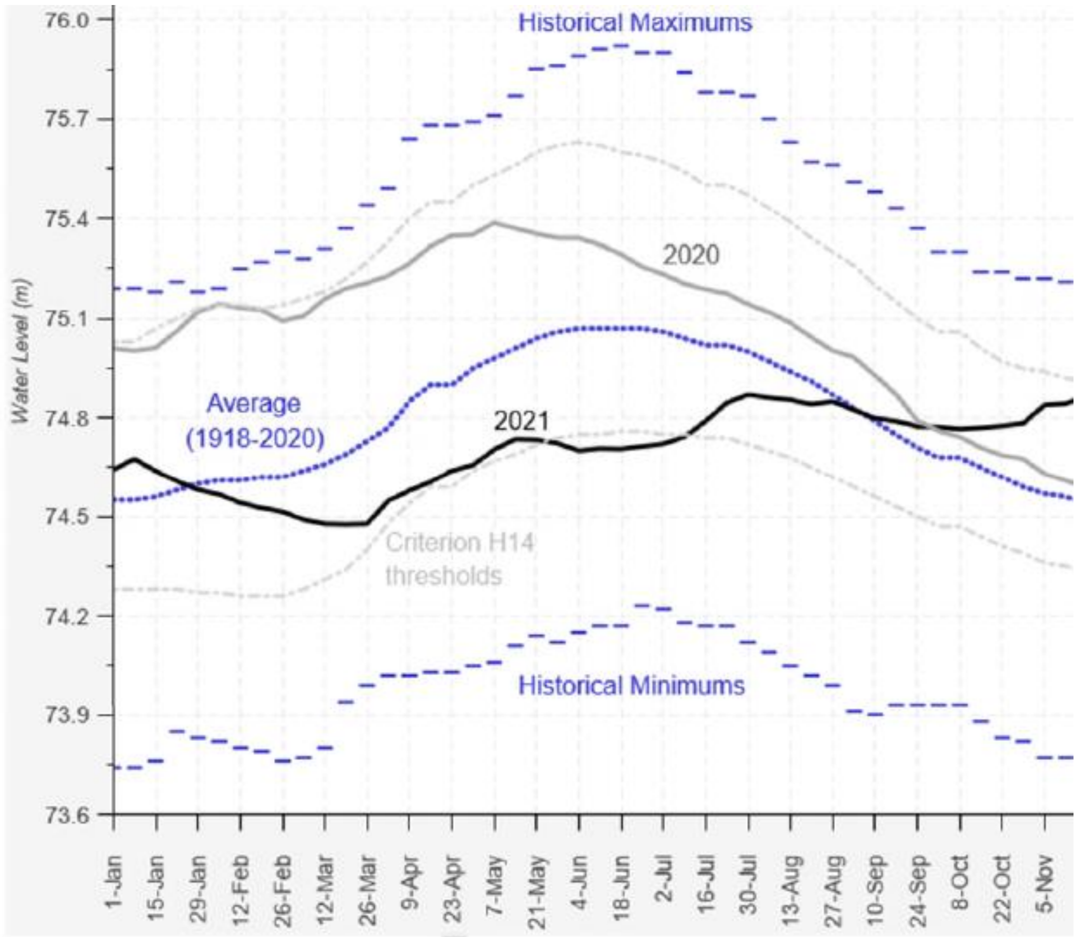


Figure 14. Lake Ontario water level showing the long-term water level average, historical peaks, 2020 water level, and 2021 water level (International Joint Commission 2021).

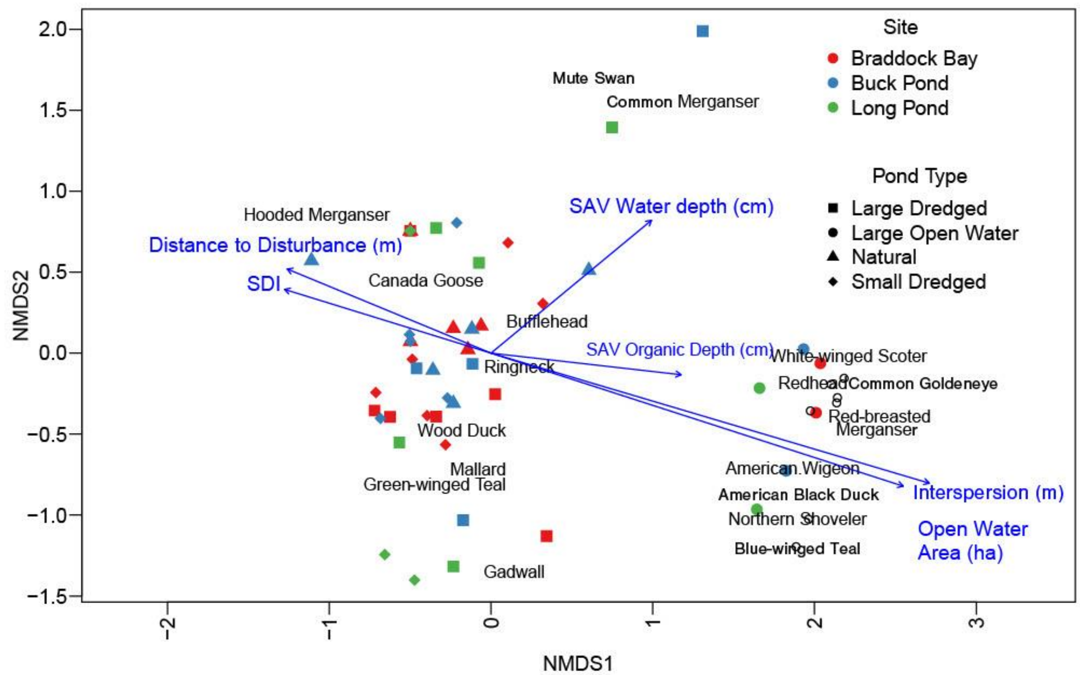


Figure 15. NMDS ordination showing grouping of waterfowl species with pond types/sites of Braddock Bay WMA. Overlain vectors are environmental variables that are significant to the ordination. Groupings of waterfowl species and pond types/sites that are close together are more similar than those that are far apart.

Appendices

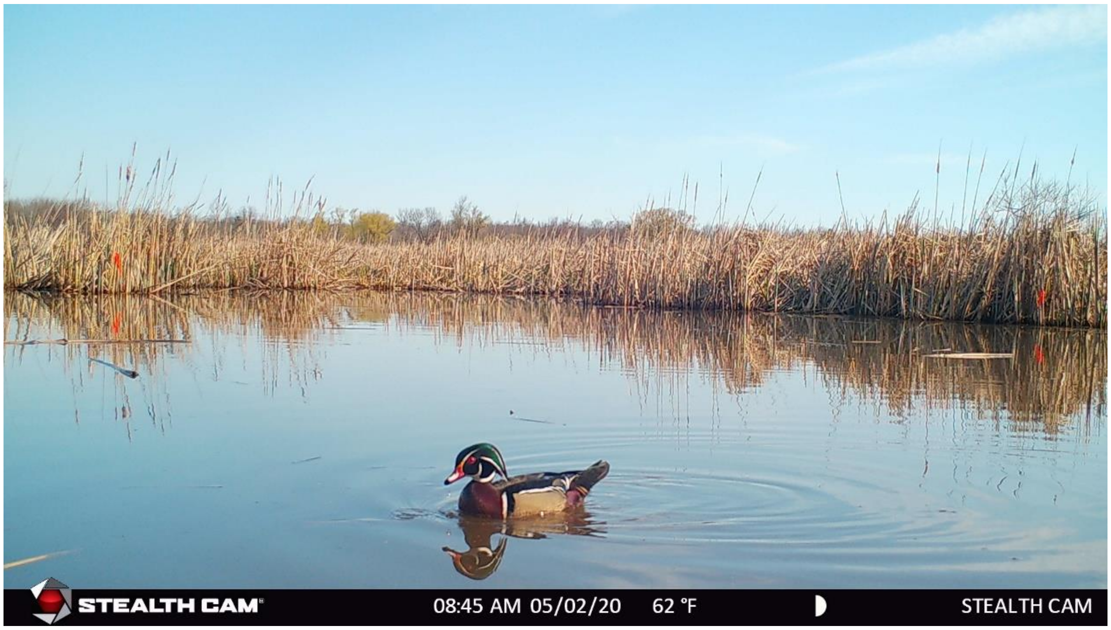
Appendix 1. Point count survey of waterfowl within the Braddock Bay large open-water area of Braddock Bay WMA.

Appendix 2. Trail camera setup on a pond of Braddock Bay WMA. Trail cameras were attached to a a PVC pipe that was pounded into the sediment at the vegetation to water interface.

Appendix 3. Example of floating markers used to delineate a known area in front of trail cameras, set within natural and dredged ponds of Braddock Bay WMA.

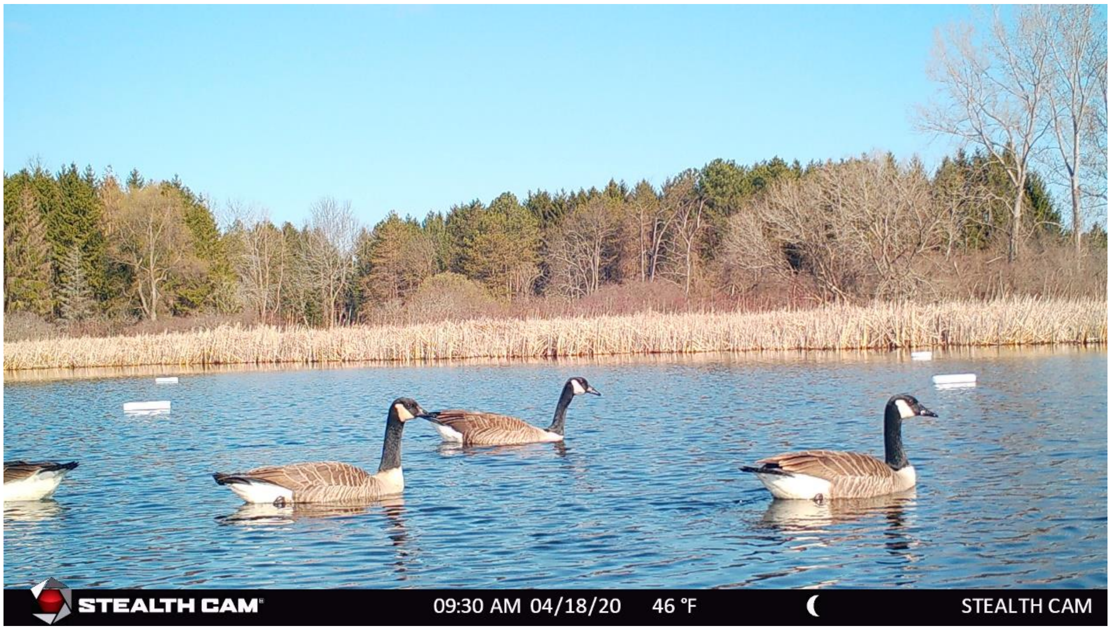
Appendix 4. Field sampling of emergent vegetation at Braddock Bay WMA.

Appendix 5. Field sampling of aquatic macroinvertebrates in Braddock Bay WMA, using the sweep net method.



Appendix 6. Trail camera image of a Wood duck recorded during waterfowl surveying of Braddock Bay WMA.

Appendix 7. Trail camera image of a Mallard recorded during waterfowl surveying of Braddock Bay WMA.



Appendix 8. Trail camera image of a Canada Geese recorded during waterfowl surveying of Braddock Bay WMA.

Appendix 9. Eurasian Wigeon pictured in the center (*Anas penelope*) between a drake American Wigeon (*Anas americana*) (Left) and hen American Wigeon (Right). These species were recorded during a point count survey of the large open-water area of Long Pond, of Braddock Bay WMA.

Appendix 10. Spring presence of vegetation species within Braddock Bay WMA, separated by pond type. Presence in spring of 2020 or spring of 2021 marked by “20” or “21”.

Species	LOWA	LDP	NP	SDP
Algae (*Nitella sp.*)				20, 21
Canadian waterweed (*Elodea canadensis*)	20, 21	20	21	20, 21
Common Bladderwort (*Utricularia vulgaris*)		21	21	21
Coontail (*Ceratophyllum demersum*)	20, 21	20, 21	20, 21	20, 21
Curly-leaf pondweed (*Potamogeton crispus*)	21			
Eelgrass (*Vallisneria americana*)		20		20, 21
Eurasian Watermilfoil (*Myriophyllum spicatum*)	20	20		
European Frogbit (*Hydrocharis morsus-rane*)		21		21
Flatstem pondweed (*Potaogeton zosteriformis*)	20	21		
Floating crystalwort (*Riccia fluitans*)		20	20	20, 21
Fringed heartwort (*Ricciocarpus natans*)				21
Greater Duckweed (*Spirodela polyrhiza*)	20	20	20	20
Hybrid Cattail (*Typha x glauca*)	20, 21	20, 21	20, 21	20, 21
Lake Sedge (*Carex lacustris*)	20	20, 21		20, 21
Lesser Duckweed *(Lemna minor*)	20, 21	20, 21	20, 21	20, 21
Pondweed (*Potamogeton sp.*)	20, 21	20, 21	20, 21	20, 21
Star Duckweed (*Lemna trisulca*)	20, 21	21	20, 21	20, 21
Unknown Algae	20, 21	20, 21	20, 21	20, 21
Unknown Moss		20, 21		20, 21
White water-lily (*Nymphaea odorata*)	21	21	20, 21	21
Yellow water-lily (*Nuphar lutea*)			21	21

Appendix 11. Spring presence of invertebrate orders within Braddock Bay WMA, separated by pond type. Presence in spring of 2020 or spring of 2021 marked by “20” or “21”.

Order	LOWA	LDP	NP	SDP
Amphipoda	20,21	20, 21	20, 21	20, 21
Bivalvia	20	20	20	20
Calanoida				20
Cladocera	21	20, 21	20, 21	20, 21
Cyclopoida	20, 21	20, 21	20, 21	20, 21
Diptera	20, 21	20, 21	20, 21	20, 21
Ephemeroptera	20, 21	20, 21	20, 21	20, 21
Gastropoda	20, 21	20, 21	20, 21	20, 21
Hemiptera		20		
Hirundinea		21		
Isopoda	21	20		
Odonata	20	20, 21	20	20, 21
Plecoptera		20		
Trichoptera	21	21	21	21
Trombidiformes	21	21	20, 21	20, 21
Tubificida	20, 21	20, 21	20, 21	20, 21