THE ROLE OF SMALL MITIGATION WETLANDS IN SUPPORTING FALL MIGRATING WATERFOWL

by

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# Abstract

While wetland mitigation aims to offset disturbances to wetland habitats, it often falls short in matching the ecological function, size, and quality of the originally disturbed area. Further, the criteria for defining wetland mitigation success are a matter of debate, and current practices often fail to evaluate success based on management goals. When evaluating mitigation success, it can be useful to establish criteria based on specific wetland functions, like habitat that supports wildlife. We used a stratified sampling design to estimate forage availability and energetic carrying capacity (ECC) for waterfowl in mitigation wetlands along a quality gradient in the Lake Champlain Valley of New York state. Vegetation and invertebrate food sources were sampled to estimate forage biomass (kg/ha) and ECC in terms of duck-energy days (DED/ha). Our results revealed significant variation in forage availability and DEDs across the quality gradient, with a more than 20-fold increase in DEDs from the lowest to highest quality sites. Submergent marsh vegetation provided the highest unweighted mean DEDs in most study areas. While invertebrate food sources were relatively insignificant overall, they played a more substantial role in poor-quality wetlands. The study highlights the importance of conserving wetland quality, not just acreage, in mitigation efforts and demonstrates that small mitigation wetlands can contribute significantly to waterfowl habitat if properly managed. These findings emphasize the need for more rigorous standards in wetland mitigation projects and suggest using alternative means, like ECC for waterfowl, for defining mitigation success based on management goals. Further, we investigated the relationship between ECC for waterfowl and waterfowl use of small mitigation wetlands. To do this, we conducted game camera surveys at three of the five study areas—one low quality, one medium quality, and one high quality—based on a rapid qualitative assessment and our ECC results. Two daily surveys were conducted using the game cameras from September 1$^{st}$ – December 17$^{th}$, 2023. We found no

relationship between site quality and number of waterfowl observed, suggesting other drivers of waterfowl habitat selection like abundance of emergent vegetation, proximity to open water, and disturbance regime. This study demonstrates the need for managing mitigation wetlands wholistically for a variety of species, both flora and fauna, with a focus on habitat heterogeneity and connectivity. Additionally, this study highlights the advantages and disadvantages of low-cost ecological sampling methods like game cameras and autonomous recording units. Further research is required to better understand waterfowl habitat selection drivers and the significance of small mitigation wetlands across the landscape for more than just waterfowl.

# Chapter 1. Energetic Carrying Capacity for Waterfowl as a Metric for Wetland Mitigation Success

## 1.1 Introduction

Despite their ecological significance, anthropogenic disturbances are responsible for an estimated 54-57% loss of natural wetlands since the 1700s (Davidson 2014). To limit wetland losses in the United States, wetland mitigation requirements were codified into law under Section 404 of the Clean Water Act (1972) and the Mitigation Rule (2008). These regulations require that impacts to wetlands must be offset by the replacement or enhancement of the resource elsewhere (Zedler 1996). Wetland mitigation can more broadly be defined as the enhancement, restoration, or creation of wetland habitats that equal or exceed the performance of the disturbed area (EPA 1995; Zedler 1996). While the goal of wetland mitigation is to conserve wetland habitat, previous research has revealed that mitigation wetlands often fall short in matching the function, size, and quality of the originally disturbed site (Goldberg & Reiss 2016; Burgin 2010; La Peyre et al. 2001). When such sub-optimal replacements are repeated across the landscape, wetland-dependent species may lose vital habitat.

The criteria for defining wetland mitigation success remain a matter of debate in the field of restoration ecology (Zedler 2007; Kentula 2000). Currently, compliance standards set by the U.S. Army Corps of Engineers (USACE) vary based on individual site goals but typically focus on the plant communities (Matthews & Endress 2008; Van Den Bosch & Matthews 2017; U.S. Army Corps of Engineers 2004). With ambiguity about performance standards and building evidence about the shortcomings of current mitigation practices (Reiss et al. 2009; Hill et al. 2013; Brinson & Rheinhardt 1996), recent research has explored alternative criteria for wetland mitigation performance with a focus on specific wetland functions, *e.g.*, supporting habitat for wildlife (Oja et al. 2021).

When mitigation wetlands are properly designed and managed, they can provide important foraging habitat for waterfowl that contributes to the greater landscape of available resources (Balcombe et al. 2005). The primary driver of habitat selection by waterfowl is forage availability (Osborn et al. 2017; Jorde et al. 1989). Waterfowl rely on wetland vegetation and invertebrate food sources to acquire energy and protein for migratory and breeding nutrient requirements (Davis et al. 2014). Previously, research efforts emphasized the forage availability during the spring migration and breeding season (Stafford et al. 2014; Greer et al. 2007; Straub et al. 2012; Weller & Batt 1988). It has since been revealed that non-breeding forage availability is linked to subsequent recruitment. Osnas et al. (2016) found that increased forage availability during the fall allows birds to increase their fitness and leave wintering grounds earlier, leading to earlier breeding and greater nesting attempts resulting in increases in reproductive success.

Currently, waterfowl habitat conservation efforts are informed by bioenergetics models that use forage biomass (kg/ha) to determine the energetic carrying capacity (ECC) of a given habitat for waterfowl (duck-energy days; DEDs) (Brasher et al. 2007; Straub 2008; Vonbank & Hagy 2016; Fino et al. 2017). Joint Ventures and other waterfowl conservation groups use ECC models to set goals and focus habitat management activities (Upper Mississippi Great Lakes Joint Venture 2007). Current research estimates waterfowl forage availability in heavily trafficked migratory corridors like the Mississippi Alluvial Valley and the upper Illinois River (Straub 2008; Vonbank & Hagy 2016). However, similar research in the Atlantic Flyway is primarily limited to New Jersey and Delaware with no available data for the Lake Champlain Valley (Fino et al. 2017; Ringelman et al. 2018; Cramer 2009). Additionally, no current research explores how ECC for waterfowl differs in mitigation wetlands of varying quality.

We estimated energetic carrying capacity for fall migrating waterfowl in small mitigation wetlands in northern New York state (Figure 1.1). Our objectives were to 1) estimate plant and invertebrate forage biomass in moist-soil (MS), emergent marsh (EM), submergent marsh (SM), scrub/shrub (SS), and *Phragmites* marsh (PM) wetland cover types in four mitigation wetlands relative to a reference site, and 2) use the forage biomass estimates to examine how ECC for waterfowl differs among mitigation wetlands of varying quality.

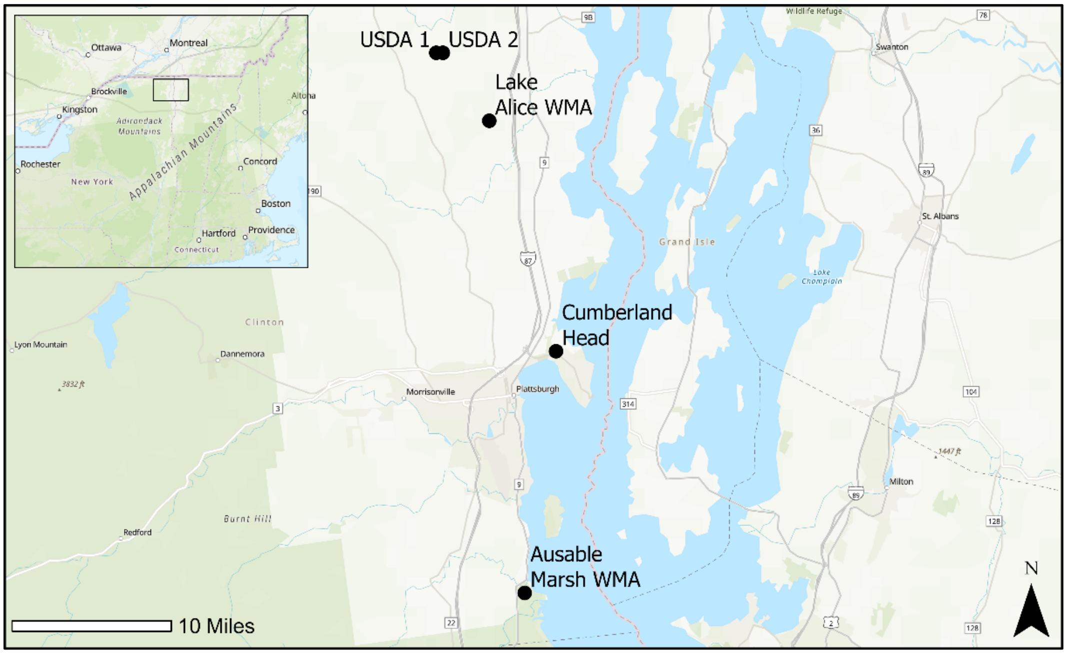


Figure 1.1 Locations of mitigation wetland study areas in northern New York, USA, from which fall energetic carrying capacity for waterfowl was determined, 2023.

### 1.2 Study Area

We defined mitigation wetlands as restored, created, or enhanced wetlands designed to offset disturbances to wetland habitat elsewhere. We selected four mitigation wetlands based on these criteria along a quality gradient (Figure 1.2). The quality of each site was assigned using a rapid assessment of the diversity of the plant community, presence of invasive species, and hydrology using methods similar to Miller and Gunsalus (1997). For our fifth site serving as our positive control, we selected Ausable Marsh, a wildlife management area operated by the New York State Department of Environmental Conservation with enhancements specifically to support waterfowl.

All study areas were located in the Lake Champlain Valley in northern New York. The Lake Champlain Basin comprises over 300,000 documented wetlands (Lake Champlain Basin Program 2013). Many of these wetlands are important for waterfowl production, providing nesting sites for wood ducks (*Aix sponsa*), American black ducks (*Anas rubripes*), and mallards (*Anas platyrhynchos*) (Binhammer & Vermont Nature Conservancy 1994). Additionally, Lake Champlain serves as a significant migratory corridor connecting waterfowl from the Gulf of St. Lawrence to the Long Island Sound and the Atlantic Coast of New Jersey (Bellrose 1968).

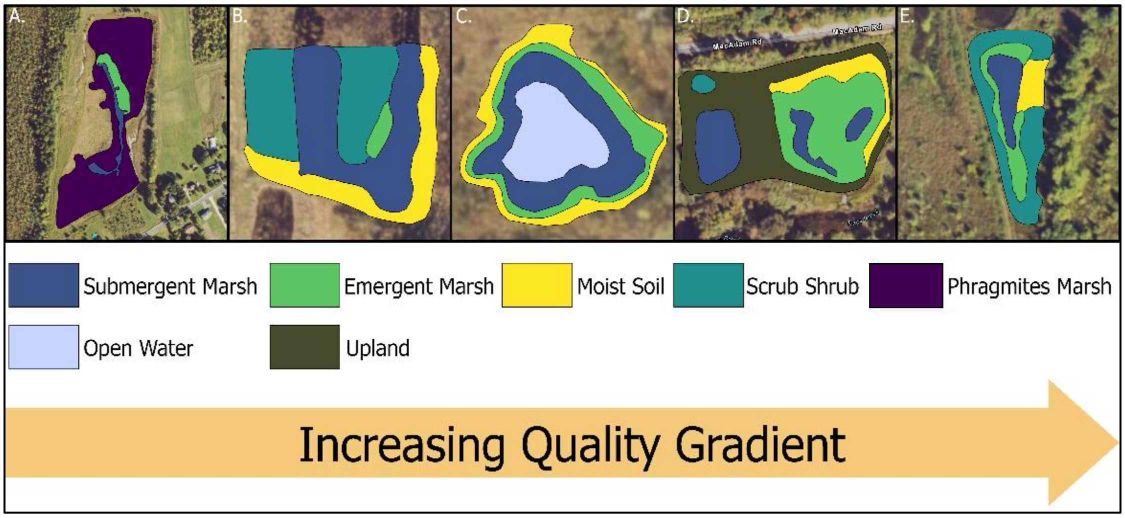


Figure 1.2 Five mitigation wetland study areas with their corresponding cover types along a quality gradient increasing from left to right. A) Cumberland Head, B) USDA 2, C) USDA 1, D) Lake Alice WMA, E) Ausable Marsh WMA.

All wetlands in our study were small wetlands that varied in size from one acre to eleven acres, but all were under the 12.4-acre jurisdictional threshold that protects wetlands in New York State (Kusler 2001). Each of the mitigation projects were completed under the authority of the New York State Department of Environmental Conservation (NYSDEC), United States Department of Agriculture (USDA), or local government's Department of Transportation (DOT). All of the mitigation wetlands, Cumberland Head, USDA 1, USDA 2, and Lake Alice WMA are created wetlands located within seven miles of Lake Champlain. The fifth site serving as the positive control, Ausable Marsh WMA, is a lake-adjacent wetland that is hydrologically connected to Lake Champlain (Diggory et al. 2008). This wetland has undergone enhancements specifically designed to improve habitat quality for waterfowl including geomorphic alterations, water control structures, and the planting of waterfowl foods like wild rice (*Zizania palustris*). Additional site information is available in Appendix A.

## 1.3 Methods

### *1.3.1 Sampling Procedure*

We used a stratified sampling design to estimate the ECC for each wetland site in 2023. We used National Wetlands Inventory (NWI) datasets and satellite imagery in ArcGIS Pro 3.1.1 to delineate wetland cover types and ground-truthed our designations during site visits in April-May 2023. We delineated vegetation zones into the following cover types: moist-soil, characterized by sedges, grasses, rushes, and forbs (*e.g.*, *Carex* spp., *Scirpus* spp, *Schoenoplectus* spp.); emergent marsh, characterized by emergent broad and narrow leaved herbs, grass-like plants, and some floating leaved herbs (*e.g.*, *Sagittaria* spp., *Typha* spp., *Lemna* spp.); submergent marsh, characterized by submerged and floating aquatic vegetation (*e.g. Potamogeton* spp., *Nuphar* spp.); scrub/shrub, characterized by woody shrubs (*e.g.*, *Cephalanthus occidentalis*); *Phragmites* marsh, characterized by the dominance of the invasive common reed (*Phragmites australis*); and unvegetated open water (Brasher et al. 2007; Cohen et al. 2020). We used handheld Global Positioning Systems (Garmin GPSMAP 62s) to create polygons of cover type boundaries for calculating area and drawing transects.

For each cover type within each site, we collected seeds, aquatic vegetation, below-ground tubers, and invertebrates known to be consumed by at least one waterfowl species (Martin & Uhler 1939; Brasher et al. 2007; Fino et al. 2017; Straub et al. 2012; Vonbank & Hagy 2016). We assumed that all waterfowl food items found within the wetland were available to waterfowl (Straub et al. 2012). We sampled from August 15$^{th}$ to September 15$^{th}$ 2023, to coincide with maximum vegetation maturity prior to significant consumption and seed dispersal (Brasher et al. 2007). Each cover type was sampled along a single representative transect where transect length varied among cover types. Sampling occurred randomly along each transect using a random number generator for a total of five replicates per cover type per site. Replicate quadrat size was reduced from 1 m$^2$ to 0.25 m$^2$ during sampling to improve sampling feasibility with personnel limitations. Vegetation food items were sampled by identifying each plant species present within the quadrat and collecting any seed heads, below-ground tubers, or submerged aquatic vegetation known to be consumed by waterfowl (Table 1.1).

Table 1.1 True metabolizable energy (TME; kcal/g dry mass) estimates of vegetation and macroinvertebrate food sources from published literature used in this study to estimate energetic carrying capacity (ECC) of mitigation wetlands along a quality gradient in northern New York state, USA, in Fall 2023.

| Scientific Taxa (vegetation [V], animal [A], or other [O]) | Common Name | TME (kcal/g) | Justification and source of TME |
|---|---|---|---|
| Algae (O) | | 2.5 | Kaminski et al. (2003)* |
| Arachnida (A) | arachnids | 0.58 | Straub (2008) |
| Bivalva (A) | bivalves | 0.22 | Straub (2008) |
| *Carex* spp. (V) | sedge | 1.36 | Straub (2008) |
| Cephalanthus spp. (V) | buttonbush | 0.78 | Straub (2008) |
| Ceratophyllum spp. (V) | coontail | 1.51 | Gross et al. (2020b) |
| Chara spp. (V) | muskgrass | 0.57 | Churchill (2015) |
| Cornus spp. (V) | dogwood | 0.89 | Straub (2008) |
| Elodea spp. (V) | waterweed | 1.66 | Gross (2020b) |
| Equisetum spp. (V) | horsetail | 2.5 | Kaminski et al. (2003)* |
| Gastropoda (A) | gastropods | 0.29 | Straub (2008) |
| Hydrocharis spp. (V) | frog-bit | 1.77 | Hydrocharacea from Straub (2008) |
| Insecta (A) | insects | 0.38 | Straub (2008) |
| Juncus spp. (V) | rush | 1.21 | Sherfy (1999) |
| Lemna spp. (V) | duckweed | 2.5 | Kaminski et al. (2003)* |
| Malacostraca (A) | crustaceans | 2.33 | Straub (2008) |
| Nuphar spp. (V) | water-lily | 2.5 | Kaminski et al. (2003)* |
| Pontederia spp. (V) | pickerel weed | 2.5 | Kaminski et al. (2003)* |
| Potamogeton spp. (V) | pondweed | 0.82 | Brasher et al. (2007) |
| Saggitaria spp. (V) | arrowhead | 3.06 | Hoffman & Bookhout (1985) |
| Schoenoplectus spp. (V) | club-bulrush | 0.82 | Livolsi et al. (2021) |
| Scirpus spp. (V) | bulrush | 0.57 | Sherfy (1999) |
| Sparganium spp. (V) | bur-reed | 0.96 | Straub (2008) |
| Zizania spp. (V) | wild rice | 3.47 | Sherfy (1999) |

* For unknowns, assume TME value of 2.5

In the laboratory, portions of each plant known to be used by waterfowl (Martin & Uhler 1939) were separated from the rest of the plant and then dried to a constant mass at 60°C and weighed to nearest 0.001 g. Invertebrate sampling was conducted when there was standing water present in the quadrat. We sampled invertebrates by sweeping a circular dipnet (500 µm mesh, 0.067 $m^2$ opening) once vertically from the substrate to the surface. Samples were then washed through a sieve bucket (595 µm mesh) and placed in polythene bags containing 70% ethanol. In the laboratory, invertebrates were separated from remaining plant material and identified to class level (Table 1.1). Invertebrates were then dried to a constant mass at 60°C and weighed to nearest 0.001 g. Forage biomass (dry mass) estimates were reported as kg/ha (Straub et al. 2012).

We calculated ECC using published true metabolizable energy (TME; kcal/g) values as described by Reinecke et al. (1989) (Table 1.1). When published TME values were not available for specific taxa, we substituted for the closest taxonomic group or assumed a value of 2.5 kcal/g (Kaminski et al. 2003; Straub et al. 2012). We then calculated ECC in terms of duck energy days (DEDs) by first multiplying food item biomass by its corresponding TME value; then we repeated this for all food items in a replicate, took the sum of all the products and divided by the daily energy requirement of a typical female mallard (292 kcal/day) (Straub 2008).

### *1.3.2 Statistical Analysis*

We performed data analysis and visualization in R version 4.2.2 (R Core Team 2024); the full analysis is documented in Appendix B. We estimated total biomass of available forage and DEDs per hectare to compare the relative value of the five wetland sites for waterfowl. We used two nested linear models with forage biomass and DEDs as the dependent variables. Site and food type (*i.e.*, invertebrate v. vegetation) were included as fixed-factors, and cover type was included as a fixed factor nested within site. We also included the proportional area of each cover type as weight, such that cover types with a larger proportional area in the site were given greater weight relative to cover types with a smaller area. We used the emmeans package (Lenth & Lenth 2018) to perform pairwise comparisons among sites and food types, using a Tukey adjustment for multiple comparisons.

To compare the relative abundance of forage and DEDs per hectare among cover types, we used unweighted, nested linear models with the relative biomass and DED value of invertebrate and vegetation food sources as dependent variables in separate models. We included site as a fixed factor, and cover type as a fixed factor nested within site. Again, we used the emmeans package to perform pairwise comparisons of cover types within each site, using a Tukey adjustment for multiple comparisons.

## 1.4 Results

Our nested linear model explained 37% of variation in forage biomass among our five mitigation wetlands ($p < 0.001$). Mean forage biomass per hectare ranged from 105 ± 5.6 kg/ha in our lowest quality site to 483 ± 16.7 kg/ha at our highest quality site (Figure 1.3A). While we detected a statistical difference ($p = 0.0315$) between the highest and lowest quality sites, there was no statistical difference among any other sites with respect to weighted mean forage biomass.

For energetic carrying capacity, our nested linear model explained 56% of variation in DEDs across our five study areas ($p < 0.001$). Weighted ECC per hectare varied from 121 ± 6.0 DED/ha at Cumberland Head to 2562 ± 34.3 DEDs/ha at Ausable Marsh WMA (Figure 1.3B). Similar to forage biomass, there was a significant difference ($p < 0.001$) between these sites on opposing ends of the quality gradient. The sites in the middle of the quality gradient showed some variation, USDA 2, USDA 1, and Lake Alice WMA were not statistically different from one another; however, USDA 1 was significantly different from Ausable Marsh WMA ($p = 0.0195$).

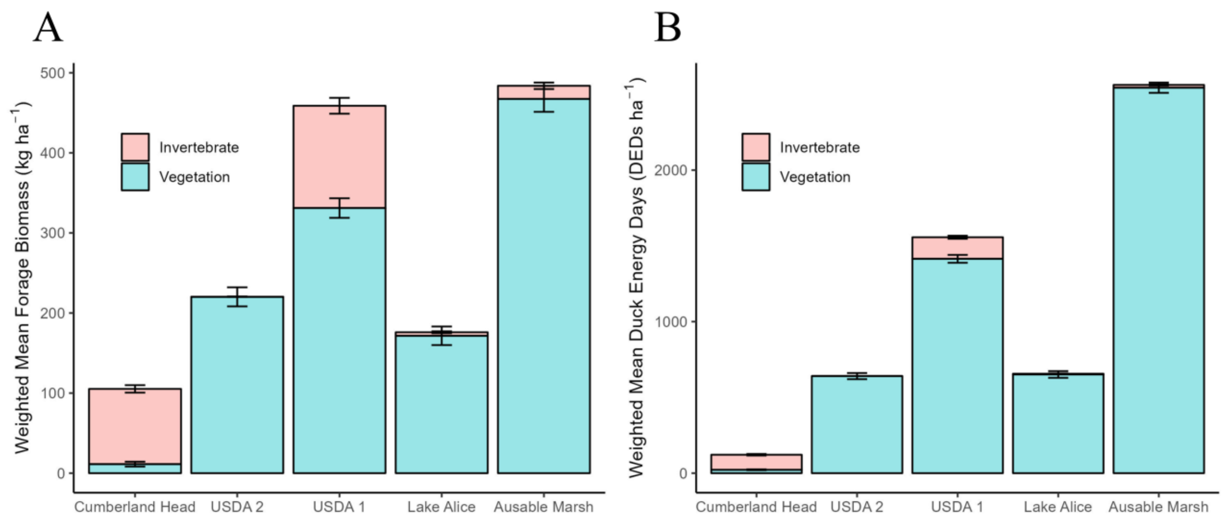


Figure 1.3 A) Weighted mean forage biomass (kg/ha) of vegetation (blue, $n$ = 85) and invertebrate (red, $n$ = 50) food sources for waterfowl at five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean biomass. B) Weighted mean Duck Energy Days (DED/ha) of vegetation (blue, $n$ = 85) and invertebrate (red, $n$ = 50) food sources for waterfowl at five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean DEDs.

Our unweighted nested linear model explained 49% of variation in forage biomass from vegetation sources ($p < 0.001$). Unweighted vegetation forage biomass at Ausable Marsh WMA was statistically different ($p < 0.05$) from all sites. Additionally, in the same pairwise comparison, Cumberland Head and USDA 1 were significantly different ($p$ = 0.042). For unweighted forage biomass from invertebrate food sources, our nested linear model explained 12% of variation ($p$ = 0.123). The wetland cover type that provided the greatest unweighted mean forage biomass varied across the sites. For vegetation food sources (Figure 1.4B), submergent marsh was statistically different ($p < 0.001$, $p$ = 0.00041, $p$ = 0.0033, $p$ = 0.0407) from all other cover types at Ausable Marsh WMA and USDA 1. Additionally, at three of the five sites, submergent marsh provides the greatest unweighted mean vegetation forage biomass. Emergent marsh contained the lowest mean vegetation forage biomass at all sites apart from USDA 2 where it produced the greatest biomass with large variation between replicates (494 ± 242.5 kg/ha). For invertebrate food sources (Figure 1.4A), unweighted mean forage biomass of cover types was similar across all sites, barring USDA 1, which was statistically different from Lake Alice ($p$ =

0.0185) and Ausable Marsh ($p = 0.0218$). Our pairwise comparisons revealed no significant differences between cover types within sites for unweighted invertebrate biomass.

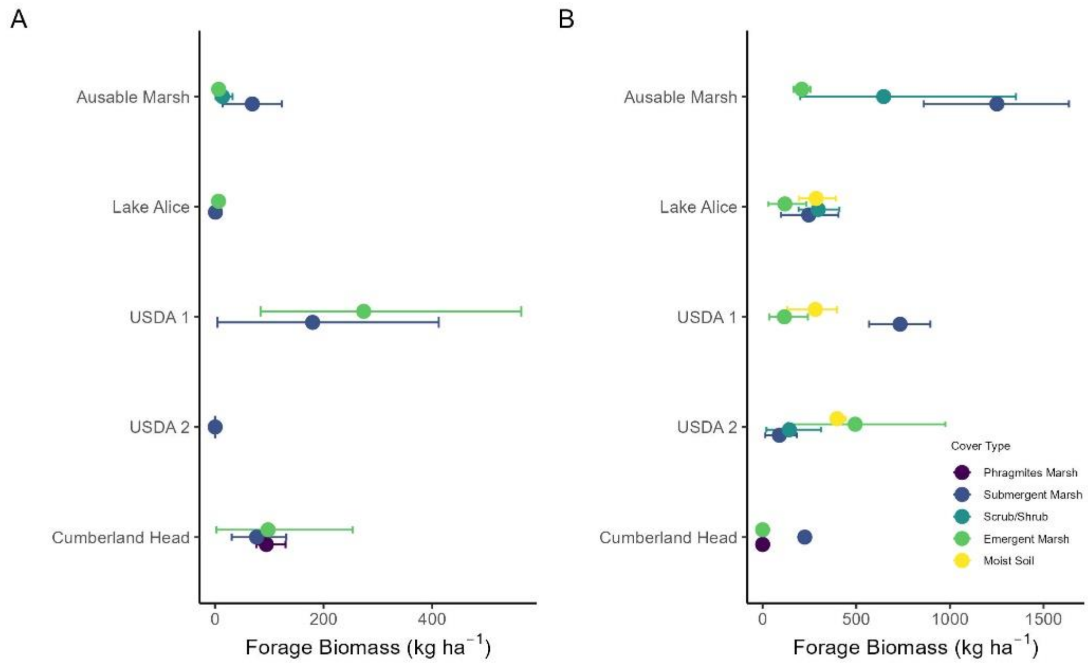


Figure 1.4 A) Unweighted mean forage biomass (kg/ha) for invertebrate food sources found in various wetland cover types in five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean forage biomass ($n = 50$). B) Unweighted mean forage biomass (kg/ha) for vegetation food sources found in various wetland cover types in five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean forage biomass ($n = 85$).

For unweighted energetic carrying capacity, our nested linear model explained 68% of variation from vegetation food sources ($p < 0.001$). Pairwise comparisons of this model revealed that Ausable Marsh WMA was significantly different ($p < 0.05$) from all other sites, and we detected a significant difference ($p = 0.002$) between Cumberland Head and USDA 1. Additionally, our unweighted nested linear model explained 14% of variation in ECC for invertebrate food sources ($p = 0.098$). Pairwise comparisons of this model revealed USDA 1 was significantly different ($p < 0.05$) from all sites except Cumberland Head. Unweighted mean ECC from vegetation food sources was generally evenly distributed among wetland cover types (Figure 1.5B). Unweighted mean vegetation DEDs was greatest in submergent marsh habitats at all sites apart from USDA 2. Submergent marsh was statistically different from other wetland cover types within two sites, Ausable Marsh WMA ($p < 0.001$) and USDA 1 ($p = 0.001$), where vegetation food sources in submergent marsh provided the greatest energy for waterfowl. The composition of unweighted invertebrate DEDs across cover types was similar for similar for all sites except USDA 1. Unweighted mean DEDs from invertebrate food sources was greatest at USDA 1 with 234 ± 148.9 DEDs/ha coming from submergent marsh and 243 ± 146.7 DEDs/ha coming from emergent vegetation (Figure 1.5A). Pairwise comparisons of unweighted DEDs from invertebrate food sources across cover types within sites reveal no significant differences.

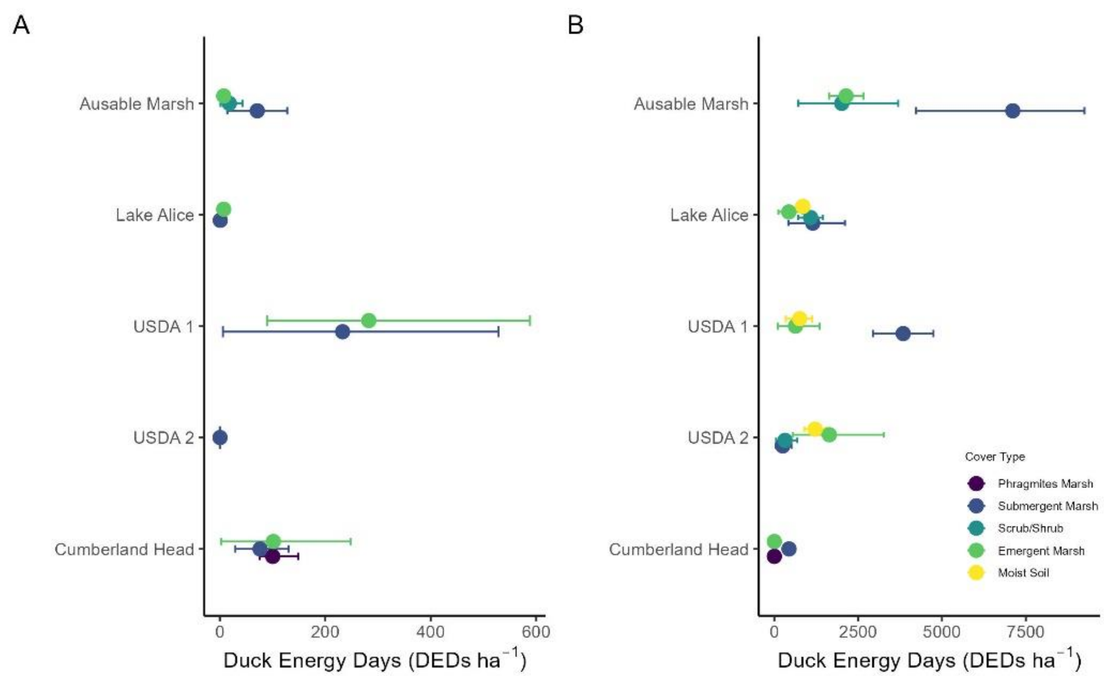


Figure 1.5 A) Unweighted mean duck energy days (DED/ha) for invertebrate food sources found in various wetland cover types in five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean DEDs ($n = 50$). B) Unweighted duck energy days (DED/ha) for vegetation food sources found in various wetland cover types in five mitigation wetland study areas in northern New York State, USA. Standard error bars represent ± 1 SE around the mean DEDs ($n = 85$).

Although Cumberland Head is the only wetland in our study that has a greater proportion (89.5%) of forage biomass from invertebrate food sources, we detected no statistical difference between food sources ($p = 0.3953$). On the highest end of the quality gradient, Ausable Marsh WMA, had a statistically significant ($p < 0.001$) difference between the forage types, with just 3.3% of its weighted mean forage biomass coming from invertebrate sources. A similar effect is observed when comparing proportion of energetic carrying capacity sourced from invertebrate and vegetation food sources; Cumberland Head has the greatest proportion from invertebrate food sources (81.8%), despite no statistical difference ($p = 0.8394$) being detected. We detected statistical differences between DEDs forage types in USDA 1 ($p = 0.001$) and Ausable Marsh WMA ($p < 0.001$). USDA 2 provided an undetectable (< 1) level of energy for waterfowl from invertebrate food sources.

### 1.5 Discussion

The findings of this study demonstrate that wetland quality strongly influences waterfowl forage availability and ECC, as evidenced by significant variation in DEDs and forage biomass across different mitigation quality levels. While sites in the middle of the quality gradient, USDA 2, USDA 1, and Lake Alice WMA, did not show statistical differences among themselves, a clear three-tiered quality gradient emerged, with Cumberland Head, USDA 1, and Ausable Marsh WMA exhibiting distinct differences. Notably, there was more than a tenfold increase in DEDs from the low-quality site (Cumberland Head) to the medium-quality site (USDA 1), and a further 60% increase from medium to the high-quality site (Ausable Marsh WMA). The highest quality wetland in the study provided over 20 times more DEDs than the lowest quality site, underscoring the importance of preserving wetland quality and function, not just wetland acreage, in mitigation efforts. Allowing repeated failed mitigation efforts to occur across the landscape risks the loss of important wetland functions and can lead to degradation of the wetland mosaic that wetland species depend on (Moreno-Mateos et al. 2012). This study emphasizes the need for further research on how mitigation quality can influence other wetland functions.

Our study found that submergent marsh vegetation resources provided more unweighted mean DEDs than any other cover type at four of our five study areas, highlighting their significant role in waterfowl resource production. Further, at three of the five study areas, the unweighted mean DEDs from submergent marsh vegetation exceeded those from all other cover types combined. This is a particularly crucial finding for mitigation sites like Cumberland Head, where the dominance of *Typha spp.* and invasive *Phragmites australis* has eliminated nearly all available emergent waterfowl forage. Our results are consistent with findings from Brasher et al. (2007) and Johnson (2007) that reveal submerged aquatic resources comprise a large portion of energy available to waterfowl. Submergent marsh wetlands are significant for their production of floating and submerged aquatic vegetation that are critical components of waterfowl diet (Weller 1981). The seeds, tubers, and foliage of several floating (*e.g.*, *Potamogeton nodosus*) and submerged aquatic species (*e.g.*, *Elodea canadensis*) are used by a variety of waterfowl species (Gross et al. 2020a). Waterfowl managers promote the growth of submerged aquatic vegetation by creating a deeper (> 0.5 m) wetland strata that is favored by hydrophytes (Weller 1989; Bookhout et al. 1989). This is an increasingly important management consideration for wetland mitigation as constructed wetlands experience high rates of *Phragmites* colonization (Havens et al. 2003).

Macroinvertebrates serve as crucial protein sources for waterfowl throughout various life stages (Havens et al. 2003). Our study found no significant correlation between mitigation quality and macroinvertebrate abundance. However, our findings suggest that invertebrate food sources may play a more significant role in poor-quality mitigation wetlands with low emergent and moist-soil plant diversity (*e.g.*, Cumberland Head). Furthermore, the cover type providing the greatest invertebrate resources to waterfowl varied among sites. This inter-site variation in invertebrate biomass and DEDs likely stems from site-specific characteristics. For instance, Gleason et al. (2018) observed differences in macroinvertebrate species richness between vegetated and open water areas within the same cover type, while finding no significant differences in macroinvertebrate communities as a whole across various wetland cover types (*e.g.*, emergent, open water). These findings, coupled with habitat heterogeneity, may explain the variance observed within cover types in our samples. Our results align with Hagy & Kaminski (2012), indicating that invertebrate forage is relatively insignificant for fall-migrating waterfowl compared to vegetation food sources. We acknowledge our methodology did not include distinct sampling for benthic invertebrates available to waterfowl and that primarily quantifying nektonic invertebrates likely underestimates total invertebrate availability.

Our study highlights the value of small, often overlooked wetlands for waterfowl forage provision. While small wetlands often fall below regulatory thresholds, these habitats comprise a network of small wetland patches that are important for maintaining connectivity for wetland-dependent species (Gibbs 1993). While much research has linked increased wetland size to greater waterfowl use (Rumble & Flake 1983; Lokemoen 1973; McKinstry & Anderson 2002), small natural and restored wetlands in the North Atlantic Flyway play an important role in supporting breeding waterfowl (Ringelman et al. 1982; Stevens et al. 2003). However, current mitigation techniques frequently lead to net loss of wetland area, with mitigation wetlands occupying less area than the original impacted wetlands (Moreno-Mateos et al. 2012). This reduction in patch size and creation of smaller, more fragmented wetland "islands," may lead to reduced species richness and altered community composition (Whigham 1999). Thus, while these small wetlands are ecologically significant for conserving wetland-dependent species and ecosystem services (Blackwell & Pilgrim 2011), the shift toward smaller and more isolated mitigation wetlands may be compromising their efficacy in maintaining biodiversity and providing ecological functions like providing stopover waterfowl habitat (Amezaga et al. 2002). Our data reveal that small mitigation wetlands can be an important component in the greater wetland network if their quality is preserved.

Overall, total weighted mean forage biomass and DEDs for waterfowl across all sites were comparable to similar published ECC studies (Vonbank & Hagy 2016; Brasher et al. 2007). While all study locations yielded lower vegetation forage availability than actively managed wetlands in the Great Lakes region (see Brasher et al. 2007), our results were similar to Straub et al. (2012) where vegetation and invertebrate food sources contribute to total forage biomass. Despite low forage biomass estimates relative to similar studies, several sites (USDA 2, USDA 1, Ausable Marsh) exceed the 181 kg/ha critical food density proposed by Hagy et al. (2017). Critical food density is the availability of forage required to meet the daily energy expenditures when the animal maximizes foraging effort. All study sites had low DEDs relative to similar ECC research on black ducks in New Jersey (Cramer 2009; Fino et al. 2017), and our estimates were again comparable to wetlands in the Midwest (see Brasher et al. 2007; Vonbank and Hagy 2016); however few studies have been conducted in other regions of the northeastern United States, including New York. Variations in estimates among studies is likely explained by differences in sites (*e.g.*, forage community composition, hydrology, climate, management regime) or differences in sampling (*e.g.*, time of year, number of samples, forage type sampled) and modeling methodology.

Many mitigation wetlands are designed for broad ecosystem functions with a focus on compliance with regulatory requirements; additionally, there lacks a standard of record keeping and available data making it challenging to draw broad conclusions on mitigation efforts (Mitsch & Wilson 1996). Although the primary goal of wetland mitigation may not be solely to manage for waterfowl, waterbirds and other avian species are commonly used as bioindicators of ecosystem health (Amat & Green 2010; Temple & Wiens 1989). Tozer et al. (2018) found that the occupancy of non-waterfowl species, including those of conservation concern, was higher in wetlands conserved under the North American Waterfowl Management Plan (NAWMP) compared to unmanaged sites. This suggests that wetland mitigation with waterfowl management objectives may have positive impacts on non-target species. Conversely, managing the plant community may unintentionally benefit waterfowl forage. Hrynevych et al. (2021) identified a positive relationship between macrophyte growth and nesting waterfowl. In the context of wetland mitigation, where permit objectives often aim to increase total diversity of wetland-dependent plants (Matthews & Endress 2008), successful implementation of these permits may inadvertently enhance waterfowl forage.

Bioenergetics models can be useful for setting habitat management goals and better understanding waterfowl forage dynamics. Due to habitat variations over different spatial and temporal scales, it is critical to apply ECC research to the proper scale prior to their application in management. This emphasizes the need for further waterfowl bioenergetics modeling at various spatial and temporal scales to better inform regional conservation planning. Additionally, determining ECC is highly dependent on prior true metabolizable energy research, which measures the energy an individual bird of a particular species derives from a particular food item. The TME values of many common waterfowl foods for many species are not yet known (Williams et al. 2014). The lack of available data requires researchers to infer TME values from published work on the closest related taxa, or follow assumptions made by prior researchers (*e.g.*, Straub et al. 2012). Further TME research for a variety of waterfowl foods is required to improve bioenergetics accuracy.

Further research is necessary to explore the extent to which waterfowl use small mitigation wetlands in a variety of regions and wetland types and how forage availability relates to habitat selection. Despite the time this methodology requires (Stafford et al. 2011), it is an accessible approach that requires limited specialized laboratory and field equipment. Additionally, sampling methods could be adapted to suit specific management or research goals to further limit time or resource constraints (*e.g.*, sampling target cover type). This research proposes an alternative to current methods of evaluating wetland mitigation success, and future research should investigate other novel methods such that success can be determined based on management goals. Research that expands our current understanding of small wetlands will further highlight the need for their conservation. Waterfowl research in the Atlantic Flyway is surprisingly limited considering the vast network of universities and conservation organizations located in its path.

## 1.6 Conclusion

The variation in forage availability among mitigation wetlands of varying quality highlights the need for more rigorous standards and monitoring in wetland mitigation projects. The creation of wetland area alone does not guarantee adequate habitat for target species. Managers should consider using energetic carrying capacity models for evaluating mitigation success in areas particularly important for waterfowl. This approach provides a more holistic evaluation of habitat value than traditional vegetation-based metrics alone.

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# Chapter 2. Using Game Cameras to Estimate Waterfowl Use of Small Mitigation Wetlands Along a Quality Gradient

## 2.1 Introduction

Despite covering less than 9% of global land area, wetlands provide a host of ecosystem services that are valuable to both humans and wildlife (Zedler and Kercher 2005). Wetlands can broadly be defined as habitats that serve as the interface between aquatic and terrestrial ecosystems (Junk et al. 2014). Nearly, half of all animal species and one third of all plant species listed under the Endangered Species Act depend on wetland habitats at some point in their life history (Nelson 1989), thus wetlands are critical for the conservation of global biodiversity. Wetlands are particularly important for conserving migratory bird species, like waterfowl, that use wetlands for foraging, breeding, and as migratory stopover habitat (Fredrickson and Reid 1988). Further, waterfowl survivorship and use has been closely linked to the proportion of available wetland area (Simpson et al. 2007; Stafford et al. 2010).

High rates of wetland losses in the United States in the 20$^{th}$ and 21$^{st}$ centuries are attributed to a variety of anthropogenic disturbances including draining for agricultural use and filling for urban development (Davidson 2014). The dramatic loss of wetlands and the potential loss of their associated benefits drove law makers to codify wetland protections into law under Section 404 of the Clean Water Act, which requires compensatory mitigation efforts if wetland areas are required to be disturbed (Zedler 1996). Permits allowing the disturbance of wetland areas require the creation, restoration, or enhancement of wetland habitats of equivalent area and function elsewhere in accordance with the nation’s “no net loss” policy of wetland conservation (Zedler 2004).

Despite such protections, mitigation wetlands often fail to meet the area and function of the originally disturbed wetlands (Turner et al. 2001). Incongruity between reference and mitigated wetlands can lead to a mosaic of wetlands across the landscape that lack important functions that were once present (Gibbs 1993).

Measuring the outcomes of wetland mitigation takes a variety of forms based on project goals, permit requirements, and resources, as such, there lacks consistency in wetland restoration criteria (Kentula 2000). Many mitigation permits and studies evaluating mitigation efforts use vegetation community structure to measure progress as wetland vegetation can rapidly colonize new habitats and is easily measured (Cole et al. 2006). However, quantifying the vegetation community alone measures a limited number of wetland functions and as such, the effect of mitigation quality on wildlife is unknown (Matthews and Endress 2008). This vegetation focused approach has led researchers to explore wildlife use of mitigation wetlands to evaluate novel methods of mitigation success criteria (Balcombe et al. 2005). Given the limitations of vegetation-focused assessments, it is crucial to explore alternative methods that can provide a more comprehensive understanding of wildlife interactions within these ecosystems, particularly for waterfowl, which are vital indicators of wetland health (Hagy et al. 2017).

To further understand the drivers of waterfowl habitat selection, it is first important to gather survey data of waterfowl use. Conducting ground surveys for waterfowl can be challenging with the risk of creating bias in surveys from human influence (Rumble and Flake 1982). In an attempt to minimize this risk, researchers have begun to implement alternative survey methods including aerial surveys from airplanes or drones (Dundas et al. 2021), eDNA detection (Honka et al. 2024), and camera trapping methods (Firth et al. 2020). Camera trapping techniques are increasingly implemented to conduct wildlife surveys with challenging access, to reduce disturbance, and to perform multiple surveys simultaneously (Gompper et al. 2006; Kelly 2008). While camera traps are typically implemented for surveying land animals, there have been increasing efforts to explore their application for avian species, including waterfowl (Feldman et al. 2024).

Waterfowl habitat conservation efforts are informed in part by bioenergetics models that use forage availability (kg/ha) to determine a given wetland's energetic carrying capacity

(ECC) for waterfowl (duck-energy days; DEDs) (Brasher et al. 2007; Straub 2008; Vonbank and Hagy 2016; Fino et al. 2017). These models allow managers to further understand the relationship between waterfowl use and forage availability (*see* Smith et al. 2011). While some studies have compared ECC for waterfowl and waterfowl use (*e.g.*, Hagy and Kaminski 2015), few studies have explored how waterfowl use varies along an energetics quality gradient, particularly in mitigation wetlands.

We estimated waterfowl use along a mitigation quality gradient using game camera surveys in the Fall of 2024 in northern New York state (Figure 2.1). Our objectives were to 1) Estimate fall waterfowl use in mitigation wetlands along a waterfowl forage quality gradient, 2) evaluate the application of game camera surveys for waterfowl.

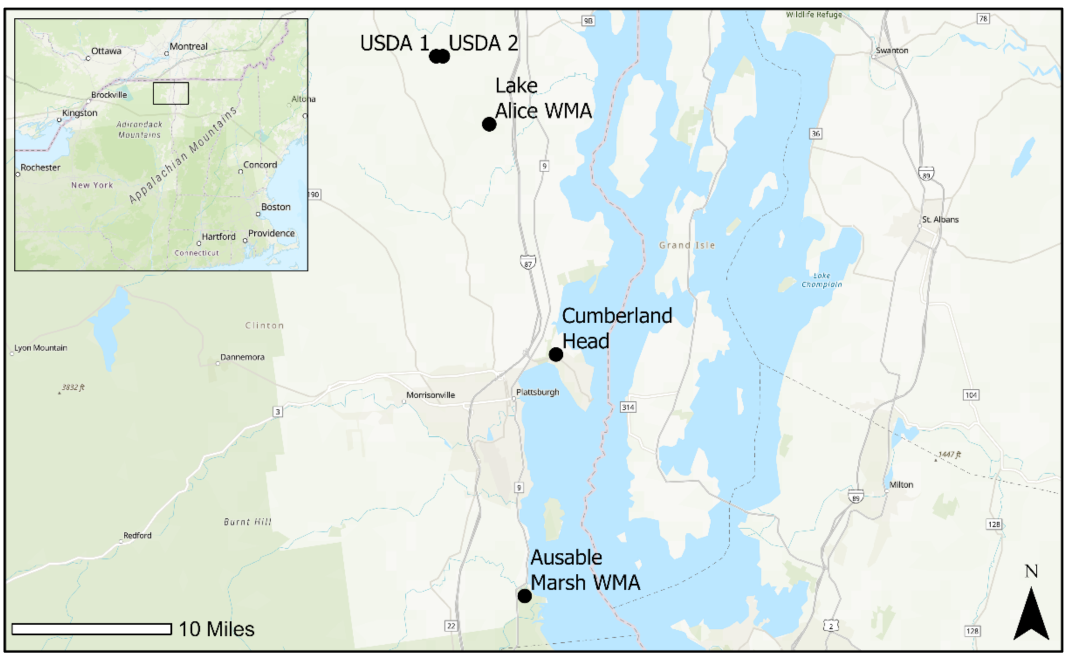


Figure 2.1 Map of study areas in the Lake Champlain Valley in northern New York State.

## 2.2 Study Area

We defined mitigation wetlands as restored, created, or enhanced wetlands designed to offset disturbances to wetland habitat elsewhere. We selected three wetlands based on these criteria along a quality gradient that were included in our waterfowl forage study (Kemmerling and Alldred 2024). In this study, wetland site quality was determined based on two factors: rapid assessment of hydrology, plant community, and presence of invasive species and waterfowl forage availability. We observed a strong alignment between rapid assessment quality and waterfowl forage availability for sites across the quality spectrum. Therefore, for the remainder of this paper, the terms “low quality”, “medium quality”, and “high quality” will be used interchangeably to refer to sites exhibiting corresponding levels of rapid assessment quality and waterfowl forage availability.

All three study areas are located in the Lake Champlain watershed in Northern New York. These wetlands are a part of a greater mosaic of an estimated 200,000 acres of diverse wetlands found in the New York portion of the Lake Champlain basin (Lake Champlain Basin Program 2013). Lake Champlain is an important migratory corridor within the greater Atlantic Flyway that connects migratory birds, including waterfowl, from the Gulf of St. Lawrence to the Long Island Sound and the Atlantic coast of New Jersey (Bellrose 1968). The wetlands in and around Lake Champlain are used by a variety of resident and migratory waterfowl species including wood ducks (*Aix sponsa*), American black ducks (*Anas rubripes*), and mallards (*Anas platyrhynchos*).

Each mitigation wetland in our study had different vegetation, hydrology, and geographic characteristics influenced by various management practices. The lowest quality mitigation wetland in our study, Cumberland Head (Figure 2.2A), is a marsh dominated by *Phragmites australis*, an invasive emergent plant that provides little value to waterfowl when unmanaged (Cross 1989). This mitigation site was originally intended as a forested wetland used to offset road development; however, the highly porous soils prevented the hydrology necessary for forested wetlands (O’Connor, oral comm., 2023). The medium quality site, Lake Alice (Figure 2.2B), is a constructed mitigation wetland with diverse vegetation communities and two sections of open water. The vegetation community is primarily composed of emergent vegetation including cattails (*Typha spp.*), sedges (*Carex spp.*), and rushes (*Scirpus spp.*). The highest quality site, Ausable Marsh (Figure 2.2C), is a wetland within a greater wildlife management area (WMA) with enhancements specifically for waterfowl. Wetland managers at this site improved the waterfowl habitat through geomorphic alterations, water control structures, and the planting of waterfowl foods like wild rice (*Zizania palustris*) (O’Connor, oral comm., 2023). Additionally, this wetland can become hydrologically connected to Lake Champlain during high water events.

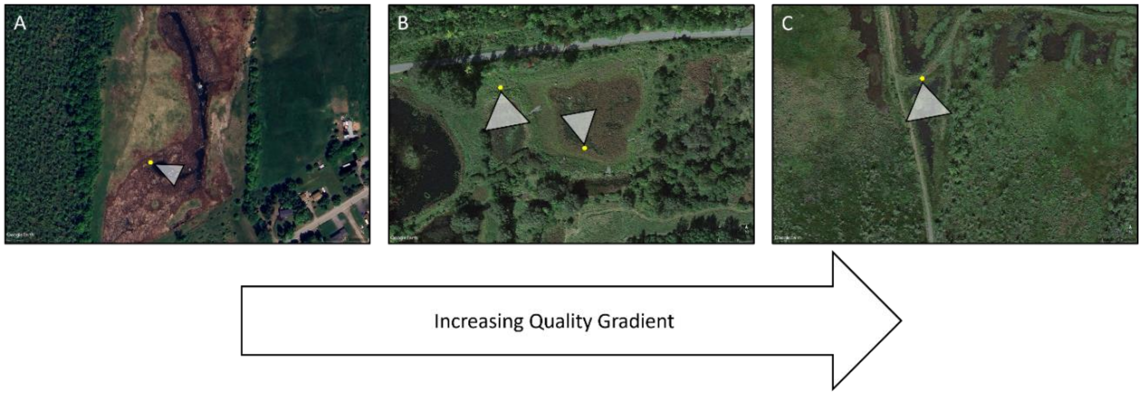


Figure 2.2 Map of camera placement (yellow) and field of view (white) of game cameras at three wetland mitigation sites of varying quality: A) Cumberland Head, low quality; B) Lake Alice, medium quality; and C) Ausable Marsh, high quality.

## 2.3 Methods

### *2.3.1 Game Camera Survey Procedure*

We conducted game camera surveys in the fall of 2024 using low-glow game cameras (Bushnell Prime L20). Each camera was mounted to a metal T-post, set at a 90º angle, and fastened using nylon straps (Figure 2.3). Each camera was placed such that the field of view contained the greatest amount of open water. In wetlands where the emergent vegetation may obstruct the view, the camera was placed on the edge of emergent vegetation and open water. Cameras were placed where regular access is feasible while limiting disturbance, theft, and a north facing angle. Each open water section received one game camera to compare waterfowl site selection and open water characteristics within wetlands.

Camera surveys were conducted concurrently at all sites from September 1$^{st}$ – December 17$^{th}$, 2024, with methods similar to those described by Berrus (2014). Cameras were programmed to capture images every five minutes for an hour at sunrise and before sunset daily. Camera data was retrieved on a weekly basis where camera cards were collected, and camera settings were regularly updated to compensate for changes sunrise and sunset times.

Images from game cameras were opened in HNU Image Manipulation Program (GIMP) to identify and count birds (Firth 2020). Each image was opened and birds within images were identified and counted. To prevent re-counting while processing images, a red mark was placed on each bird after identification (Firth 2020). Birds that we were unable to identify due to image quality or lighting were indicated as “unknown species.” Non-waterfowl birds were rarely observed and as such were left out of analysis.


Figure 2.3 A) Game camera fixed to a metal t-post and fastened at a 90º angle, B) Example photograph collected during a game camera survey for waterfowl conducted from September 1st – December 17th, 2023. Individual birds are marked with a single dot to prevent recounting.

### *2.3.2 Statistical Analysis*

We performed all analyses and visualization in R version 4.2.2 (R Core Team 2024). We used a significance threshold of $\alpha = 0.05$ for all statistical tests. We calculated descriptive statistics including total counts, mean, standard deviation, and average daily observations for each site quality level (low, medium, high). We used the “vegan” package in R to compute Shannon-Weiner diversity index (H’) and Simpson’s diversity index (1-D) for each site quality level to quantify evenness and species richness (Oksanen et al. 2020).

To compare total waterfowl abundance across site quality levels, we used a Chi-Square test of independence. This test assessed whether observed differences in total waterfowl counts among low-, medium-, and high-quality sites deviated significantly from expected distributions under the null hypothesis of equal use.

## 2.4 Results

In total, 828 individual birds were counted across the 2808 photographs taken during the survey period. At the low-quality site, birds were observed on 41 days of the total survey period for a total of 524 observed individuals (Table 2.1). At the medium-quality site, birds were observed on 33 days of the survey period for a total of 235 observed individuals. The highest quality site observed birds on 13 of the 106-day survey period for a total of 69 observed individuals. A Chi-Square goodness-of-fit test comparing these total counts across the site quality gradient revealed a highly significant difference in use among sites ($\chi^2 = 384.18$, df = 2, $p < 0.001$).

Table 2.1 Species observed, total number of individuals observed, species occurrence, and percent occurrence of waterfowl detected in camera surveys across three mitigation wetlands of varying quality in the Lake Champlain Valley, New York, USA. Surveys conducted from September 1st – December 17th, 2023.

| Site Name (*quality*) | Species Name | Total Individuals Observed | Species Occurrence | Percent Occurrence |
|---|---|---|---|---|
| Cumberland Head (low) | *Anas platyrhynchos* | 151 | 60 | 29 |
| | *Anas rubripes* | 135 | 41 | 26 |
| | Unknown species | 238 | 71 | 45 |
| | Site Total | 524 | | |
| Lake Alice (medium) | *Anas platyrhynchos* | 2 | 2 | 1 |
| | *Aix sponsa* | 61 | 48 | 26 |
| | *Branta canadensis* | 133 | 68 | 57 |
| | Unknown species | 39 | 39 | 16 |
| | Site Total | 235 | | |
| Ausable Marsh (high) | *Anas platyrhynchos* | 6 | 6 | 9 |
| | *Anas rubripes* | 2 | 1 | 3 |
| | *Aix sponsa* | 9 | 6 | 13 |
| | Uknown species | 52 | 30 | 75 |
| | Site Total | 69 | | |

Species richness was similar across all sites (Table 2.2). Each site had images where individual birds could not be identified and observations of waterfowl in this case were sorted into an "unknown species" category. At the low-quality site, Cumberland Head, two species were identifiable, American black duck (*Anas rubripes*) and mallard (*Anas platyrhynchos*). The medium quality site, Lake Alice, also had observations of the mallard with the addition of the Canada goose (*Branta canadensis*) and wood duck (*Aix sponsa*). At the high-quality site, Ausable Marsh, the mallard, American black duck, and wood duck were all observed. The unknown species category was the highest at Ausable Marsh (>75%) and made up less than 50% of the observations at Lake Alice and Cumberland Head (Figure 2.4).

Table 2.2 Shannon-Weiner diversity index (H'), Simpson's diversity index (1-D), and species richness across three wetland mitigation quality levels from game camera surveys for waterfowl conducted from September 1st – December 17th, 2023. All study areas are located in the Lake Champlain Valley, New York, USA.

| Site Quality | H' | 1-D | Species Richness |
|---|---|---|---|
| Low | 4.972010 | 0.99194 | 3 |
| Medium | 4.993555 | 0.99281 | 4 |
| High | 3.671552 | 0.97206 | 4 |

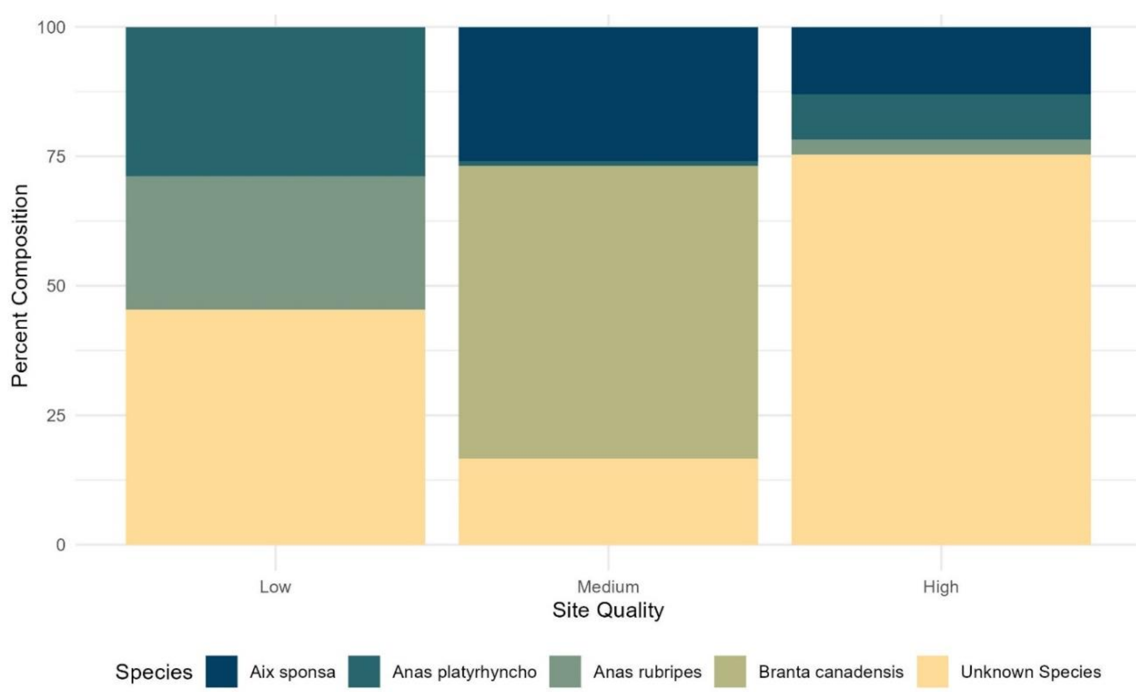


Figure 2.4 Percent composition of waterfowl species detected in camera surveys across three mitigation wetlands of varying quality in the Lake Champlain Valley, New York, USA. Surveys conducted from September 1st – December 17th, 2023.

Daily bird counts were variable across the survey period for all sites (Figure 2.5). The low and medium quality sites demonstrated two distinct peaks in observations: one in early September and another in mid-October. The high-quality site showed a single peak in early September, followed by a decline in observations until early October. In November, waterfowl detections were infrequent across all sites, with only 32 total observations recorded. No waterfowl were detected at any site during December.

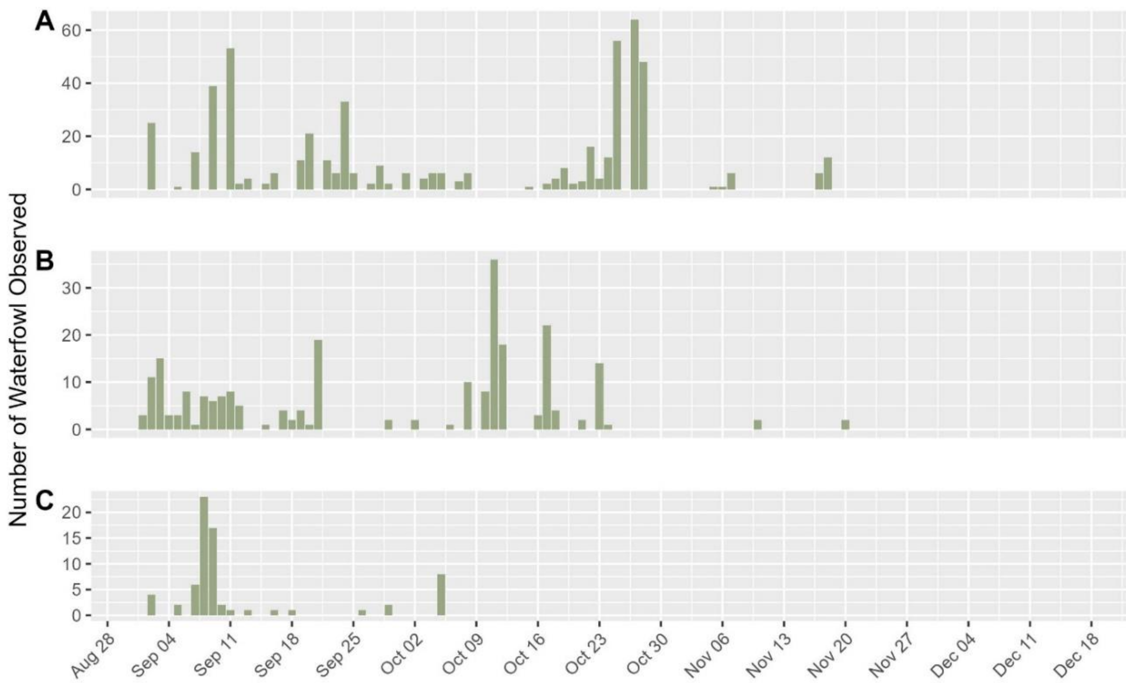


Figure 2.5 Number of individual waterfowl observed over the survey period; a) results from the low-quality site, b) results from the medium-quality site, c) results from the high-quality site.

## 2.5 Discussion

Our findings demonstrate that forage alone is not the most important driver of waterfowl habitat selection. While Cumberland Head was found to have the lowest ECC for waterfowl (Figure 1.3), it had the highest number of waterfowl observations over the survey period. Further, the site with the greatest DEDs had the lowest number of waterfowl observations. This suggests that while forage availability may be an important factor for waterfowl habitat selection, there are other wetland characteristics necessary to attract waterfowl. Osborn et al. (2017) suggested that dabbling duck wetland selection was positively correlated with not only increasing DEDs, but also with increasing emergent vegetation. While Cumberland Head had low DEDs, it was dominated by tall emergent vegetation like cattail (*Typha spp.*) and common reed (*Phragmites australis*). In particular, the submergent marsh cover type where our camera was placed was enclosed with common reed on all sides providing abundant cover for waterfowl. While common reed is understood to not be consumed by waterfowl (Livolsi 2015), its tall stalks can reach up to 13 feet and have occasionally been used by waterfowl for cover or nesting when interspersed with open water (Cross 1989).

Wetland size and proximity to other waterbodies can also influence waterfowl habitat selection and distribution (Stevens et al. 2003; Rumble and Flake 1983; McKinstry and Anderson 2002). Our results found the study area with the largest wetland size (Table A1) had the greatest number of waterfowl observed and the smallest wetland had the lowest number of waterfowl observations. While all our study areas were within 100 meters of other wetlands or waterbodies, the mitigation wetland Cumberland Head is located on a peninsula that extends into Lake Champlain. Lake Champlain serves as a significant migratory corridor for species like the American black duck (Lavretsky et al. 2014). Our data suggest that the combination of the proximity to the lake and abundant emergent vegetation as cover may be enough for waterfowl species to use this habitat despite the low forage availability.

Determining the true relationship between waterfowl forage availability and habitat selection in managed habitat on both private and public land is often challenging due to the various associated disturbances (*e.g.*, vehicles, foot traffic, hunting, recreation) (Pears et al. 2012; Osborn et al. 2017). The disturbance regimes varied between study areas depending on location and management regulations. Our study area, Ausable Marsh, was a subset of a larger wetland complex that makes up the Ausable Marsh Wildlife Management Area (WMA) managed by New York State DEC. It is open to the public for recreation including hunting and is adjacent to a regularly used trail. Similarly, the mitigation wetland, Lake Alice, is part of Lake Alice (WMA); however, waterfowl hunting is not permitted within the mitigation wetland itself. Cumberland Head is a county owned mitigation wetland where waterfowl hunting is prohibited, and public access is limited. It is difficult to say to what extent human disturbance may have influenced waterfowl habitat selection in our study as we did not directly quantify use; however, the site with the least restrictions on recreation and access did also have the lowest number of waterfowl observed. While our sample size was limited and waterfowl site selection could be more heavily influenced by habitat characteristics like forage availability, vegetation community composition, and wetland size, it is impossible to rule out human influence on waterfowl distribution.

Waterfowl species composition showed little variation between site quality (Table 2.1) and species richness was relatively low across all sites, with mallards and American black ducks being the most observed species. This finding is consistent with previous research that demonstrates the significance of the migratory corridor that connects the Atlantic Ocean, Hudson River, Lake Champlain, and the St. Lawrence River, particularly for mallards and American black ducks (Binhammer and Vermont Nature Conservancy 1994; Lavretsky et al. 2014; Lavretsky and Sedinger 2023). The low Shannon-Weiner diversity index (H') and Simpson's diversity index (1-D) values may be attributed to the limited size of the wetland areas the waterfowl can use. Elliot et al. (2020) found that wetland area was the primary driver of wetland bird species richness and abundance. Further, the authors found that maintaining habitat heterogeneity will enhance wetland bird diversity. Thus, while our data reflect the importance of Lake Champlain within the Atlantic Flyway for mallards and American black ducks, small wetlands could experience lower diversity due to the limited habitat heterogeneity that can exist in a small wetland.

Temporal patterns in waterfowl observations varied across study areas, with notable peaks and declines (Figure 6). The low- and medium-quality sites exhibited similar increases in observations during early September and mid-October, while the high-quality site experienced peak observations in early September, followed by a decline until observations ceased after the first week of October. These patterns partially align with findings from Lavretsky et al. (2014), who reported peak American black duck movement in the Lake Champlain valley during October and November. Our observation peaks in October were consistent with this pattern, as was the absence of December observations, which coincides with birds reaching their wintering grounds and the freezing of open water in our study areas. However, the limited number of observations in November (n = 32) across all sites is unexpected and may reflect changing migration patterns. Climate change is altering seasonal temperature and precipitation patterns that waterfowl use as migration cues, making it increasingly challenging to predict waterfowl migration timing and locations (Frei et al. 2024). This climate-induced variability could explain the discrepancies between our observations and historical migration patterns, highlighting the need for continued monitoring and adaptive management strategies in wetland conservation efforts.

The game camera methodology applied in this study proved to be an accessible tool for evaluating the relationship between waterfowl use and habitat quality, but it presented several challenges that could be addressed in future applications. While game cameras provide a unique non-invasive methodology to conduct multiple surveys simultaneously and continuously, game cameras that require battery replacement or image card retrieval may mean more disturbances during retrieval. Thus, if card retrieval is required, camera location should limit disturbance of target species. Further, financial constraints limited our ability to use higher resolution cameras, which made species identification challenging at long distances or when individuals were partially covered. This accounts for our large number of individuals recorded in the “unknown species” category. The large number of “unknown species” at Ausable Marsh could be due to the camera being located farther from the observation area compared to other sites. Future projects considering similar methodology for waterfowl, or other species, should consider desired image quality and card retrieval practices prior to implementation. Environmental factors, particularly the angle of the sun during the survey period must be considered as north facing cameras will limit glare and improve image quality. When applying camera traps in wetlands with multiple pockets of water, it is recommended to use methods outlined by Firth et al. (2020) that measure the exact field of view such that results can be extrapolated to other pockets of water of a known area. Additionally, our wetland sample size was constrained by landowner permission restrictions, which could potentially limit the generalization of our findings. Despite this limitation, game camera methodologies are a low-cost, accessible approach that can produce meaningful data when survey protocols and study area conditions are carefully considered.

## 2.6 Conclusion

This study demonstrates that while forage is an important component of waterfowl habitat selection, other wetland characteristics like emergent vegetation, proximity to other waterbodies, wetland size, and disturbance regime also play a significant role. Game camera methods proved effective in continuously monitoring waterfowl use of submergent marsh and open water areas, although image quality and camera placement are necessary considerations. This study emphasizes the need for waterfowl managers to implement wetland mitigation designs that consider all aspects of waterfowl life history requirements. This includes not only forage availability but cover from predation, proximity to other wetlands to offer a diverse array of habitat characteristics, and incorporating management regulations that balance waterfowl needs and recreational opportunities for local communities.

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# Appendix A. Supplementary Site Information

Appendix A provides additional information in support of Chapter 1, including:

- Table A1 Supplementary site information, including geographic coordinates, area, and duck energy days per hectare.
- Table A2 Most abundant plant species at each location, their total biomass, and percent cover.
- Table A3 Weighted mean forage biomass and duck energy days for waterfowl food items (vegetation and invertebrate) at each study location
- Table A4 Unweighted mean forage biomass and duck energy days for vegetation food sources, comparing value among cover types for each study location.
- Table A5 Unweighted mean forage biomass and duck energy days for invertebrate food sources, comparing value among cover types for each study location.

Table A1 Wetland coordinates, area (acres), and Duck energy days (DEDs) per hectare of three mitigation wetlands in the Lake Champlain Valley of New York State, USA. DEDs were calculated in the fall of 2023.

| Site Name | Wetland Coordinates | | Area (acres) | DEDs/ha (± SE) |
|---|---|---|---|---|
| | Latitude | Longitude | | |
| Cumberland Head | 44° 43' 40.6884" N | 73° 24' 48.6822" W | 12.06 | 121 ± 6.0 |
| Lake Alice | 44° 52' 34.9932" N | 73° 28' 24.9126" W | 3.44 | 656 ± 22.1 |
| Ausable Marsh | 44° 34' 19.9374" N | 73° 26' 31.2102" W | 1.19 | 2562 ± 34.3 |

Table A2 The most abundant plant species by total biomass (kg/ha) at each of the five mitigation wetlands.

| Site Name | Taxon | Total Biomass (kg/ha) | Relative Abundance (%) |
|---|---|---|---|
| Cumberland Head | *Chara vulgaris* | 1125.1 | 99.9 |
| | *Juncus effusus* | 1.1 | 0.1 |
| USDA 2 | *Scirpus cyperinus* | 1628.2 | 29.0 |
| | *Juncus effusus* | 1462.8 | 26.0 |
| | *Sparganium eurycarpum* | 607.5 | 10.8 |
| | *Scirpus atrovirens* | 545.0 | 9.7 |
| | *Potamogeton spirillus* | 446.6 | 8.0 |
| USDA 1 | *Ceratophyllum demersum* | 3949.0 | 70.0 |
| | *Scirpus cyperinus* | 826.6 | 14.6 |
| | *Juncus effusus* | 442.7 | 7.8 |
| | *Schoenoplectus tabernaemontani* | 137.2 | 2.4 |
| | *Scirpus atrovirens* | 110.5 | 2.0 |
| Lake Alice | *Scirpus atrovirens* | 767.1 | 16.2 |
| | *Potamogeton natans* | 702.1 | 14.8 |
| | *Juncus effusus* | 570.4 | 12.0 |
| | *Schoenoplectus tabernaemontani* | 508.2 | 10.7 |
| | *Cephalanthus occidentalis* | 408.6 | 8.6 |
| Ausable Marsh | *Elodea canadensis* | 6254.4 | 59.3 |
| | *Cephalanthus occidentalis* | 2820.4 | 26.8 |
| | *Zizania aquatica* | 637.1 | 6.0 |
| | *Hydrocharis morsus-raae* | 261.2 | 2.5 |
| | *Sagittaria* spp. | 153.9 | 1.5 |

Table A3 Weighted Mean Forage Biomass (kg/ha) and Energetic Carrying Capacity (DED/ha), Standard Error, and Standard Deviation for waterfowl food items in five mitigation wetlands in northern New York State, USA. Samples collected from August 15th – September 15th, 2023.

| | Forage Biomass | | | Energetic Carrying Capacity | | |
|---|---|---|---|---|---|---|
| Site Name[a] | x̄ | SE | SD | x̄ | SE | SD |
| Cumberland Head | | | | | | |
| Vegetation | 11 | 3.1 | 6.8 | 22 | 2.9 | 6.5 |
| Invertebrate | 94 | 4.7 | 9.2 | 99 | 5.2 | 10.0 |
| USDA2 | | | | | | |
| Vegetation | 220 | 11.9 | 26.6 | 641 | 20.3 | 45.4 |
| Invertebrate | 0.15 | 0.28 | 0.62 | 0 | 0.3 | 0.6 |
| USDA1 | | | | | | |
| Vegetation | 331 | 12.2 | 27.3 | 1414 | 26.0 | 58.1 |
| Invertebrate | 128 | 9.9 | 22.2 | 143 | 10.1 | 22.6 |
| Lake Alice | | | | | | |
| Vegetation | 172 | 11.6 | 25.9 | 651 | 22.1 | 49.3 |
| Invertebrate | 4 | 1.2 | 2.5 | 5 | 1.2 | 2.4 |
| Ausable Marsh | | | | | | |
| Vegetation | 467 | 16.2 | 36.1 | 2544 | 34.0 | 76.0 |
| Invertebrate | 16 | 4.1 | 8.7 | 18 | 4.3 | 9.3 |

[a] Sites are along a quality gradient from lowest quality (top) to highest quality (bottom).

Table A4 Unweighted mean forage biomass (kg/ha) and energetic carrying capacity (DEDs/ha) for fall migrating waterfowl from vegetation food sources at five mitigation wetlands in northern New York, USA. Sampled from August 15th – September 15th, 2023.

| | Forage Biomass | | | Energetic Carrying Capacity | | |
|---|---|---|---|---|---|---|
| Site Name | x̄ | SE | SD | x̄ | SE | SD |
| Ausable Marsh | | | | | | |
| emergent marsh | 209.52 | 26.05 | 58.25 | 2141.45 | 291.80 | 652.49 |
| scrub/shrub | 646.34 | 343.70 | 768.54 | 2011.31 | 886.13 | 1981.44 |
| submergent marsh | 1250.88 | 236.02 | 527.76 | 7111.17 | 1341.77 | 3000.29 |
| Lake Alice | | | | | | |
| emergent marsh | 118.83 | 63.17 | 141.25 | 432.54 | 201.31 | 450.15 |
| moist-soil | 285.70 | 54.70 | 122.30 | 852.72 | 110.54 | 247.18 |
| scrub/shrub | 297.02 | 64.58 | 144.41 | 1081.04 | 214.63 | 479.94 |
| submergent marsh | 246.11 | 89.79 | 200.78 | 1146.74 | 503.69 | 1126.28 |
| USDA 1 | | | | | | |
| emergent marsh | 116.27 | 63.89 | 142.86 | 631.46 | 362.56 | 810.71 |
| moist-soil | 281.52 | 80.52 | 180.06 | 758.61 | 225.06 | 503.24 |
| submergent marsh | 734.09 | 95.64 | 213.86 | 3843.42 | 520.41 | 1163.68 |
| USDA 2 | | | | | | |
| emergent | 494.31 | 242.54 | 542.34 | 1640.88 | 841.75 | 1882.21 |
| moist-soil | 397.74 | 21.85 | 48.86 | 1215.69 | 187.76 | 419.83 |
| scrub/shrub | 142.98 | 85.60 | 191.42 | 319.61 | 185.48 | 414.75 |
| submergent marsh | 89.53 | 49.15 | 109.91 | 251.42 | 138.04 | 308.66 |
| Cumberland Head | | | | | | |
| emergent marsh | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 | 0.00 |
| phragmites marsh | 0.21 | 0.21 | 0.47 | 0.88 | 0.88 | 1.96 |
| submergent marsh | 225.02 | 10.28 | 22.99 | 439.27 | 20.07 | 44.87 |

Table A5 Unweighted mean forage biomass (kg/ha) and energetic carrying capacity (DEDs/ha) for fall migrating waterfowl from invertebrate food sources at five mitigation wetlands in northern New York, USA. Sampled from August 15th – September 15th, 2023.

| | Forage Biomass | | | Energetic Carrying Capacity | | |
|---|---|---|---|---|---|---|
| Site Name | x̄ | SE | SD | x̄ | SE | SD |
| Ausable Marsh | | | | | | |
| emergent marsh | 6.78 | 3.38 | 7.55 | 7.48 | 3.42 | 7.65 |
| scrub/shrub | 13.99 | 9.84 | 19.68 | 18.10 | 13.12 | 26.25 |
| submergent marsh | 68.74 | 30.60 | 68.41 | 71.00 | 32.04 | 71.64 |
| Lake Alice | | | | | | |
| emergent marsh | 6.60 | 2.90 | 5.81 | 7.48 | 2.88 | 5.76 |
| submergent marsh | 0.93 | 0.57 | 1.27 | 0.96 | 0.58 | 1.30 |
| USDA 1 | | | | | | |
| emergent marsh | 273.81 | 144.07 | 322.15 | 282.59 | 146.77 | 328.19 |
| submergent marsh | 179.79 | 130.43 | 291.64 | 232.91 | 148.90 | 332.95 |
| USDA 2 | | | | | | |
| submergent marsh | 0.39 | 0.18 | 0.40 | 0.46 | 0.20 | 0.45 |
| Cumberland Head | | | | | | |
| emergent marsh | 97.76 | 73.89 | 147.78 | 101.47 | 77.70 | 155.40 |
| *Phragmites* marsh | 94.62 | 17.69 | 30.64 | 100.35 | 24.27 | 42.03 |
| submergent marsh | 76.68 | 28.41 | 63.54 | 76.38 | 28.26 | 63.18 |

# Appendix B. R Script

The following document contains the script for performing the unweighted and weighted forage biomass and duck-energy days calculations with the data we collected. Included is a data dictionary along with a step-by-step R script applicable for our data or similarly formatted data.

For more information and access to raw data visit:

https://figshare.com/articles/dataset/The_Role_of_Small_Mitigation_Wetlands_in_Supporting_Fall_Migrating_Waterfowl/27154458

**Data Dictionary**

Here we list all variables in our data with a brief description.

Table B1 Data dictionary for the role of mitigation wetlands in supporting fall migrating waterfowl.

| Variable | Definition |
|---|---|
| sample_point | A five-to-seven-digit code that indicates which wetland site was sampled, followed by an abreviation of the cover type, followed by a number representing the replicate number. |
| site_name | Indicates which wetland site was sampled. |
| cover_type | Indicates which wetland covertype was sampled. |
| replicate_number | Indicates which replicate the sample comes from. |
| tme | True Metabolizable Energy (TME) value in kilocalories per gram. |
| tme_taxa | Assigns the taxonomic group of the sample associated with a True Metabolizable Energy (TME) value. |
| scientific_name | Indicates the scientific name of the waterfowl food item. |
| p_cover | Indicates the percent coverage of that specific sample within the replicate plot. |
| plot_size_m2 | Indicates the size of the quadrat used at that specific sample location in square meters. |

| Variable | Definition |
| --- | --- |
| dry_mass_g | Indicates the dry mass of the food item in grams. |
| kg_ha | Indicates the biomass of the food item scaled up in kilograms per hectare. |
| g_m2 | Mass of the sample per meter squared. |
| class | Indicates the taxonomic rank of the invertebrate to the class level. |
| area_acres | Indicates the area of the specified cover type in acres. |
| area_hectares | Indicates the area of the specified cover type in hectares. |
| proportion | Indicates the proportion of the total wetland area the specified cover type occupies. |
| SM | Submergent marsh wetland cover type. |
| PM | Phragmites marsh wetland cover type. |
| EM | Emergent marsh wetland cover type. |
| SS | Scrub/shrub wetland cover type. |
| MS | Moist-soil wetland cover type. |
| AM | Ausable marsh wetland site. |
| CH | Cumberland Head wetland site. |
| LA | Lake Alice wetland site. |
| USDA1 | The first USDA conservation easement wetland site. |
| USDA2 | The second USDA conservation easement wetland site. |

### Script Setup

```
#### Load Packages ####
library(tidyverse)
library(emmeans)
library(patchwork)

#### Import Data Files ####
veg <- read.csv("data/vegetation_data.csv")
inv <- read.csv("data/invertebrate_data.csv")
cover <- read.csv("data/covertype_data.csv")
```

### Order Quality Gradients

```
#### Order Quality Gradient ####
veg$site_name <- ordered(veg$site_name, levels = c("cumberland head", "usda2",
                                                   "usda1", "lake alice",
                                                   "ausable marsh"))
veg$cover_type <- ordered(veg$cover_type, levels = c("phragmites marsh", "submergent ma
rsh",
                                                     "scrub/shurb", "emergent marsh", "
moist-soil"))
#Order quality gradient
inv$site_name <- ordered(inv$site_name, levels = c("cumberland head", "usda2",
                                                   "usda1", "lake alice",
```

```
                                                      "ausable marsh"))
inv$cover_type <- ordered(inv$cover_type, levels = c("phragmites marsh", "submergent ma
rsh",
                                                        "scrub/shurb", "emergent marsh", "
moist-soil"))
#Order quality gradient
cover$site_name <- ordered(cover$site_name, levels = c("cumberland head", "usda2",
                                                        "usda1", "lake alice",
                                                        "ausable marsh"))
cover$cover_type <- ordered(cover$cover_type, levels = c("phragmites marsh", "submergen
t marsh",
                                                            "scrub/shurb", "emergent marsh
", "moist-soil"
```

## Unweighted Calculations

```
#### Unweighted Calculations ####

#site area calculation
site_area <- cover %>%
  group_by(site_name) %>%
  summarize(area = sum(area_hectares))

#Calculate vegetation forage biomass within each replicate
veg_bm <- veg %>%
  group_by(site_name, cover_type, replicate_number) %>%
  summarize(kg_ha = sum(kg_ha, na.rm = T)) %>%
  mutate(type = "vegetation")

#Mean vegetation forage biomass across each cover type
veg_bm_covertypes <- veg_bm %>%
  group_by(site_name, cover_type) %>%
  summarize(mean = mean(kg_ha, na.rm = T),
            sd = sd(kg_ha, na.rm = T),
            se = sd/sqrt(length(na.omit(kg_ha))))

#Calculate invertebrate forage biomass within each replicate
inv_bm <- inv %>%
  group_by(site_name, cover_type, replicate_number) %>%
  summarize(kg_ha = sum(kg_ha, na.rm = T)) %>%
  mutate(type = "invertebrate")

#Mean invertebrate forage biomass across each cover type
inv_bm_covertypes <- inv_bm %>%
  group_by(site_name, cover_type) %>%
  summarize(mean = mean(kg_ha, na.rm = T),
            sd = sd(kg_ha, na.rm = T),
            se = sd/sqrt(length(na.omit(kg_ha))))

#Calculate vegetation duck-use days within each replicate
veg_dud <- veg %>%
  #First step, multiplying the food item biomass by the true metabolizable energy (TME)
value. Multiply by 1000 to convert units.
  mutate(kcal_ha = kg_ha * tme * 1000) %>%
  #Claculate the average duck-use days per unit area by taking the average of (the sum
of (food biomass * TME) and dividing by the daily energy requirement of an average fema
le mallard sized duck.)
  group_by(site_name, cover_type, replicate_number) %>%
```

```
  summarize(dud = sum(kcal_ha, na.rm = T)/292) %>% #gives total dud in replicate
  mutate(type = "vegetation")

#Mean vegetation duck-use days across each cover type
veg_dud_covertypes <- veg_dud %>%
  group_by(site_name, cover_type) %>%
  summarize(dud_m = mean(dud, na.rm = T),
                     sd = sd(dud, na.rm = T),
                     se = sd/sqrt(length(na.omit(dud))))

#Calculate invertebrate duck-use days within each replicate
inv_dud <- inv %>%
  mutate(kcal_ha = kg_ha * tme * 1000) %>%
  group_by(site_name, cover_type, replicate_number) %>%
  summarize(dud = (sum(kcal_ha))/292) %>%
  mutate(type = "invertebrate")

#Mean invertebrate duck-use days across each cover type
inv_dud_covertypes <- inv_dud %>%
  group_by(site_name, cover_type) %>%
  summarize(dud_m = mean(dud, na.rm = T),
            sd = sd(dud, na.rm = T),
            se = sd/sqrt(length(na.omit(dud))))
```

## Weighted Calculations

```
#### Weighted Calculations ####

#vegetation and invertebrate combined for forage biomass stacked bar
veg_inv_bm <- veg_bm %>%
  bind_rows(inv_bm) %>%
  left_join(cover, by = c("site_name", "cover_type"))

#weighted mean combined forage biomass to account for cover type area to compare sites
along quality gradient
veg_inv_bm_w <- veg_inv_bm %>%
  group_by(site_name, cover_type, type) %>%
  summarize(kg_ha_a = mean(kg_ha, na.rm = T),
            proportion = mean(proportion, na.rm = T),
            kg_ha_w = kg_ha_a * proportion,
            sd = sqrt(sum(proportion * ((kg_ha - kg_ha_w)^2))/(sum(kg_ha)-1)),
            se = sd/sqrt(length(na.omit(kg_ha)))) %>%
  ungroup() %>%
  group_by(site_name, type) %>%
  summarize(weighted_biomass = sum(kg_ha_w),
            sd = sum(sd),
            se = sum(se))

#vegetation and invertebrate combined for dud stacked bar
veg_inv <- veg_dud %>%
  bind_rows(inv_dud) %>%
  left_join(cover, by = c("site_name", "cover_type"))

#weighted mean duck-use days to account for cover type area to compare sites along qual
ity gradient
veg_inv_dud <- veg_inv %>%
  group_by(site_name, cover_type, type) %>%
  summarize(dud_a = mean(dud, na.rm = T),
```

```
            proportion = mean(proportion, na.rm = T),
            dud_w = dud_a * proportion,
            sd = sqrt(sum(proportion * ((dud - dud_w)^2))/(sum(dud)-1)),
            se = sd/sqrt(length(na.omit(dud)))) %>%
  ungroup() %>%
  group_by(site_name, type) %>%
  summarize(dud_t = sum(dud_w),
            sd = sum(sd),
            se = sum(se))

bm_over <- veg_inv_bm_w %>%
  group_by(site_name) %>%
  summarize(wbm = sum(weighted_biomass),
            sd = sum(sd),
            se = sum(se))

dud_over <- veg_inv_dud %>%
  group_by(site_name) %>%
  summarize(dud_m = sum(dud_t),
            sd = sum(sd),
            se = sum(se))
```

### Statistical Modeling

```
#### Statistical Modeling ####

#Run model for mean weighted forage biomass along quality gradient
bm_mod <- lm(kg_ha~site_name + site_name:cover_type + site_name:type,
              weights = proportion, data = veg_inv_bm)
#inspect model residuals
hist(bm_mod$residuals)

#compare sites
bm_site <- emmeans(bm_mod, specs = pairwise ~ site_name)
bm_site$contrasts
#Compare forage types within site
bm_type <- emmeans(bm_mod, specs = pairwise ~ type|site_name)
bm_type$contrasts

#Statistical model for vegetation biomass across cover_type and site
veg_mod_bm <- lm(kg_ha~site_name + site_name:cover_type, data = veg_bm)
#inspect model residuals
hist(veg_mod_bm$residuals)

#compare sites
emm_veg_site_bm <- emmeans(veg_mod_bm, specs = pairwise ~ site_name)
emm_veg_site_bm$contrasts
#Compare cover types within site
emm_veg_cover <- emmeans(veg_mod_bm, specs = pairwise ~ cover_type|site_name)
emm_veg_cover$contrasts

#Statistical model for invertebrate biomass across cover_type and site
inv_mod_bm <- lm(kg_ha~site_name + site_name:cover_type, data = inv_bm)
#inspect model residuals
hist(inv_mod_bm$residuals)

#compare sites
emm_inv_site_bm <- emmeans(inv_mod_bm, specs = pairwise ~ site_name)
```

```
emm_inv_site_bm$contrasts
#Compare cover types within site
emm_inv_cover <- emmeans(inv_mod_bm, specs = pairwise ~ cover_type|site_name)
emm_inv_cover$contrasts



#DEDs


## Run model for mean weighted duck-use days along quality gradient
dud_mod <- lm(dud~site_name + site_name:cover_type + site_name:type,
              weights = proportion, data = veg_inv)
#inspect model residuals
hist(dud_mod$residuals)

#compare sites
dud_site <- emmeans(dud_mod, specs = pairwise ~ site_name)
dud_site$contrasts
#Compare forage types within site
dud_type <- emmeans(dud_mod, specs = pairwise ~ type|site_name)
dud_type$contrasts

#Statistical model for vegetation duck-use days across cover_type and site
veg_mod <- lm(dud~site_name + site_name:cover_type, data = veg_dud)
#inspect model residuals
hist(veg_mod$residuals)

#compare sites
emm_veg_site <- emmeans(veg_mod, specs = pairwise ~ site_name)
emm_veg_site$contrasts
#Compare cover types within site
emm_veg_cover <- emmeans(veg_mod, specs = pairwise ~ cover_type|site_name)
emm_veg_cover$contrasts

#Statistical model for invertebrate duck-use days across cover_type and site
inv_mod <- lm(dud~site_name + site_name:cover_type, data = inv_dud)
#inspect model residuals
hist(inv_mod$residuals)

#compare sites
emm_inv_site <- emmeans(inv_mod, specs = pairwise ~ site_name)
emm_inv_site$contrasts
#Compare cover types within site
emm_inv_cover <- emmeans(inv_mod, specs = pairwise ~ cover_type|site_name)
emm_inv_cover$contrasts
```

## Plotting

```
#### Plotting ####


cols = c("submergent marsh" = "#3B528B",
         "emergent marsh" = "#5DC863",
         "scrub/shurb" = "#21908C",
         "moist-soil" = "#FDE725",
         "phragmites marsh" = "#440154")


#Plot of weighted mean forage biomass across quality gradient
veg_inv_bm_error <- veg_inv_bm_w %>%
  arrange(type, desc(weighted_biomass)) %>%
```

```
  arrange(site_name, desc(type)) %>%
  group_by(site_name) %>%
  #compute cumulative mean for error bars
  mutate(mean_new = cumsum(weighted_biomass)) %>%
  ungroup()

ggplot(veg_inv_bm_w, aes(x = site_name, y = weighted_biomass, fill = type)) +
  geom_bar(stat = "identity", alpha = 0.4,  color = "#000000")+
  geom_errorbar(data = veg_inv_bm_error,
                aes(x = site_name, ymax = mean_new + se, ymin = mean_new - se),
                width = 0.2)+
  theme_classic() +
  labs(x = "",
       y = expression(paste("Weighted Mean Forage Biomass (kg ", ha^-1, ")")),
       fill = "",
       color = "") +
  scale_fill_discrete(labels = c("Invertebrate", "Vegetation")) +
  scale_x_discrete(labels=c("Cumberland Head","USDA 2", "USDA 1", "Lake Alice", "Ausabl
e Marsh")) +
  theme(legend.position = c(0.2, 0.8))
```

```
#Plot of weighted mean duck-use days across quality gradient
#Create error for error bars (return to this when we have se)
veg_inv_error <- veg_inv_dud %>%
  arrange(type, desc(dud_t)) %>%
  arrange(site_name, desc(type)) %>%
  group_by(site_name) %>%
  #compute cumulative mean for error bars
  mutate(mean_new = cumsum(dud_t)) %>%
  ungroup()

ggplot(veg_inv_dud, aes(x = site_name, y = dud_t, fill = type)) +
  geom_bar(stat = "identity", alpha = 0.4, color = "#000000")+
  geom_errorbar(data = veg_inv_error,
                aes(x = site_name, ymax = mean_new + se, ymin = mean_new - se),
                width = 0.2)+
  theme_classic() +
  labs(x = "",
       y = expression(paste("Weighted Mean Duck Energy Days (DEDs ", ha^-1, ")")),
       fill = "",
       color = "") +
  scale_fill_discrete(labels = c("Invertebrate", "Vegetation")) +
  scale_x_discrete(labels=c("Cumberland Head","USDA 2", "USDA 1", "Lake Alice", "Ausabl
e Marsh")) +
  theme(legend.position = c(0.2, 0.8))
```

```
#Plot of unweighted vegetation duck-use days between cover types across sites
veg_plot <- ggplot(veg_dud, aes(x = site_name, y = dud, color = cover_type)) +
  stat_summary(fun.data = mean_cl_boot, geom = "errorbar", width = 0.2, position = posi
tion_dodge(width = 0.2)) +  # Error bars with dodge
  stat_summary(fun = mean, geom = "point", size = 3, position = position_dodge(width =
0.2)) +  # Mean points with dodge
  coord_flip() +
  theme_classic() +
  labs(x = "",
       y = expression(paste("Duck Energy Days (DEDs ", ha^-1, ")")),
```

```
       color = "Cover Type") +
  scale_x_discrete(labels = c("Cumberland Head", "USDA 2", "USDA 1", "Lake Alice", "Aus
able Marsh")) +
  scale_color_manual(values = cols,
                     labels = c("Phragmites Marsh", "Submergent Marsh", "Scrub/Shrub",
"Emergent Marsh", "Moist-soil")) +
  theme(legend.position = c(0.8, 0.17),legend.key.size = unit(.4, 'cm'), legend.text =
element_text(size = 6), legend.title = element_text(size = 6))

#Plot of unweighted invertebrate duck-use days between cover types across sites
inv_plot <- ggplot(inv_dud, aes(x = site_name, y = dud, color = cover_type)) +
  stat_summary(fun.data = mean_cl_boot, geom = "errorbar", width = 0.2, position = posi
tion_dodge(width = 0.2)) +  # Error bars with dodge
  stat_summary(fun = mean, geom = "point", size = 3, position = position_dodge(width =
0.2)) +  # Mean points with dodge
  coord_flip() +
  theme_classic() +
  labs(x = "",
       y = expression(paste("Duck Energy Days (DEDs ", ha^-1, ")")),
       color = "Cover Type") +
  scale_x_discrete(labels = c("Cumberland Head", "USDA 2", "USDA 1", "Lake Alice", "Aus
able Marsh")) +
  scale_color_manual(values = cols,
                     labels = c("Phragmites Marsh", "Submergent Marsh", "Scrub/Shrub",
"Emergent Marsh", "Moist-soil")) +
  theme(legend.position = "none")

##Plot of unweighted vegetation and invertebrate duck-use days between cover types acro
ss sites
inv_plot + veg_plot +
  plot_annotation(tag_levels = "A")
```

```
#Plot of unweighted vegetation forage biomass between cover types across sites
veg_plot_bm<- ggplot(veg_bm, aes(x = site_name, y = kg_ha, color = cover_type)) +
  stat_summary(fun.data = mean_cl_boot, geom = "errorbar", width = 0.2, position = posi
tion_dodge(width = 0.2)) +  # Error bars with dodge
  stat_summary(fun = mean, geom = "point", size = 3, position = position_dodge(width =
0.2)) +  # Mean points with dodge
  coord_flip() +
  theme_classic() +
  labs(x = "",
       y = expression(paste("Forage Biomass (kg ", ha^-1, ")")),
       color = "Cover Type") +
  scale_x_discrete(labels=c("Cumberland Head","USDA 2", "USDA 1", "Lake Alice", "Ausabl
e Marsh")) +
  scale_color_manual(values = cols,
                     labels = c("Phragmites Marsh", "Submergent Marsh", "Scrub/Shrub",
"Emergent Marsh", "Moist-soil")) +
  theme(legend.position = c(0.8, 0.17),legend.key.size = unit(.4, 'cm'), legend.text =
element_text(size = 6), legend.title = element_text(size = 6))



#Plot of unweighted invertebrate forage biomass between cover types across sites
inv_plot_bm <- ggplot(inv_bm, aes(x = site_name, y = kg_ha, color = cover_type)) +
  stat_summary(fun.data = mean_cl_boot, geom = "errorbar", width = 0.2, position = posi
tion_dodge(width = 0.2)) +  # Error bars with dodge
  stat_summary(fun = mean, geom = "point", size = 3, position = position_dodge(width =
0.2)) +  # Mean points with dodge
```

```
  coord_flip() +
  theme_classic() +
  labs(x = "",
       y = expression(paste("Forage Biomass (kg ", ha^-1, ")")),
       color = "Cover Type") +
  scale_x_discrete(labels=c("Cumberland Head","USDA 2", "USDA 1", "Lake Alice", "Ausabl
e Marsh")) +
  scale_color_manual(values = cols,
                     labels = c("Phragmites Marsh", "Submergent Marsh", "Scrub/Shrub",
"Emergent Marsh", "Moist-soil")) +
  theme(legend.position = "none")

##Plot of unweighted vegetation and invertebrate forage biomass between cover types acr
oss sites
inv_plot_bm + veg_plot_bm +
  plot_annotation(tag_levels = "A")
```