Assessing the Waterfowl Forage Value of Wetland Plant Communities

by

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Author’s declaration

I hereby declare that I am the sole author of this thesis. This is a true copy of the thesis, including any required final revisions, as accepted by my examiners.

I understand that my thesis may be made electronically available to the public.

Abstract

Invasive *Phragmites australis* continues to threaten wetland biodiversity and reduce food availability for waterfowl across North America. Assessing how vegetation responds to suppression efforts, and how these changes influence forage quality, is essential for informing management. In this study, I adapted the Vegetation Forage Quality Index (vFQI) for use in southern Ontario by developing updated forage value coefficients through expert elicitation. I also introduced a new tool, the Weighted Mean Waterfowl-forage Coefficient (WMWCs), modified to reduce the influence of species richness on the index score. I tested both indices in wetlands of the Long Point and Big Creek National Wildlife Areas, where *P. australis* suppression was conducted to evaluate the impact of suppression on waterfowl forage value. Field surveys (2022-2023) revealed that the WMWCs, but not the vFQI, successfully distinguished among invaded, treated, and reference sites, detecting improved forage quality in treated areas two years post-treatment. However, neither index was strongly correlated with empirical seed mass data from sediment cores, suggesting that seed biomass alone may not fully reflect foraging value. This study highlights the limitations of richness-sensitive indices like the vFQI and demonstrates the potential of WMWCs as a more robust indicator of wetland forage quality for waterfowl. Recommendations for future work include validation with bird observations and long-term monitoring to better capture temporal recovery dynamics.

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List of Abbreviations

CLM	Cumulative Link Model
CWMP	(Great Lakes) Coastal Wetland Monitoring Program
CWS	Canadian Wildlife Service
DED	Duck-Energy Days
DMD	Dry Matter Digestibility
DMRs	Daily Ration Models
DUD	Duck-Use Days
ECC	Energetic Carrying Capacity
FQA	Floristic Quality Assessment
FQI	Floristic Quality Index
MEC	Metabolizable Energy Coefficient
NWA	National Wildlife Area
SDMs	Spatial Depletion Models
TME	True Metabolizable Energy
vFQI	Vegetative Forage Quality Index
WMCCs	Weighted Mean Coefficient of Conservatism
WMWCs	Weighted Mean Waterfowl-forage Coefficient

1.0 General Introduction

Under the North American Waterfowl Management Plan (NAWMP) - an agreement established between Canada, the United States, and later Mexico to implement the objectives of the 1916 Convention for the Protection of Migratory Birds - North American nations committed to the conservation and sustainable management of migratory waterfowl populations and their habitats (NAWMP, 2018). In Canada, this international commitment is supported by national legislation such as the Canada Wildlife Act (1973) and the Migratory Birds Convention Act (1994) which provide the legal basis for protecting migratory birds and managing important habitats. These acts guide the work of the Canadian Wildlife Service (CWS), which is responsible for monitoring waterfowl populations and managing National Wildlife Areas (NWAs) (CWS, 1986; Government of Canada, 2025). A key part of the NWA mandate is to protect wetlands that are critical to waterfowl survival and productivity. Today, waterfowl face a range of growing threats, including habitat loss, climate change, agricultural intensification, and the spread of invasive species (Wilcox et al., 2003; Wong et al., 2011; Mitsch & Hernandez, 2013; Brasher et al., 2019). As a result, wetland managers encounter numerous challenges in maintaining and restoring wetland ecosystems to support the breeding, migration, and wintering needs of waterfowl, with particular attention to ensuring the availability of important food resources during critical periods of their annual cycle.

1.1 Waterfowl

‘Waterfowl’ is a broad term that typically refers to aquatic birds adapted to life in or around water, particularly members of the order *Anseriformes*, which includes ducks, geese, and swans

(Bellrose et al., 1980; Wayland, 1984). While loons, grebes, and other water-associated birds are sometimes informally grouped as waterfowl, they are taxonomically distinct and not typically included in conservation frameworks focused on *Anseriformes* (Baldassarre, 2014). Canada, particularly the province of Ontario, serves as breeding, migratory, and/or wintering habitat for approximately 30 of these species (Prince et al., 1992). Based on differences in morphology, foraging behavior, body size, buoyancy, and habitat use, they are commonly divided into several functional groups, such as dabbling ducks, diving ducks, geese and swans, and other specialized feeders (Wayland, 1984; OMNR, 2000).

Dabbling ducks typically forage in shallow water or mudflats, consuming seeds, vegetative components of aquatic plants, and invertebrates (Anderson et al., 2000; Monfils et al., 2014; Palumbo et al., 2019; Klimas et al., 2022a). They prefer shallow, calm environments such as riffles, sandbars, and secondary channels, commonly characterized by moderate water turbidity and higher levels of dissolved nitrogen, while avoiding areas with deeper open water and high forest cover (Savard Jean-Pierre et al., 1994; Johnson et al., 1996; Fleming et al., 2015; Masto et al., 2023).

In contrast, diving ducks select deeper and more open water bodies such as estuarine zones, where they feed by submerging to capture aquatic invertebrates, mollusks, and small fish (Prince et al., 1992; Anteau & Afton, 2009; Gross et al., 2020). They prefer habitats characterized by greater water depth, higher conductivity, and elevated calcium levels, while generally avoiding shallow, highly turbulent environments (Savard Jean-Pierre et al., 1994; Murkin et al., 1997; Taft et al., 2002; Torrence & Butler, 2006; Baschuk et al., 2012).

While geese and swans are primarily herbivorous grazers, relying on terrestrial vegetation, crops, or shallow aquatic plants (Bailey et al., 2008; Fox et al., 2017; Gehring et al., 2020), they also follow a capital breeding strategy, accumulating large energy reserves prior to reaching breeding areas (Arzel et al., 2006). Their predominantly herbivorous diet supports this energetic investment, although some species may also supplement their intake with local resources before nesting (Mitchell & Eichholz, 2020). The success of their migration and reproduction is closely tied to the quality and availability of food along migratory routes (Nolet, 2006).

Despite differences in foraging strategies and habitat preferences, all waterfowl species depend on wetland ecosystems that provide sufficient food resources to support critical stages of their life cycles. Plant-based foods represent a major component of waterfowl diets, as evidenced by a preliminary literature review of 13 peer-reviewed studies published between 1992 and 2022, which identified more than 70 plant taxa consumed by 18 waterfowl species (Appendix A). This reliance on plant material becomes particularly important during specific periods of the annual cycle, when energetic and nutritional demands fluctuate significantly.

1.2 Seasonal diet

Waterfowl diets shift seasonally in response to changing energetic and nutritional demands. During autumn and early winter, carbohydrate-rich foods such as seeds, grains, and tubers help birds accumulate fat reserves for migration and thermoregulation (Prince et al., 1992). These plant-based resources consistently form the majority of dabbling duck diets throughout these colder periods, with seeds of moist-soil and aquatic plants commonly dominating the digestive tract contents (Callicutt et al., 2011). Throughout the period of the autumn and early winter molt, some species of dabbling ducks, particularly non-breeding Green-winged Teal (*Anas carolinensis* J. F. Gmelin, 1789), increase their consumption of invertebrates to meet higher protein requirements for feather growth; however, plant seeds still represent the main component of their diet, comprising approximately 70% of total food intake (Anderson et al., 2000).

However, field observations conducted during the non-breeding period (autumn through winter) in western Tennessee, USA, further confirmed that other dabbling duck species predominantly rely on plant-based foods (Osborn et al., 2017). Mallard (*Anas platyrhynchos* Linnaeus, 1758) and Northern Pintail (*Anas acuta* Linnaeus, 1758) primarily consume seeds of aquatic and moist-soil plants with only minor contributions from invertebrates, while Gadwall (*Mareca strepera* Linnaeus, 1758) forage mainly on submersed aquatic vegetation early in the season before switching to seeds later in winter. Green-winged Teal, in contrast, initially depend more heavily on aquatic invertebrates before supplementing their diet with seeds.

As the breeding season approaches, many waterfowl species shift toward greater consumption of protein-rich invertebrates to support egg development. Dabbling ducks increase their intake of invertebrates during late winter and spring to meet protein demands for egg formation, although seeds continue to provide the primary carbohydrate base and still constitute a substantial proportion of the diet (Arzel et al., 2006). A study of 481 wetlands in the Upper Mississippi River and Great Lakes region, USA, during spring migration demonstrated that plant food biomass consistently exceeded invertebrate biomass across all habitat types, highlighting the ongoing importance of seeds as a key food resource for migrating ducks (Straub et al., 2012). Similarly, research conducted in the Rainwater Basin of Nebraska, USA, showed that although seed availability may be limited by the extent of surface water, seeds remain a more reliable food resource compared to invertebrates, which, despite their high protein value, exhibit greater variability in abundance across sites and seasons (Schepker et al., 2019).

This pattern is further supported by studies showing that while species such as Mallard, Blue-winged Teal (*Spatula discors* Linnaeus, 1766), and Gadwall increase their intake of invertebrates during spring migration to replenish protein reserves, they still maintain a diet dominated by plant seeds (Tidwell et al., 2013; Hitchcock et al., 2021). In contrast, Green-winged Teal do not exhibit a notable shift toward an animal-based diet during spring migration, continuing to rely primarily on plant seeds, which account for over 80% of their total food intake (Klimas et al., 2022a). The importance of seeds for Green-winged Teal is further emphasized by findings showing that for every 10% increase in the proportion of seeds in the diet, body lipid content increased by approximately 2.4%, suggesting that access to seed-rich habitats may play a critical role in enabling individuals to build sufficient energy reserves for continued migration and successful breeding (Klimas et al., 2020).

Compared to dabbling ducks, diving ducks shift spring diets to be more carnivorous. During spring migration, Lesser Scaup (*Aythya affinis* Eyton, 1838) increasingly select open-water habitats with low emergent vegetation and high turbidity, where invertebrates, particularly amphipods, are more available (Anteau & Afton, 2009). Nevertheless, seeds and other plant materials are still essential dietary components. Analysis of gut contents for Lesser Scaup and Ring-necked Duck (*Aythya collaris* Donovan, 1809) showed that the ratio of seeds to invertebrates during spring migration was 50% to 50%, respectively (Hitchcock et al., 2021). In the same way, Redheads (*Aythya americana* Eyton, 1838), although primarily reliant on seeds and aquatic vegetation, substantially increased their consumption of invertebrates during the pre-laying and laying periods to meet elevated protein demands associated with reproduction (Kenow & Rusch, 1996).

Recent studies of seasonal body mass dynamics in seven species of dabbling ducks and six species of diving ducks across three regions of California's Central Valley, USA, revealed that dabbling ducks typically experience greater body mass loss than diving ducks, particularly during mid-winter (Herzog et al., 2024). This pattern is attributed to a combination of reduced food availability and higher energetic demands associated with molting and early pair formation in dabbling ducks. Additionally, research on Mallards during spring and fall migration in the Illinois and Central Mississippi River Valleys, USA, showed that their use of wetlands was strongly tied to the extent of emergent vegetation, the presence of refuge areas, and wetland size, while open-water areas were used less during spring (Stafford et al., 2007). Together, these patterns place increasing responsibility on wetland managers to ensure seasonally appropriate resources are available to support waterfowl populations year-round.

1.3 Management strategy

To ensure the availability of seasonally appropriate and nutritionally sufficient resources for waterfowl populations, wetland managers employ a combination of active and passive management strategies (NRCS & USFWS, 2007). Active management involves direct interventions to stimulate the growth of food-producing vegetation and improve habitat conditions. Common active techniques include water-level manipulation (i.e., controlled flooding and drawdowns), disking, mowing, and targeted herbicide application (Gray et al., 2013). Passive management focuses on facilitating natural ecological processes with minimal intervention, such as supporting natural succession and maintaining buffer zones around wetlands to protect habitat quality and reduce external disturbances (EPA, 2005).

A considerable number of studies have compared seed productivity between wetlands under active and passive management. Overall, most research demonstrates that active management generally results in significantly greater seed biomass, providing more abundant food resources for migrating and wintering waterfowl. For example, at Chautauqua National Wildlife Refuge in Illinois, USA, actively-managed wetlands with moist-soil annual vegetation produced greater seed biomass (1454 kg/ha) compared to areas dominated by perennial vegetation (404 kg/ha) or willows (25 kg/ha), resulting in greater waterfowl carrying capacities (Bowyer et al., 2005). Similarly, a survey of 72 moist-soil units across Arkansas, Louisiana, Mississippi, and Missouri, USA, found that actively-managed wetlands produced, on average, 157 kg/ha more seeds compared to passively-managed wetlands, with early-successional grasses dominating active sites. Contrarily, passively-managed sites were often overgrown with vines and woody vegetation (Kross et al., 2008). In Illinois, USA, an assessment of 49 wetlands revealed that sites employing water-level drawdowns combined with soil disturbance produced approximately 235 kg/ha more moist-soil seed biomass than passively-managed sites (Stafford et al., 2011). In western Oregon and Washington, USA, actively-managed wetlands produced more than twice as much total seed biomass as passively-managed sites, while unmanaged areas had the lowest plant diversity and were dominated by invasive perennials plant species (Evans-Peters et al., 2012). Even when seed biomass does not differ significantly, active management can enhance habitat value. A multi-state study of 65 wetlands showed that managed sites supported up to three times more dabbling ducks than unmanaged ones, despite similar seed and tuber biomass (Tapp et al., 2018). Greater use of managed wetlands was likely due to better food accessibility, habitat structure, or invertebrate production (Osborn et al., 2017). Autumn mowing of moist-soil vegetation was found to increase waterbird use by improving early-season foraging conditions, even though seed and tuber biomass later equalized across treatments (Hagy & Kaminski, 2012b). Finally, a study of 123 wetlands in Ohio, USA, showed that although both management types provided sufficient food for waterfowl in autumn, energetic carrying capacities declined by over 80% by spring, highlighting the need to maintain resources through late winter and spring (Brasher et al., 2007).

Despite the general trend of higher seed production under active management, several studies have reported cases where differences between actively- and passively-managed wetlands were minimal. A study of 32 seasonal wetlands in Arkansas and Mississippi, USA, found no significant difference in total seed biomass between management types. Surprisingly, passively-managed sites outperformed actively-managed ones in Arkansas during a year of exceptionally high precipitation (Olmstead et al., 2013). Since results vary between studies, depending on region, habitat type, and environmental conditions, additional research is needed to fully understand the relative benefits and drawbacks of active and passive wetland management for waterfowl foraging.

1.4 Foraging assessment methods

Differences in habitat quality under various management strategies can be subtle and are not always captured by total seed biomass alone. This underscores the importance of using more comprehensive methods to evaluate wetland foraging conditions for waterfowl. Most recent studies, including those aforementioned, have employed a variety of methods to evaluate the foraging quality of wetland habitats for waterfowl. These methods can be systematically categorized into a three-tiered approach based on the level of detail and intensity of data collection.

The first tier involves the use of remote sensing technologies such as satellite imagery and aerial photography. These tools are widely recognized as essential for broad-scale habitat assessments, providing rapid and efficient insights into vegetation coverage patterns, flood conditions, and overall habitat suitability for waterfowl (Bly et al., 2010; Sahour et al., 2021). Remote sensing allows managers and researchers to monitor extensive landscapes and identify areas of potential habitat availability with minimal ground effort.

The second tier incorporates rapid field assessments or ecological indices to assess habitat quality for waterfowl. For example, vegetation-based assessments that rely on systematic field surveys using transects and/or quadrats to quantify plant species composition, coverage, and biomass - parameters that inform assessments of both habitat structure and forage availability (EPA, 2002; Gonsalves & Cartwright, 2016). Among the most commonly-used are indices based on the Floristic Quality Assessment (FQA) approach, which evaluate habitat condition based on the presence and abundance of plant species with varying tolerance to disturbance (Spyreas, 2019). The Floristic Quality Index (FQI), in particular, combines species richness and conservatism values (C-values) to estimate the ecological integrity of plant communities (Bourdaghs et al., 2006; DeBerry & Perry, 2015). By contrast, the Vegetative Forage Quality Index (vFQI) is an FQA-inspired but distinct index focused on forage value for waterfowl: it replaces conservatism C-values (0-10) with expert-derived forage coefficients (typically 1-4) and weights them by each taxon's proportional cover in the community (Fleming et al., 2012; Farley et al., 2022). Unlike the FQI, which focuses on ecological integrity, the vFQI emphasizes energetic value for waterfowl. By integrating species identity with nutritional value, such indices allow for a refined understanding of the energetic potential embedded in wetland plant communities.

The third tier focuses on intensive field data to directly quantify the use of food resources. Among these, a range of bioenergetic models, from simpler to more complex, are used to convert available forage biomass into estimates of potential energy carrying capacity for waterfowl (Straub, 2008; Brasher, 2010). These estimates are typically expressed in units such as Duck-Energy Days per hectare (DED/ha) (Reinecke et al., 1989; Brasher et al., 2007; Gray et al., 2009; Stafford et al., 2011; Osborn et al., 2017) and Duck-Use Days per hectare (DUD/ha) (Bowyer et al., 2005; King et al., 2006; Stafford et al., 2007; Fino et al., 2017), providing a standardized framework for evaluating habitat quality and informing management decisions. These model-based estimates are further complemented by field-based observations of waterfowl behavior, including feeding, resting, and nesting patterns, which offer additional insights into how birds use available resources across different habitats (Hagy & Kaminski, 2012b; Tapp et al., 2018; Lindstrom et al., 2020; McDuie et al., 2021).

Alternatively, or in parallel within this tier, methods such as sediment sampling for seeds and invertebrate sampling for protein-rich prey provide precise, quantitative data on the actual availability of nutritional resources for waterfowl (Reinecke & Hartke, 2005; Brasher et al., 2007; Schepker et al., 2019). These direct measurements of seed biomass, for example, often form the numerator of DED calculations, linking food availability to energetic carrying capacity. Additional methods, including stomach content analysis, offer direct evidence of dietary intake and can be used to validate habitat-based predictions (Walley, 2016). In some cases, reproductive success metrics such as nest density, clutch size, or hatch rates are also evaluated as ultimate indicators of habitat suitability (Arnold et al., 2007; Howerter et al., 2014; Devries et al., 2023). Alternatively, DUD estimates can be grounded in direct counts of waterfowl using a site, such as aerial- or ground-based surveys that measure actual bird presence and duration of stay over time

(Bowyer et al., 2005). These observations provide a direct link between habitat characteristics and bird use patterns (Brand et al., 2014). Together, these field-based approaches are critical for detailed habitat evaluations where understanding the actual food base and its consequences for waterfowl fitness is essential to inform effective management.

While first-tier methods, such as remote sensing and aerial imagery, provide efficient and broad-scale evaluations of wetland habitat conditions, they may overlook finer-scale variations in forage quality due to their generalized nature. In contrast, third-tier methods, including direct quantification of seeds through sediment sampling, offer highly detailed and ground-truthed assessments of food resource availability but are labour-intensive and time-consuming, particularly during data collection and processing. Given these trade-offs, the focus of this research is placed on second-tier approaches, specifically the use of vegetation-based indices such as the Vegetative Forage Quality Index (vFQI), to support efficient and scalable evaluations of habitat quality for waterfowl.

1.5 Research objectives

In my first data chapter, I modify the vFQI index for southern Ontario by collaborating with experts to derive regionally-calibrated forage quality coefficients that reflect the most abundant wetland plant species in the region. In the second data chapter, I apply the newly calibrated vFQI tool to evaluate the impacts of invasive Common Reed (*Phragmites australis* ssp. *australis*) suppression efforts on waterfowl forage quality within the NWAs of the Long Point World Biosphere Reserve. This analysis is based on plant biodiversity and sediment core data collected in Long Point and Big Creek NWAs in 2022 and 2023.

2.0 Adapting the Vegetative Forage Quality Index (vFQI) to southern Ontario

2.1 Introduction

In this chapter, I first step back to situate the vFQI within the broader practice of ecosystem assessment and the role of expert judgement. I outline how forage and habitat quality are evaluated across other management systems, noting when expert input is used. I then return to waterfowl and review approaches for assessing food resources, focusing on bioenergetic models and habitat indices. Finally, I present the vFQI adaptation for southern Ontario, describe how the expert-derived coefficients were developed and analysed, and evaluate their correspondence with independent nutritional measures.

2.1.1 Cross-system context for expert-based habitat and forage indices

Across wildlife and ecosystem management, a common strategy links three elements: the energy demand of focal organisms, the energy supply provided by habitats, and the spatio-temporal matching of demand and supply (Kearney & Porter, 2009; Millspaugh & Thompson, 2009). Within this strategy, two complementary perspectives are commonly applied: a demand-based view that asks "what does the organism need?" and a supply-based view that asks "what does the habitat provide?" From a demand-based perspective, managers estimate species' energetic requirements for maintenance, thermoregulation, movement, and reproduction, and translate them into daily or seasonal energy budgets (e.g., inputs to bioenergetic carrying-capacity metrics) (Lin & Chang, 2022). From a supply-based perspective, they monitor the quantity, quality, and accessibility of forage or prey through field sampling and laboratory assays (Winkler et al., 2021). Together, these data streams are integrated into indices or models of carrying capacity and habitat suitability (Del Monte-Luna et al., 2004; Chapman & Byron, 2018). Expert elicitation complements both perspectives when data are incomplete or context-dependent, to score taxa, weight criteria, or set decision thresholds (Roche & Campagne, 2019).

In livestock systems, forage quality is assessed by integrating laboratory metrics (e.g., crude protein, neutral and acid detergent fiber, lignin, total digestible nutrients, digestible/metabolizable/net energy, and in vitro/in situ digestibility) with field-based measures of availability (e.g., biomass, utilization, residual dry matter) and carrying capacity expressed in Animal Units and Animal Unit Months (Roginski et al., 2003; Lynn et al., 2020). From a demand-based perspective, Animal Units and Animal Unit Months translate herd energy and dry-matter requirements into standardized units that guide stocking rates and grazing plans. From a supply-based perspective, laboratory metrics and field measurements quantify what pastures can provide in terms of quality and quantity. Composite indices such as Relative Feed Value and Relative Forage Quality translate forage laboratory metrics (e.g., acid detergent fibre, neutral detergent fibre, neutral detergent fibre digestibility) into expected digestible dry matter and intake, thereby bridging supply and demand with an animal-referenced ranking of hay and pasture quality (Moore & Undersander, 2002). Meanwhile, qualitative scorecards and expert-based ecosystem assessments aggregate ordinal ratings into management-ready categories and rely on trained observer judgement to assign indicator scores using standardized descriptors (Toledo et al., 2016). For example, the Pasture Condition Score is a field scorecard used to rapidly evaluate pasture condition. Trained observers visually rate 10 indicators on 1–5 scales using standardized descriptions; indicator scores are summed into an overall pasture condition score to guide management actions and track change over time (USDA, 2020). Although qualitative, the protocol encourages supplemental measurements (e.g., clipping for biomass) to reduce subjectivity, and expert judgment is central to assigning each indicator's category (Sanderson, 2014).

Beyond livestock systems, wildlife managers working with free-ranging ungulates (e.g., deer, elk, moose, bison) deploy similar demand- and supply-oriented approaches. In ungulate management, forage quality and availability are assessed with vegetation sampling, laboratory proxies of diet quality, non-invasive diet metrics such as faecal nitrogen, movement analyses using Resource Selection Functions and Step-Selection Functions, and, in some cases, Nutritional Carrying-Capacity models that combine animal requirements with forage supply (Demarais et al., 2012; Krausman & Bleich, 2013; Van Beeck Calkoen et al., 2020). Faecal nitrogen is widely used as an index of dietary quality in free-ranging ruminants (Monteith et al., 2014), while Resource Selection Functions and Step-Selection Functions quantify how animals select habitats relative to availability at seasonal scales (Signer et al., 2019). Expert elicitation plays a role when local data are sparse, for example, when classifying plant palatability, adapting habitat-suitability curves, or setting initial diet constraints in Nutritional Carrying-Capacity that are later calibrated with monitoring (USGS, 2025). Habitat Suitability Index models for cervids express habitat quality for a target species or life stage on a 0-1 scale (DFO, 2015); suitability curves for variables such as cover, forage accessibility, snow, and disturbance are developed from field-use or preference data or professional judgement, and when site data are lacking practitioners use expert opinion or adapted curves (Anstedt, 2016; Sable et al., 2019). When site data are limited, the shape, thresholds, and weights of suitability curves are adapted from literature and professional judgement (Roberts, 2004; Lindquist et al., 2021). Classic Nutritional Carrying-Capacity approaches place explicit nutritional constraints (digestible energy and digestible protein) on diets assembled from available forages to estimate seasonal carrying capacity for elk or deer (Beck et al., 2006; Rowland et al., 2018). When site-specific forage chemistry, intake rates, or accessibility data are limited, practitioners initialize diets and constraints using expert knowledge and literature values, then refine them with field data (e.g., faecal nitrogen, body condition, demographics) as they accumulate (Lopez et al., 2025; USGS, 2025).

Similar principles also appear in fisheries and aquatic conservation: supply is quantified via prey-base surveys and laboratory estimates of prey energy density, while demand is represented with bioenergetics models that link consumption, temperature, and growth; habitat-suitability frameworks translate environmental variables into standardised scores (Jorgensen et al., 2016; Deslauriers et al., 2017). In practice, Habitat Suitability Index models express habitat quality (Hightower et al., 2012; DFO, 2015; Wegscheider et al., 2024) and are often paired with the Habitat Assessment Tool, which links habitat change to population response and can embed Habitat Suitability Index outputs (Tymoshuk et al., 2017); both rely on documented assumptions and, where needed, locally adapted suitability curves informed by expert judgement (DFO, 2019). Expert judgement is used alongside monitoring to shape suitability curves, weight criteria, and initialise diet matrices and parameters, with assumptions documented and updated as new data become available (DeMaster et al., 2004; Crawford et al., 2020).

Taken together, this cross-system overview shows that expert elicitation is widely used alongside field measurements to structure and interpret ecosystem assessments. Across livestock, wild ungulate, and fisheries applications, expert input helps translate partial data into management-ready information. Waterfowl management faces many of the same constraints (heterogeneous habitats, limited forage-chemistry and intake data, strong seasonal change), yet expert-based vegetation indices have seen more limited use; the Vegetative Forage Quality Index (vFQI) is the main example, and its application has been constrained by gaps in regional forage coefficients. In this chapter I address that gap by developing southern Ontario coefficients through a structured expert elicitation with confidence ratings and by comparing the resulting values with independent nutritional evidence. With this motivation in place, I now narrow the focus to waterfowl and review the two approaches used to evaluate their food resources: bioenergetic models and vegetation-based indices, beginning with bioenergetic models.

2.1.2 Assessing food resources for waterfowl: bioenergetic models

Effective management and conservation of waterfowl habitats require objective, standardized methods to assess habitat conditions, including their food value. Two complementary tools are widely used: bioenergetic models that estimate Energetic Carrying Capacity (ECC) and ecological indices that summarize vegetation-based forage quality (including the vFQI). I begin with bioenergetic models.

Bioenergetic models are conceptual approaches designed to estimate the ECC of a habitat - that is, the amount of energy (in kilocalories) a habitat can provide to support waterfowl during a specific period of time (Straub, 2008; Brasher, 2010). These models are commonly used in conservation planning to determine whether available habitats can meet the energetic needs of target bird populations, especially during migration or wintering seasons.

Brasher (2010) categorized ECC models into two main types: Daily Ration Models (DRMs) and Spatial Depletion Models (SDMs). DRMs are relatively simple and calculate ECC by dividing the total available energy within a habitat by the daily energy requirement of a single bird. This approach produces estimates in standardized units such as Duck Energy Days (DED), representing the amount of energy required to sustain one duck for one day (Reinecke et al., 1989; Brasher, 2010; Osborn et al., 2017; Livolsi et al., 2021), or Duck Use Days (DUD), which quantify the observed or predicted number of duck-days a habitat supports over time (Bowyer et al., 2005; King et al., 2006; Stafford et al., 2007; Fino et al., 2017). SDMs, in contrast, are more complex and incorporate spatial factors such as water depth, patch isolation, disturbance, and non-linear relationships between energetic supply and actual use, aiming for a more ecologically grounded representation of habitat capacity (Straub, 2008; Brasher, 2010; Boudreau et al., 2024).

Despite their widespread application, several limitations of ECC models have been identified in recent research. For example, a study conducted in the Mississippi Alluvial Valley showed that dabbling ducks often remained in feeding areas even after food availability dropped below predicted energetic thresholds, suggesting that models based solely on caloric estimates may fail to capture behavioral flexibility, food accessibility, seasonal variation, or disturbance (Hagy & Kaminski, 2015). Similarly, Beatty et al. (2015) conducted an empirical evaluation in the central and eastern United States showed that ECC models effectively predicted Mallard space use in only one of five seasonal scenarios; in most cases, simpler models performed equally well or better, especially under winter conditions when factors like ice cover, hunting pressure, and landscape structure became dominant. Additionally, a study by Martin et al. (2022) comparing different rapid monitoring methods for estimating moist-soil seed production in the Mississippi Alluvial Valley highlighted critical limitations of ECC models, including coarse resolution of seed estimates, difficulty in capturing interannual variability, and over-reliance on broad habitat classifications rather than species-specific or patch-level measurements. These findings highlight the need for integrating ecological realism and behavioral context into energetic models to improve their reliability in management planning.

2.1.3 Assessing food resources for waterfowl: vegetation-based indices

As an alternative, and often a complement to bioenergetic models, vegetation-based indices offer a standardized approach to evaluating the forage quality and ecological condition of wetland plant communities for waterfowl. These indices can function independently or be integrated into energetic models by informing estimates of plant composition, food availability, and habitat quality. Vegetation-based indices can generally be divided into two types: those that reflect the direct energetic value of plant species for waterfowl, and those based on the Floristic Quality Assessment (FQA) approach, which emphasizes ecological integrity through expert-derived coefficients (Dugger et al., 2007; Spyreas, 2019).

Specific examples of the direct energetic method include the True Metabolizable Energy (TME), Metabolizable Energy Coefficient (MEC), and Dry Matter Digestibility (DMD). One of the most precise energetic indices is the TME index, which estimates the actual amount of energy a bird derives from consuming seeds of a particular plant species (Checkett et al., 2002). TME is estimated through feeding trials in which a bird is fasted, fed a known amount of test food, and monitored (Dugger et al., 2007). Excreta are collected and analyzed to calculate the net energy absorbed, adjusting for baseline losses measured during a separate fasting trial (Lancaster et al., 2019). The TME value is then derived by subtracting the excreta energy from the gross energy of the ingested food and dividing by the food mass. Even though the index is considered accurate, there is a limited number of values of this index (Appendix B) due to the significant complexity of its calculation associated with the difficulty of catching birds, collecting and analyzing their excrement and energy absorbed due to vital activity (Gross et al., 2020). It is also worth noting that, for a single plant species, the value of this index can differ among bird species, such that extensive testing is needed to determine TME values for a native plant community as forage for a diverse waterfowl community (Brasher et al., 2007). Other digestion-based indicators, such as the MEC and DMD, have been used in laboratory settings to characterize the nutritional value of plant material (Joyner et al., 1987; Van Tets & Sanson, 1996). However, these indices are rarely applied in ecological research on waterfowl, and their values are limited to a few studies.

The alternative vegetation-based index method is based on the Floristic Quality Assessment (FQA) approach, which is designed to evaluate the ecological value or naturalness of a site by calculating the mean conservatism coefficient (C-value) of the plant species present (Spyreas, 2019). These indices allow researchers to draw conclusions about habitat conditions, even when empirical data are lacking, by relying on expert-assigned values that reflect each species' tolerance to disturbance and fidelity to natural habitats. Originally, FQA-based indices were developed to assess the ecological condition of a site by evaluating how natural or disturbed the plant community is (Swink & Wilhelm, 1994). The most commonly used index is the Floristic Quality Index (FQI), which combines the mean C-value with species richness to estimate overall plant community quality (Bourdaghs et al., 2006; DeBerry & Perry, 2015). The coefficient of conservatism is a value from 0 to 10 that reflects a plant taxon's tolerance to disturbance and its association with natural habitats (Spyreas, 2019). Higher values represent taxa that are more sensitive to disturbance and typically found in undisturbed, high-quality natural areas, whereas lower values represent taxa that are more robust. In FQI frameworks, the C-value is an expert-assigned 0–10 score that captures a native taxon's sensitivity to disturbance and fidelity to natural habitats; non-native taxa are typically scored 0 or excluded from the calculation, depending on regional protocols (Spyreas, 2019). Based on the FQI approach, the Vegetation Forage Quality Index (vFQI) was developed as a rapid assessment tool to estimate the forage value of plant communities for waterfowl in the Mississippi region (Fleming et al., 2012) and was later adapted for use in restored wetlands of the Montezuma Wetlands Complex, New York, to evaluate the outcomes of water level management on habitat quality for migratory birds (Farley et al., 2022). In contrast to FQI, the vFQI uses expert-derived forage-value C-coefficients for each plant taxon, that is, scores representing its value as waterfowl forage. Farley et al. (2022) used a 1-4 scale (1 = poor, 2 = fair, 3 = good, 4 = excellent), whereas Fleming et al. (2012) used a 10-30 scale with an "unknown" option (Table 2-1). The vFQI is calculated by weighting each plant species' forage coefficient by its proportional occurrence in the plant community, then adjusting the result by the total plant richness. This index assumes that plant species with higher forage coefficients produce more desirable and energy-rich food items for waterfowl. Therefore, I expect that species assigned higher vFQI values will also exhibit traits that increase energetic intake per unit of foraging effort. These may include greater seed production per unit area, larger seed mass, higher seed energy density (e.g., fat or protein content), presence of underground storage organs such as tubers, higher seedbank density, and longer seedbank persistence. If the vFQI scores are ecologically meaningful, they should show positive association with these traits, which are directly or indirectly related to the energetic value of wetland plant species.

In the original study by Flemming et al. (2012), a comparison of three wetland management strategies in the Mississippi Alluvial Valley across 54 sites using vFQI demonstrated significant differences in the forage quality of plant communities for waterfowl. Actively managed wetlands with early drawdown consistently exhibited higher vFQI values, greater plant species richness, and a higher proportion of annual grasses compared to late-drawdown or passively managed wetlands. The index was also negatively associated with perennial grass cover, reflecting its sensitivity to vegetation composition relevant for food availability and forage quality.

A subsequent study in the Mississippi Alluvial Valley evaluated how different wetland management strategies influence wintering waterfowl by comparing actively- and passively-managed private lands (Fleming et al., 2015). The vFQI was positively associated with duck density under late drawdown conditions, suggesting its potential utility as a predictor of habitat use when hydrology is favorable, although this pattern did not hold for early-drawdown or passively-managed sites. Across all treatments, vFQI-based models explained up to 27% of the variation in waterfowl density. The index also correlated positively with plant species richness and the proportion of energetically valuable species, while exhibiting a negative relationship with woody vegetation cover.

A study evaluating the restoration of 47 wetlands in northern New York employed the vFQI to assess the recovery of plant community forage value for waterfowl (Benson et al., 2019). By comparing these restored sites to 18 reference wetlands, the researchers found that vFQI scores, species richness, and the proportional cover of forage-relevant taxa were largely comparable, indicating successful restoration of energetic habitat quality. Moreover, the application of vFQI revealed a negative relationship between invasive plant cover and forage value, particularly in submerged and emergent zones, highlighting the importance of invasive species control in efforts to sustain high-quality foraging habitats for waterfowl.

In a study of the effects of water level management on vegetation and bird use of wetlands in the Montezuma Wetlands Complex, New York, Farley et al. (2022) used vFQI supplemented with updated forage coefficients for native plant species. They found that in the fall, vFQI values were significantly higher in fully and partially drained wetlands compared to passively-managed sites. Over three years, the researchers also collected more than 300 sediment cores annually (6.75 cm in diameter, 10 cm deep) from evenly distributed points across wetland types. After processing, they determined the dry weight of seeds and calculated their density in kilograms per hectare. Seed content analysis showed a similar pattern: passively-managed wetlands had, on average, half the seed and tuber density compared to fully- or partially-drained wetlands. Although this parallel trend provides support for the vFQI, the direct correlation between seed or tuber biomass and vFQI values was not tested (Farley et al., 2022). Therefore, the authors recommend further validation through studies that directly compare plant community vFQI scores with seed and tuber availability, especially as new forage coefficients are added.

We used the vFQI during field surveys conducted in August 2022 and 2023 in two National Wildlife Areas (NWAs): the Big Creek NWA and the Long Point NWA, located within Ontario's Long Point and Walsingham Forest Priority Place. These surveys were carried out in collaboration with the Canadian Wildlife Service to evaluate whether the forage value of wetland vegetation for waterfowl was recovering following the suppression of invasive Common Reed (*Phragmites australis* ssp. *australis*; hereafter *P. australis*). The Integrated Conservation Action Plan (ECCC, 2018) for this Priority Place set a goal of suppressing *P. australis*, aiming for 90% native vegetation coverage in coastal wetland, beach, and dune ecosystems by 2025. During this work, we found that the main impediment to using the vFQI to assess wetlands in southern Ontario is the lack of regionally-derived forage quality coefficients for native plant species. Only about 50% of the observed taxa had published coefficients available for vFQI calculation (Appendix C), which limits the tool's applicability in regional management contexts.

To address this gap, the objectives in this chapter are twofold: firstly, to calibrate the vFQI for southern Ontario by working with experts to derive forage quality coefficients for native plant species that are common to coastal marsh in southern Ontario; and secondly, to compare the newly-derived coefficients with existing true metabolizable energy (TME) values and functional trait data for native plant species. In this study, I additionally incorporate expert confidence ratings as part of the elicitation process, allowing for the quantification of uncertainty and improving the robustness of the derived coefficients (O'Hagan, 2019). This approach provides a more transparent estimate of expert agreement and reliability, representing a novel application in the context of forage quality assessments for waterfowl and vFQI development. I hypothesize that species assigned higher vFQI coefficients will exhibit characteristics linked to greater energetic returns for waterfowl. Because the expert-derived vFQI coefficients reflect a multidimensional concept (e.g. nutritional quality, availability, and accessibility) of each plant taxon, I aim to test whether they show any concordance with a single-axis energetic metric. Specifically, I will evaluate whether taxa with higher published True Metabolizable Energy (TME) values tend to receive higher vFQI coefficients. Such alignment would suggest that vFQI coefficients capture real energetic value observed through direct methods. I also expect that vFQI coefficients will be positively associated with specific plant traits known to influence food quality for waterfowl. Confirming such an agreement would strengthen the ecological credibility of vFQI and highlight its usefulness as a practical alternative to more time- and resource-intensive energetic evaluations.

2.2 Methods

2.2.1 Selection of plant taxa for evaluation

To develop the expert survey, I assembled a list of 74 plant taxa representative of wetland vegetation in Ecoregion 7E of Ontario (Crins et al., 2009). The selection was based on three complementary sources (Appendix D): 72 taxa were identified through analysis of vegetation data from 46 wetland sites surveyed under the Great Lakes Coastal Wetland Monitoring Program (CWMP); 29 taxa lacking published forage value coefficients were added based on field observations conducted by the Waterloo Wetland Laboratory in the Long Point and Big Creek NWAs during August 2022 and 2023; 30 taxa were included from previous vFQI studies (i.e., Fleming et al., 2012; Farley et al., 2022) to allow comparison with coefficient values that have been previously derived. Specifically, 15 taxa had coefficients in both Fleming and Farley, six taxa were assessed only by Fleming et al. (2012), and nine taxa were scored only by Farley et al. (2022). Due to overlap among sources, the final curated list includes 74 unique taxa selected for their regional relevance and potential nutritional value for waterfowl.

2.2.2 Expert committee and assessment

I followed the same expert elicitation approach used in previous vFQI studies (Fleming et al., 2012; Farley et al., 2022) to develop forage value C coefficients for wetland plant taxa in southern Ontario (Table 2-1). Experts were identified through governmental and non-governmental organizations, as well as through authors of relevant peer-reviewed studies. Of 38 identified experts, 29 responded and 22 completed the survey, resulting in a 58% participation rate (Appendix E). All participants held a MSc or PhD degree and/or had over five years of experience in the field. The survey was conducted via Google Forms and included 74 previously-identified plant taxa. To reduce survey length, species were grouped at the genus level where appropriate, based on similarities in traits such as seed production, tuber presence, and growth form (Appendix F).

Experts scored each taxon's forage value on a four-point scale (1 - poor, 4 - excellent), considering nutritional quality, availability, and accessibility to waterfowl. A separate confidence rating was also provided for each score (1 - not confident, 4 - completely confident) to represent the certainty each expert held in regard to their forage value score for each plant. Optional comments were solicited, and links to the USDA Plants Database (USDA & NRCS, 2024) were included to assist evaluations without biasing experts through selective prevision of plant morphology or trait details. It was important not to bias the scoring of experts with trait-based information, given objective two of my analysis was to evaluate whether coefficients were related to plant traits. Final forage value coefficients were calculated as the median of expert scores. This decision reflects the fact that the 1-4 scoring system used in the survey represents a Likert-type ordinal scale, where differences between adjacent categories are not necessarily equal (Cliff, 1996).

Table 2-1. Comparison of evaluation parameters and calculation of the C coefficients for vFQI.

Study	**Fleming et al., 2012**	**Farley et al., 2022**	**Kramarenko & Rooney**	
Location	Mississippi, USA	New York, USA	Ontario, Canada	
№ of taxa	135	89	74	
№ of experts	14	7	22	
Assessment method	Rating	Rating	Rating	
Rating	Forage value	Forage value	Forage value	Score confidence
	30 - excellent 20 - average 10 - poor UK - unknown	4 - excellent 3 - good 2 - fair 1 - poor	4 - excellent 3 - good 2 - fair 1 - poor	4 - completely confident 3 - fairly confident 2 - slightly confident 1 - not confident
Calculation of the C-value	Average ranked value	Average ranked value	Median	

2.2.3 True Metabolizable Energy values

I conducted a literature search in the Web of Science database (*Web of Science*, 2025) to identify peer-reviewed studies reporting TME values for wetland plants consumed by waterfowl. This search resulted in a compilation of TME values for 29 plant taxa based on feeding trials involving six waterfowl species, drawn from six studies published between 2002 and 2024 (Appendix B). I specifically selected studies where TME values were directly measured through metabolism trials with waterfowl, rather than estimated using functional plant traits indicative of potential nutritional value. Other direct energetic indices, such as DMD and MEC, were excluded from analysis due to the lack of sufficient published values. For the purpose of comparing TME values to vFQI plant coefficients, I calculated the mean TME value for each plant taxon by averaging all available TME measurements across different waterfowl species. This approach was used to reduce species-specific variation and provide a general estimate of the plant's energetic forage value.

2.2.4 Plant functional traits data

For the purpose of comparing plant traits to the vFQI plant coefficients, I extracted trait data relevant to waterfowl forage value from the TRY database. The TRY database is an online comprehensive global repository of plant functional traits compiled from multiple contributors, including researchers and ecological databases (Kattge et al., 2011). It serves as a valuable resource for ecological and evolutionary studies by providing standardized trait data for plant species worldwide. From the TRY database, I obtained data on 21 functional traits for 119 plant species (Appendix G).

2.2.5 Statistical analysis

I conducted all statistical analyses conducted using R version 4.3.1 (R Core Team, 2023) within the RStudio environment version 2024.12.1 (Posit Software, PBC, 2024). I imported and prepared data using the *readr* package for efficient CSV file reading (Wickham et al., 2024) and used the *tidyr* package for data reshaping and formatting of long labels (Wickham et al., 2024). Finally, I used the *dplyr* package (Wickham et al., 2023) to process and summarize the data.

Since my coefficient values were calculated as medians of expert scores, they represent ordinal data. Therefore, I used cumulative link models (CLMs) for the main statistical analysis, as this method is more appropriate for ordinal outcomes and continuous predictors (Christensen, 2019). I performed the CLMs using the *clm()* function from the *ordinal* package (Christensen, 2024). I used the broom package to organize model outputs, including coefficients, standard errors, z-values, and confidence intervals into data frames to allow further tabulation and export (Robinson et al., 2025). For visualization, I used *ggplot2* to produce publication-quality bar plots and predicted probability plots based on the CLM results (Wickham et al., 2025). I used *showtext* for font rendering in plots, ensuring consistent typography across devices (Qiu & Raggett, 2024). I formatted axis scales using *scales* for clean numeric labeling (Wickham et al., 2025) and combined or annotated multiple plots with *ggpubr* for consistent layout (Kassambara, 2023). Long trait names were adjusted using *stringr* to improve readability (Wickham, 2023).

To maintain consistency with previous studies (i.e., Fleming et al., 2012; Farley et al., 2022) that used Spearman's rank correlation to examine relationships between forage value coefficients, I additionally calculated Spearman's correlation coefficients using the *cor.test* function in base R (Appendix H). I performed linear regression analyses using the *lm()* function from the *stats* package in base R.

2.3 Results

2.3.1 Expert scores

I calculated the C coefficient for each of the 74 plant taxa as the median of expert-assigned forage value scores (Figure 2-1, Appendix I).

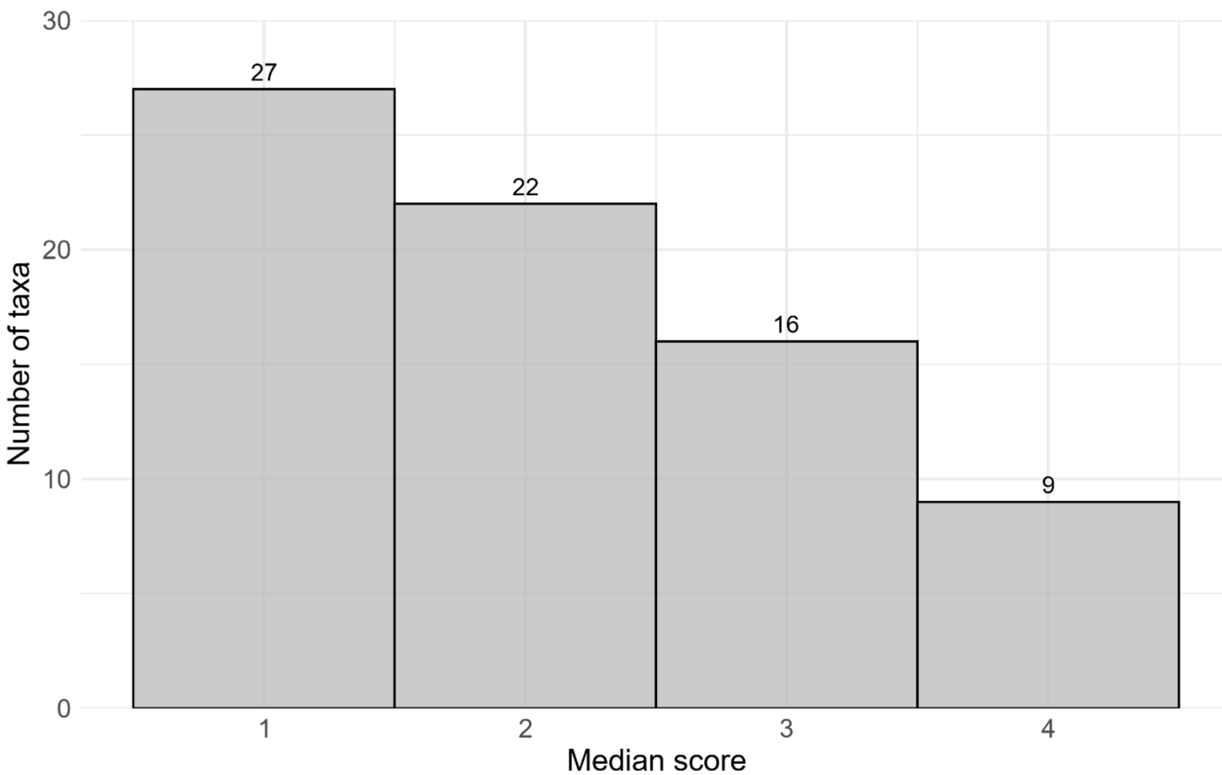

Figure 2-1. Distribution of plant median scores (n = 74), as determined by experts via the Google Form survey.

The histogram illustrates the distribution of plant median scores across four groups (1, 2, 3, and 4). For the purpose of this analysis, four plant taxa with a median score of 1.5 were grouped with those having a median score of 2, while two plant taxa with a median score of 3.5 were grouped with those having a median score of 4. I found a positive correlation between expert confidence and assigned forage value scores (Spearman's $\rho = 0.95$, $p = 0.051$), suggesting that higher confidence was generally associated with higher ratings (Figure 2-2).

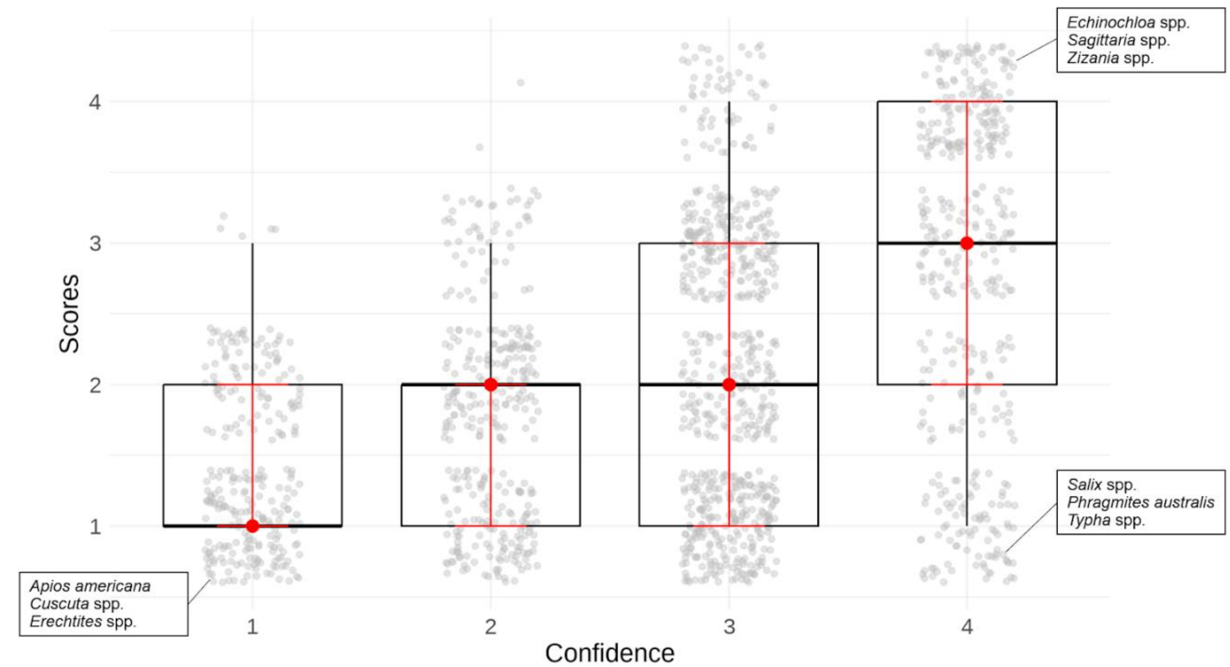

Figure 2-2. Distribution of expert-assigned forage value scores by confidence level. Red points represent the median score for each level, grey points are jittered within each level to reduce overlap and show the full distribution, black boxes denote the interquartile range (IQR) with the horizontal line marking the median, and whiskers extend to the most extreme values within 1.5 × IQR. Representative plant taxa are labeled to illustrate extreme combinations.

I analyzed the variance of expert-assigned scores across confidence levels using linear regression. The results revealed a strong positive agreement ($R^2 = 0.985$, $F = 130.91$, $p = 0.008$), indicating that score variance increased with confidence level (Figure 2-3).

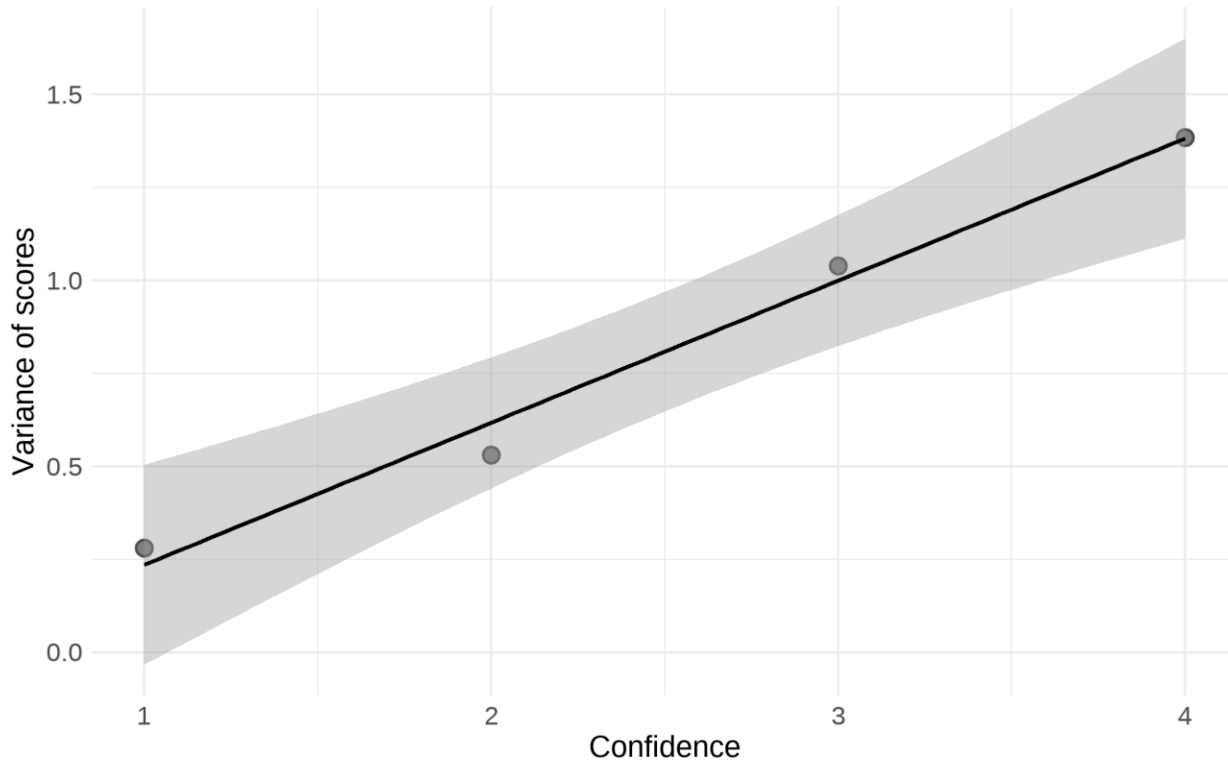

Figure 2-3. Variance of expert-assigned scores across confidence levels. Points represent variance estimates; the black line shows the linear regression fit with a 95% confidence interval ($R^2 = 0.985$, $F = 130.91$, $p = 0.008$)

2.3.2 Agreement with published coefficients

The ordinal regression model using values derived by Farley et al. (2022) as a predictor demonstrated a statistically significant positive association with the forage scores derived in this study. The model estimated a coefficient of $\beta = 7.43$ (±2.71 SE), with a z-value of 2.74, $p = 0.006$, and a 95% confidence interval ranging from 3.63 to 14.95 (Figure 2-4). These results indicate that higher scores derived by Farley et al. (2022) are positively associated with higher expert-assigned forage ratings obtained through my survey.

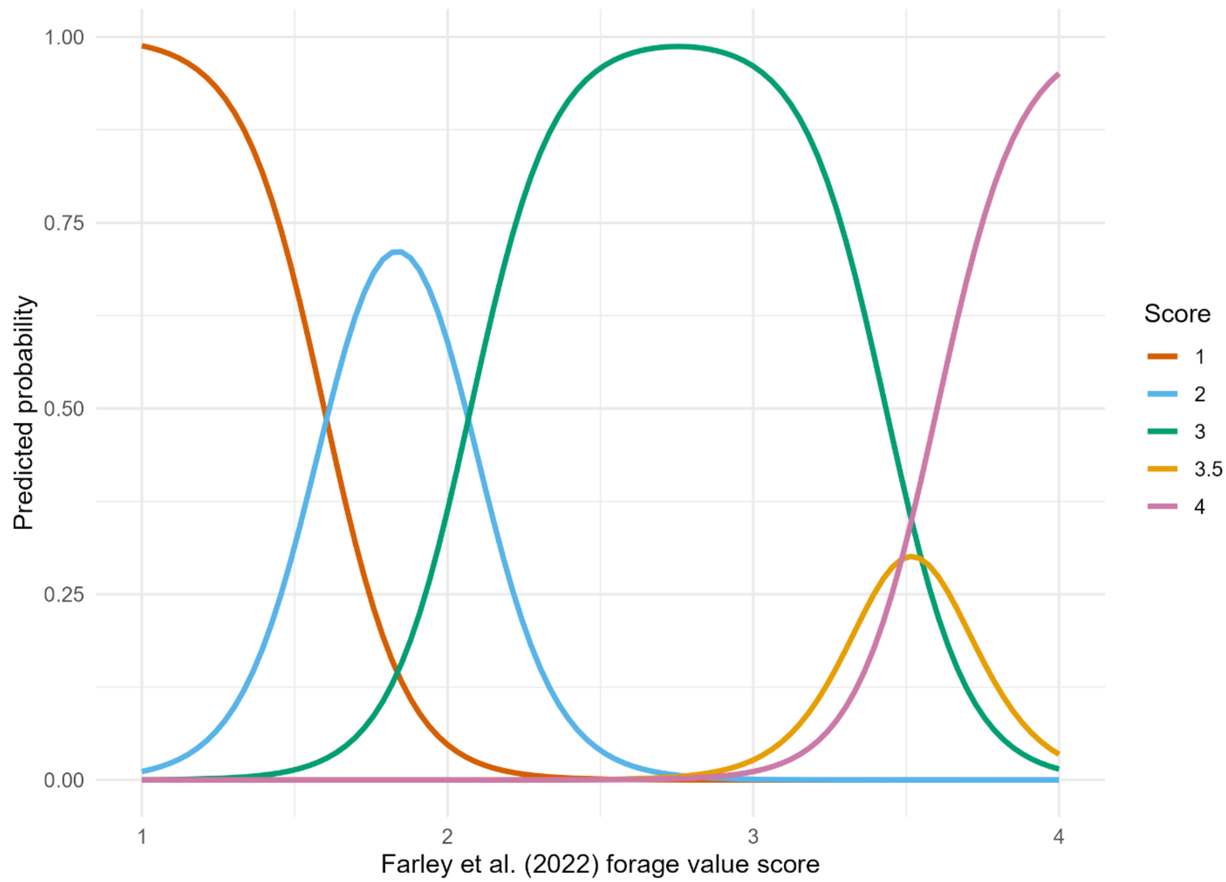

Figure 2-4. Predicted probabilities of expert-assigned forage scores based on Farley et al. (2022) using an ordinal regression model.

In contrast, the model using values derived by Fleming et al. (2012) as a predictor showed a positive but non-significant association ($\beta = 18.23 \pm 10.69$ SE), with a z-value of 1.71, $p = 0.088$, and a 95% confidence interval from 6.15 to 54.47 (Figure 2-5). The direction of effect aligns with expectations, and although $p = 0.088$ exceeds the conventional 0.05 threshold, it is suggestive of concordance; the wide interval and reduced dynamic range of Fleming's three-class scale (Table 2-1) likely contributed to the weaker statistical support.

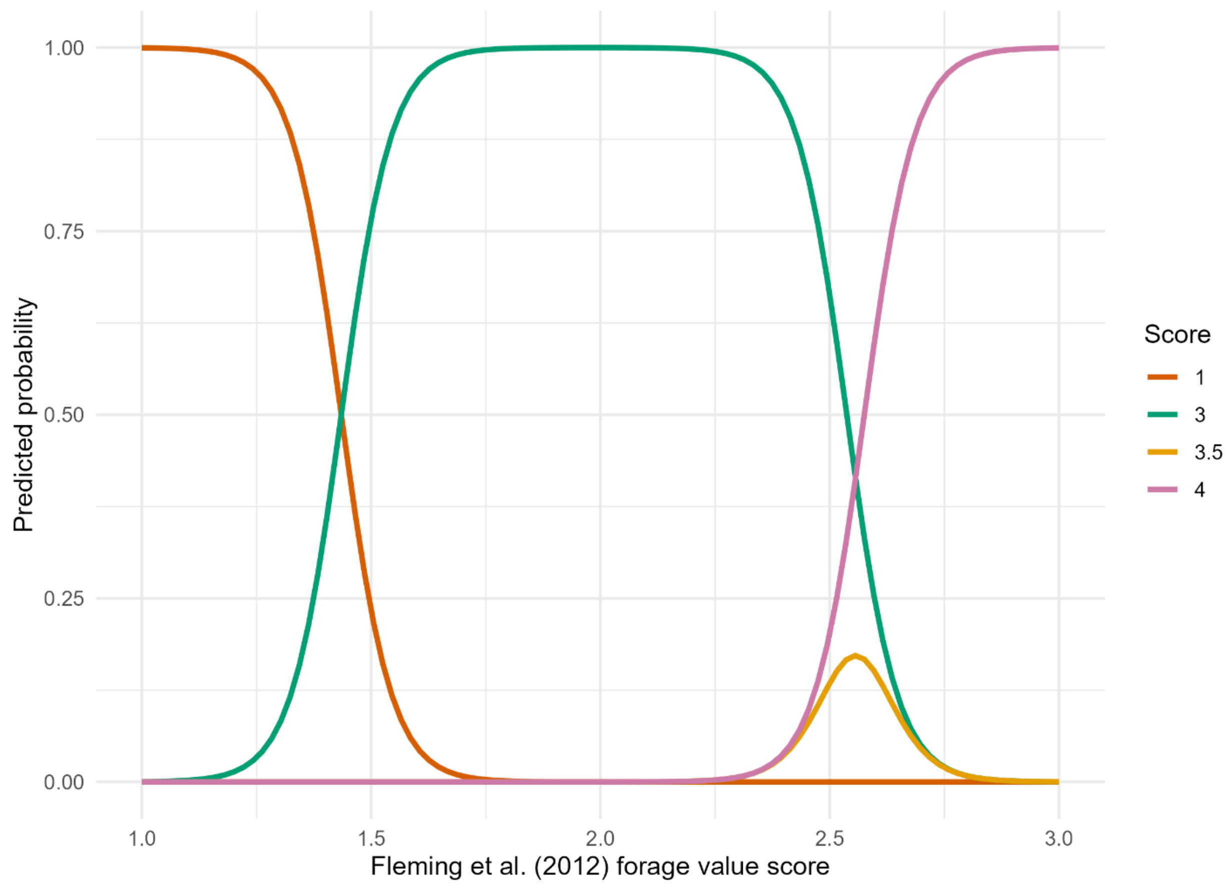

Figure 2-5. Predicted probabilities of expert-assigned forage scores based on Fleming et al. (2012) using an ordinal regression model.

2.3.3 Association with TME

The cumulative link model fitted with TME as the predictor did not find a significant association (Figure 2-6) with expert-assigned vFQI coefficients ($\beta = 0.21 \pm 0.31$ SE, $z = 0.69$, $p = 0.49$; 95% CI -0.38 to 0.83). Nevertheless, the category-specific probability curves in Figure 2-6 are directionally consistent with expectations: the predicted probability of a score of 4 increases with TME, the probability of a 2 decreases, and the probability of a 3 remains comparatively flat. his indicates that TME captures part of the signal considered by experts, even though the global effect was not statistically significant.

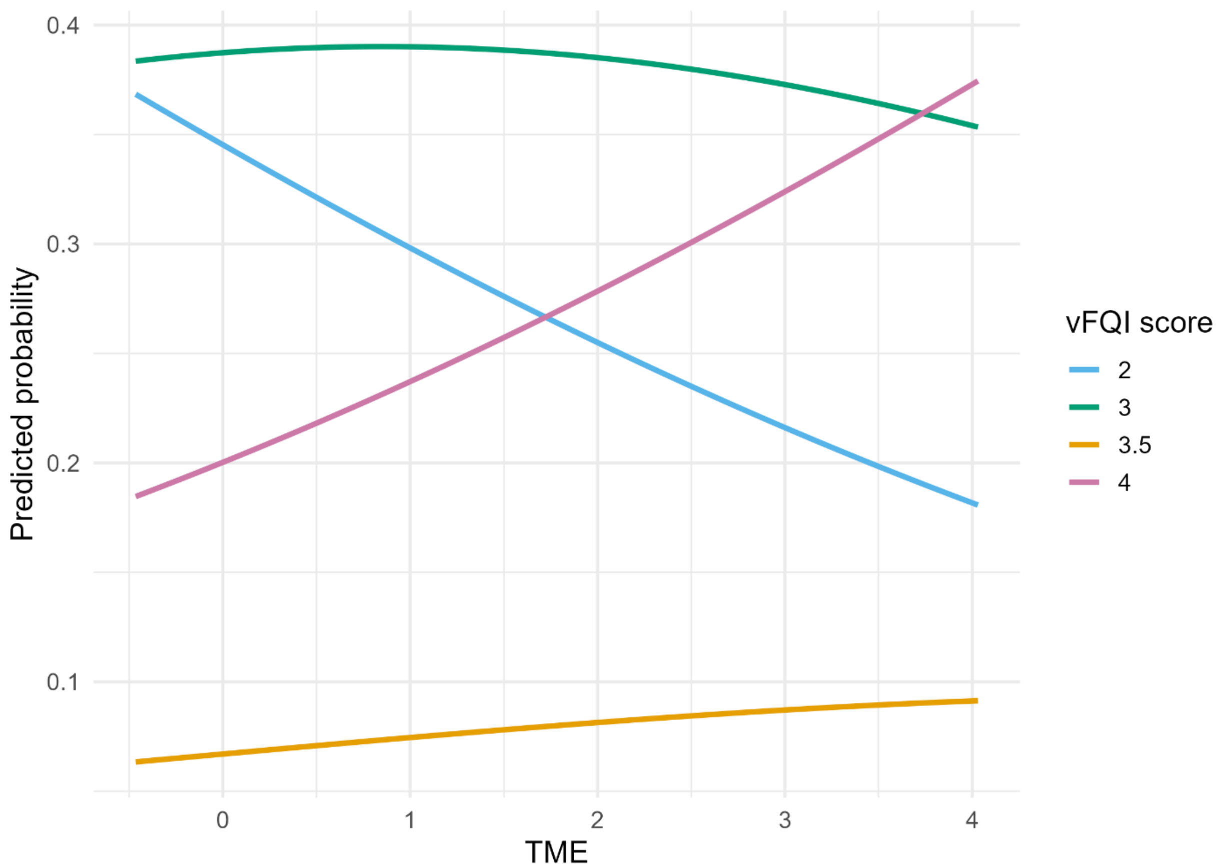

Figure 2-6. Predicted probabilities of expert-assigned forage scores across published TME values for 26 plant taxa. To maximize the number of species included in the comparison, I incorporated six plant taxa (*Amaranthus* spp., *Chenopodium album*, *Elodea canadensis*, *Paspalum laeve*, *Rhynchospora corniculata*, and *Rumex crispus*) for which vFQI coefficients were available from Fleming et al. (2012) and Farley et al. (2022), but which were not a part of the expert survey conducted in this study. For visualization purposes, vFQI coefficients for these species were rounded to the nearest integer to simplify the predicted probability curves.

2.3.4 Agreement with plant traits

I evaluated the influence of 21 plant functional traits on expert-assigned vFQI scores using cumulative link models (Appendix J). Three traits showed statistically significant effects (Figure 2-7). Dispersal unit dry mass was the strongest positive predictor ($\beta = 0.91 \pm 0.42$ SE, $z = 2.17$, $p = 0.030$), indicating heavier dispersal units increase the chance of a higher forage rating. Leaf N content also had a significant positive effect ($\beta = 0.47 \pm 0.21$ SE, $z = 2.22$, $p =$

0.026), suggesting experts gave higher forage-value scores to plants with richer foliage nitrogen. By contrast, seedbank duration was a significant negative predictor (β = -0.57 ± 0.28 SE, z = -2.04, p = 0.041), meaning longer-lasting seedbanks tend to receive lower forage-value scores. All other traits, including seed oil content (p = 0.093) and root C/N ratio (p = 0.277), showed no significant association with vFQI ratings.

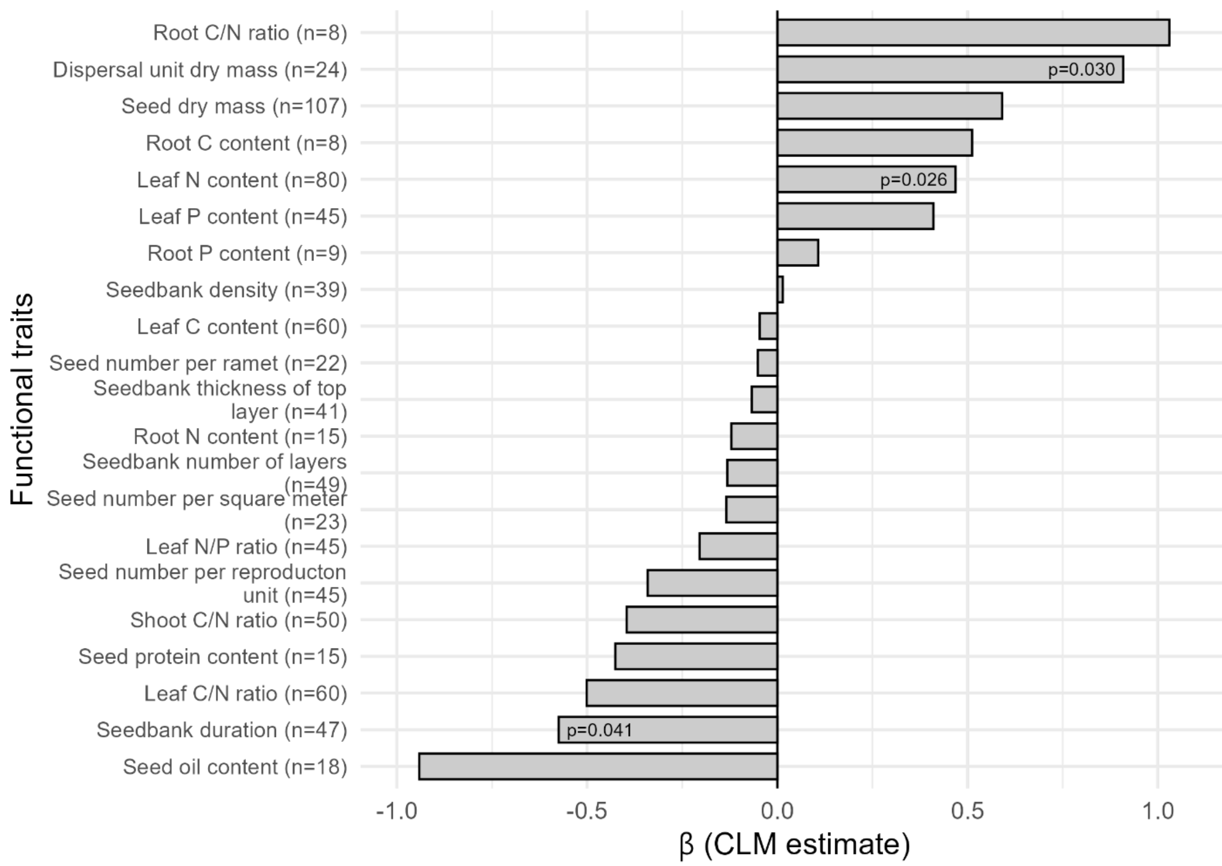

Figure 2-7. Predicted effect sizes (β) of plant functional traits on expert-assigned vFQI scores based on cumulative link models. Bars show model estimates (β) for each trait, ordered by effect size. Positive values indicate traits associated with higher forage value scores; negative values indicate traits associated with lower forage value scores. p-values < 0.05 are labeled for traits with statistically significant effects. The value of n represents the number of plant taxa for which data were available for each trait.

2.4 Discussion

This chapter addressed two primary objectives. The first objective was to derive expert-informed forage value coefficients for the most common wetland plant species in southern Ontario, allowing for regional adaptation of the vFQI. The second objective was to explore how these expert-assigned coefficients correspond with independent metrics of nutritional value, including experimentally-derived TME values and plant functional traits.

To address the first objective, I conducted an expert elicitation survey with 22 participants specializing in waterfowl ecology and wetland plant communities to derive forage value coefficients for 74 plant taxa. The inclusion of confidence ratings in the survey design introduced a novel element not previously applied in forage assessments, including those by Fleming et al. (2012) and Farley et al. (2022), or in waterfowl diet studies more broadly.

To address the second objective, I examined the degree of correspondence between the expert-assigned coefficients and other indicators of plant nutritional quality. Specifically, I compared the coefficients to experimentally measured TME values for a subset of taxa, as well as to a series of plant functional traits that may influence waterfowl foraging decisions.

Accordingly, in the sections below, I first discuss the development of the forage value coefficients, including the choice of aggregation method, patterns of expert evaluations across taxa, confidence-associated trends, and comparison with previously published coefficients. Following this, I discuss the associations of the expert-derived coefficients with experimentally-derived TME values and plant functional traits.

2.4.1 C-coefficient calculations

Following the completion of the expert survey, I considered three possible methods for calculating forage value coefficients: 1) the arithmetic mean, 2) the confidence-weighted mean, and 3) the median. In previous studies, Fleming et al. (2012) and Farley et al. (2022) used the arithmetic mean to calculate forage coefficients (Table 2-1). The arithmetic mean allows for finer differentiation between plant taxa based on expert evaluations. However, it is highly sensitive to outliers, which can disproportionately influence final scores, and more importantly, assumes equal intervals between rating categories. This assumption is problematic in the context of ordinal data, where the difference between a score of 1 and 2 may not reflect the same magnitude of difference as between 3 and 4. I also considered the confidence-weighted mean, as it incorporates expert confidence by giving greater influence on responses with higher confidence scores. While this approach reduces the impact of uncertain or inconsistent evaluations, it still relies on the assumption of equal distances between ordinal categories. Therefore, it does not fully address the limitations associated with ordinal data. Likert-type scales are widely recognized as ordinal in nature, and using arithmetic means with such data can be statistically problematic (Cliff, 1996; Jamieson, 2004; Norman, 2010). Because experts in my study assigned coefficients following a 1-4 Likert-type scale, I opted to use the median to derive coefficients from the different expert values. The median does not rely on assumptions about the spacing between ordinal categories and is more robust to extreme values. However, it limits the possible coefficient values to a smaller set, rather than generating a continuous distribution of coefficients. Given the goals of this study and the nature of the data, I nonetheless selected the median to calculate forage value coefficients.

2.4.2 Distribution of expert scores

The observed distribution of forage value coefficients reveals clear variation in how plant taxa were evaluated by experts. Most plant species received low to moderate median ratings, while only a small subset (9/74 taxa) was classified as having high forage value (i.e., median score = 4), as shown in Figure 2-1. This distribution supports the idea that wetland plant communities differ significantly in their suitability for waterfowl nutrition, with only certain taxa providing highly valuable food resources. Such patterns are supported by field studies documenting species-specific preferences in the diets of wild waterfowl. A study of nonbreeding Green-winged Teal (*Anas crecca*) in Texas playas found that birds did not consume seeds in proportion to availability. Across twelve wetlands, all age-sex cohorts consistently preferred certain taxa, such as Water Smartweed (*Polygonum amphibium*), Pale Spikerush (*Eleocharis macrostachya*), Knotgrass (*Paspalum paspaloides*), and Barnyard Grass (*Echinochloa crus-galli*), while other seeds were underutilized (Anderson et al., 2000). Similar trends were observed during spring migration in Nebraska's Rainwater Basin, where 59 female Mallards and Blue-winged Teal selected seeds from Wild Millet (*Echinochloa* spp.) and Cutgrass (*Leersia* spp.) more frequently than expected (Tidwell et al., 2013). This pattern suggests that waterfowl do not feed randomly but rather show strong forage preferences. This foraging preference has important implications for habitat management. For example, in the Mississippi Alluvial Valley, Hagy and Kaminski (2012a) found that excluding 17 rarely-consumed seed taxa from food availability estimates resulted in a 30% reduction in total biomass. They note that including low-utility species in energetic models can substantially overestimate habitat quality for waterfowl.

Despite these examples of selective foraging, some researchers have questioned the reliability of identifying universally preferred food plants. A review of autumn-winter diet studies found that only 5% of investigations used unbiased sampling methods capable of detecting true dietary selection, while the majority relied on gizzard contents from hunter-harvested birds without measuring food availability. As a result, many reported preferences may reflect sampling artefacts or regional constraints rather than consistent foraging patterns across contexts (Callicutt et al., 2011).

Nevertheless, our results show that experts consistently distinguished among plant taxa, grouping them into clusters based on perceived forage value and confidence (Figure 2-2). Some taxa - such as Wild Millet (*Echinochloa* spp.), Duck Potato (*Sagittaria* spp.), and Wild Rice (*Zizania* spp.) received consistently high scores with high confidence (score = 4, confidence = 4), suggesting strong agreement regarding their high forage value. In contrast, taxa such as Willow (*Salix* spp.), Common Reed (*P. australis*), and Cattail (*Typha* spp.) were confidently rated as low-value (score = 1, confidence = 4). Other species, including Common Groundnut (*Apios americana*), Dodder (*Cuscuta* spp.), and Fireweeds (*Erechtites* spp.), were typically assigned low scores with low confidence, which may indicate unfamiliarity or ecological ambiguity in their role as waterfowl forage.

To help elucidate possible explanations for expert scoring, I reviewed the 136 comments regarding 74 plant taxa left by 11 out of 22 experts who provided written remarks in their survey responses. I aggregated these comments under six categories based on combinations of confidence levels and coefficient scores (Appendix K). When plants received low scores (i.e., 1-2) with high confidence (i.e., 2.5-3), experts often provided clear and consistent reasoning for their assessments. These comments included references to known low energetic value, poor accessibility, or lack of importance for waterfowl diets, citing studies or personal observations that informed their coefficients. In these cases, we can be reasonably confident that these species provide little nutritional value to waterfowl. In contrast, many plants that received low scores (i.e., 1-2) were also rated with low confidence (i.e., 1-2). This pattern raises the question of whether the low scores in these cases reflect true low forage value or simply a lack of information. It is possible that these plants have not been studied thoroughly because they are not typically consumed by waterfowl. Alternatively, the absence of information may simply reflect research priorities, where scientists have focused their efforts on species already known to be important in waterfowl diets. Anecdotally, plants that received low coefficient values (i.e., 1-2) with low confidence (i.e., 1-2) were associated with comments that experts were unfamiliar with the taxon or did not associate them with wetland habitats or food for waterfowl. Some experts expressed that they were uncertain about the energy value or accessibility of these plants. As confidence in the scores increased (i.e., 2-4) across the other groups, comments about unfamiliarity with the plant taxa or their association with wetlands became less frequent. This suggests that experts were careful when evaluating unknown or less familiar species and tended to express lower confidence in such cases. Plants that obtained high scores with higher confidence (i.e., 3-4) elicited comments from experts noting high energetic value, sometimes referencing TME directly or providing detail about seeds per ha or seeds per seedhead. Comments indicated that scores were based on a combination of personal field observations, literature sources, and knowledge of plant traits such as seed size, tubers, or germination conditions. Experts often mentioned that their conclusions were not based on a single source of information but rather formed by combining direct experience with external data or scientific references. This suggests that confidence scoring was mostly related to the availability of information about dietary preferences, and nutritional or energetic content of the plants. However, the association between coefficient scores and confidence means that we cannot be certain whether little information is known about low coefficient/low confidence plants because they are not consumed by waterfowl or simply because they have not yet been studied adequately. Scientists likely have prioritized studying the nutritional and energetic value of plants they know waterfowl consume, and so despite the low confidence for some low coefficient plants, I conclude that the coefficient values are still useful in assessing plant community forage value for waterfowl.

I also attempted to explore whether expert forage value scores and confidence levels differed between invasive and native plants using the VasCan database to determine species status in Ontario. However, this analysis proved inconclusive due to the genus-level structure of the survey and the presence of both native and introduced species within many genera (Appendix L).

2.4.3 The interdependence of expert confidence and scores

Neither Fleming et al. (2012) nor Farley et al. (2022) included expert confidence ratings when collecting forage value coefficients for waterfowl dietary studies. Although this approach has not previously been applied in forage value assessments for waterfowl, it is a widely-used practice in expert elicitation studies across various fields such as environmental risk assessment, species conservation, ecological modeling, and decision science (Martin et al., 2012; McBride et al., 2012; Hemming et al., 2018). Collecting confidence levels allows researchers to better capture and quantify uncertainty behind expert judgments, identify knowledge gaps and data deficiencies, reduce the influence of cognitive biases, increase the transparency of the evaluation process, improve the weighting and aggregation of expert opinions, and ultimately enhance the interpretability and reliability of the final assessments by distinguishing between well-supported and more speculative evaluations (O'Hagan, 2019). In this study, I incorporated confidence ratings into my expert survey to explore several interrelated questions: 1) whether confidence levels correlate with the assigned forage value scores, and 2) whether lower-confidence scores are characterized by greater variability indicating uncertainty. These analyses aimed to better understand both the evaluation process itself and potential knowledge gaps regarding specific plant taxa.

In addition to examining score frequencies, I evaluated the relationship between expert-assigned forage values and their confidence levels. Before conducting the statistical analysis, I did not expect to see any relationship between confidence level and the score given. However, as a result, I observed a positive correlation between expert confidence and assigned forage value scores (Spearman's $\rho = 0.95$, $p = 0.051$).

Additionally, I hypothesized that lower-confidence scores would show greater variability, suggesting uncertainty among experts. These expectations were based on the assumption that confident experts rely on clear personal experience or knowledge, while uncertain ones might default to more cautious or neutral evaluations. Higher-confidence scores, in contrast, were expected to show greater consistency and tighter clustering. Interestingly, the relationship between confidence and score dispersion was more complex than initially expected. As shown in Figure 2-3, there was a positive relationship between confidence level and score variance ($R^2 = 0.985$, $F = 130.91$, $p = 0.008$), indicating that experts who expressed higher confidence did not always converge on a single score, but instead used a wider range of values. One possible explanation is that experts who feel more confident in their evaluations feel more justified in using the full range of the scale, leading to increased variation in their responses. This phenomenon has been referred to in the literature as an "expertise effect" where individuals with deeper knowledge are more willing to express strong, divergent opinions based on nuanced distinctions (Harvey et al., 2004; Askarisichani et al., 2020; Lackner et al., 2023). In contrast, experts with lower confidence may default to middle-category or conservative scores, leading to less variance within that group (S.-W. Lin & Bier, 2008; Larrick & Feiler, 2015; Pereira & Öhberg, 2024). Another interpretation is that plant taxa receiving low-confidence ratings are those that are poorly studied or ecologically ambiguous in their role as waterfowl forage. The low confidence may reflect a genuine lack of data or experience with certain taxa, which in turn leads to scoring convergence around lower values due to limited information. This pattern was also reflected in expert comments provided during the survey (Appendix K). Experts frequently explained that for some plant taxa, especially less studied or regionally rare species, their limited familiarity led them to assign lower scores out of caution rather than due to an assessment of truly low forage value. This suggests that some taxa receiving low median scores may not necessarily have poor forage quality but represent targets for future research to clarify their nutritional role for waterfowl.

An alternative approach was considered, which involved removing low-confidence scores to assess their impact on median forage values. However, this method was ultimately rejected, as species that experts identified as poorly known often lost most or all their scores following such filtering (Appendix M). Moreover, due to the limited sample size of this study, excluding these scores could introduce bias and reduce the robustness of the results.

Overall, the inclusion of confidence ratings provided valuable additional insights into expert evaluation patterns. Specifically, it revealed that: 1) higher confidence was generally associated with higher assigned forage values; 2) confidence influenced scoring behavior, with highly confident experts using a wider portion of the scale while less confident experts tended to cluster around central scores; and 3) some taxa may have received lower scores not because of inherently low forage value, but due to insufficient data and resulting caution among experts. Together, these patterns suggest that confidence levels shape both the central tendency and internal agreement of expert ratings. Thus, incorporating expert confidence into future analyses may improve the interpretability and reliability of expert-based assessments of forage value.

2.4.4 Comparison with previously published forage value coefficients

Farley et al. (2022) evaluated the vegetative forage quality index (vFQI) using expert-derived forage quality C coefficients, following the approach of Fleming et al. (2012). To compare their findings, they analyzed 18 plant taxa common to both studies and found a strong correlation ($\rho = 0.95$) between the assigned coefficients. In my study, I included 30 plant taxa that had been evaluated in previous vFQI studies: 22 shared with Fleming et al. (2012) and 25 shared with Farley et al. (2022; Appendix I). Because I took the median coefficient values rather than average the coefficient values from each expert, my vFQI coefficients comprise ordinal data and therefore a cumulative link model was more appropriate than relying solely on Spearman's rank correlation analysis (Appendix H). While Spearman's rank correlation is not incorrect for ordinal data and can reveal the strength of monotonic associations, it treats the two variables symmetrically. In contrast, the cumulative link model explicitly models the ordinal coefficients as the response variable, allowing me to articulate the directionality of the relationship (predictor vs. response). For this reason, CLM was the more suitable choice in the context of my study (Christensen, 2019). In general, agreement among studies was highest for plants with coefficients equal to 1, 3 or 4, which were most common among the 30 plants that I compared. I found a strong and statistically significant agreement among my new coefficients and those from Farley et al. (2022) ($\beta = 7.43$ (± 2.71 SE), $z = 2.74$, $p = 0.006$). The agreement between the new coefficients and those from Fleming et al. (2012) was weaker and of marginal statistical significance ($\beta = 18.23$ (± 10.69 SE), $z = 1.71$, $p = 0.088$). Two factors likely explain this contrast. First, the scales were aligned differently: Farley et al. (2022) used a 1-4 coefficient scale, whereas Fleming et al. (2012) used a coarser three-class 10/20/30 system with an “unknown” category (Table 2-1), which compresses variance and reduces discriminatory power. Second, the overlap in taxa was smaller for Fleming, with fewer shared species (22 vs. 25) and no new coefficients equal to “2” among those 22 (Figure 2-5), limiting comparability in the middle of the scale. Given these constraints, $p = 0.088$ is still informative because the effect is directionally consistent and suggestive of concordance, even if it does not meet the conventional 0.05 threshold. A further, temporal explanation is that the 13-year interval since Fleming et al. (2012) may have introduced new information and shifted expert perceptions, which could also contribute to the stronger alignment with Farley et al. (2022).

Overall, these findings indicate a reasonable degree of consistency in expert-based forage value assessments across studies conducted in different regions and time periods. The strong association suggests that, despite potential variation in local expert pools or taxon familiarity, the relative ranking of plant taxa remains largely stable. This consistency supports the robustness of the expert elicitation method and reinforces the validity of using expert-derived coefficients in wetland management contexts. In addition, the observed agreement allows for the integration of coefficients from all three studies into a shared reference database. Such a combined resource can support broader applications of the vFQI framework, facilitate cross-regional comparisons, and improve coverage for plant species that have not yet been assessed in specific regions like southern Ontario.

2.4.5 Association with experimentally derived TME values

Because the vFQI coefficients were elicited as a multidimensional judgement (e.g. integrating nutritional quality, accessibility, and availability), a one-to-one correspondence with TME, which indexes only the energetic content of ingested material under controlled conditions, was not anticipated, and my expectations were deliberately cautious. I therefore framed the TME comparison as a partial validation. If expert coefficients capture genuine energetic signal, higher coefficients should, on average, align with higher TME values.

The cumulative-link model with TME as the sole predictor did not yield a statistically significant overall association with the ordinal vFQI scores ($\beta = 0.21 \pm 0.31$ SE, $z = 0.69$, $p = 0.49$; 95% CI -0.38 to 0.83). Interestingly, while the model did not identify a statistically significant association, the predicted probability plot (Figure 2-6) reveals some tendencies in the relationship between the two measures. Plants with expert-assigned coefficients of 2 were more likely to have low TME values, while plants with coefficients of 4 tended to be associated with higher TME values. In contrast, plants with intermediate coefficients of 3 or 3.5 were distributed across both low and high TME values, indicating greater variability and context-dependence.

Across multiple studies, the TME method is consistently recognized for its key strength: it estimates the actual amount of energy absorbed by waterfowl after accounting for metabolic losses, offering a more realistic measure of forage quality than gross energy (Checkett et al., 2002). TME is particularly useful in conservation planning, as it supports energetic carrying capacity models by providing species-specific energy values for wetland plants (Brasher et al., 2007). The method allows for correction of non-food energy excreted in feces and urine and requires only small food quantities, making it suitable for rare or hard-to-collect samples (Checkett et al., 2002; Lancaster et al., 2019). It also facilitates direct comparisons among plant species and across studies when standardized properly, and highlights differences in digestive efficiency among duck species with different foraging strategies (Dugger et al., 2007; Gross et al., 2020).

Despite its strengths, numerous studies have highlighted methodological and conceptual limitations of the TME approach that may affect the reliability and ecological relevance of the resulting values. A recurring issue is the use of captive-reared birds instead of wild individuals, which may distort energy assimilation estimates due to differences in digestive physiology shaped by life history and diet (Checkett et al., 2002; Dugger et al., 2007). Additionally, forced-feeding methods, though necessary for standardization, often induce stress responses that alter digestion, reduce feed retention, and may result in regurgitation or incomplete processing of the food (Checkett et al., 2002; Dugger et al., 2007; Gross et al., 2020; Larson et al., 2024). Several studies also report contamination of fecal samples with feathers or remnants of pre-trial food, complicating excreta analysis and energy calculations (Lancaster et al., 2019; Larson et al., 2024). Moreover, the absence of a standardized protocol across studies for fasting duration, feeding technique, or excreta collection introduces additional variation and hinders direct comparisons (Gross et al., 2020; Larson et al., 2024). For instance, Gross et al. (2020) noted that the widely-used correction factor for urinary nitrogen is based on poultry studies from the 1950s and has not been validated for wild waterfowl. TME estimates also show high variability, with significant differences in values for the same plant taxa between species and even among individuals under identical conditions (Lancaster et al., 2019; Gross et al., 2020). Beyond methodological constraints, TME trials are resource-intensive, requiring specialized equipment, trained personnel, and considerable time (Dugger et al., 2007; Lancaster et al., 2019). Finally, TME reflects only the chemical energy content of food and does not incorporate other critical ecological and physiological factors such as food availability, seasonal dietary shifts, digestive adaptation, foraging time, or landscape-level accessibility (including constraints of the environment), all of which play an essential role in determining the actual foraging value of wetland plants for waterfowl (Checkett et al., 2002; Brasher et al., 2007; Dugger et al., 2007). In addition, physical characteristics of seeds, such as hard seed coats, play a key role in determining their digestibility and energetic value, as they can resist digestion and reduce energy absorption by waterfowl. A study by Dugger et al. (2007) showed that Common Spike Rush (*Eleocharis palustris*) had the lowest recorded TME, likely due to its hard seed coat that passed through the digestive tract largely intact. Gross et al. (2020) also reported negative TME values for species such as Sago Pondweed (*Stuckenia pectinata*) and Wild Celery (*Vallisneria americana*), indicating that birds expended more energy digesting the material than they gained, likely as a result of high fiber content. This finding is somewhat unexpected given the established role of these plants as waterfowl forage in the Great Lakes and Mississippi River regions (Varro, 2003; Wersal et al., 2006). Moreover, Gross et al. (2020) emphasized that TME measurements in such trials typically reflect the energy content of seeds alone, whereas in natural settings, waterfowl consume not only seeds but also tubers, leaves, and other vegetative parts that may have different nutritional profiles.

Taken together, the muted overall signal likely reflects both data limitations and the fact that the two metrics capture different facets of “forage value.” The available TME coverage spans relatively few taxa and heterogeneous protocols, so small sample sizes, between-study methodological variation, and digestive-physiology idiosyncrasies reduce the reliability and comparability of TME values. Equally important, vFQI coefficients were elicited as a multidimensional judgement that integrates nutritional quality with accessibility and availability, whereas TME isolates energetic yield under controlled conditions. Experts also incorporated a broader suite of contextual considerations in their ratings (Appendix K).

Even so, the ordinal patterns in Figure 2-6 are encouraging: the probability of a vFQI score of 4 increases with higher TME, whereas that of a score of 2 decreases, while intermediate scores are mixed. As taxonomic coverage and standardisation of TME measurements improves, the form and strength of this connection may become clearer. Overall, these tendencies suggest that vFQI coefficients resonate with direct energetic value even as they intentionally integrate a wider set of attributes. A practical next step is a behavioral validation study to test whether ducks spend more time foraging and accrue more duck-use days in plots dominated by higher-coefficient taxa.

2.4.6 Agreement with plant functional traits

I initially expected to observe positive agreement between expert-assigned forage value coefficients and plant functional traits that are often considered indicators of nutritional quality for waterfowl.

Among the traits analyzed, three showed significant associations with expert-assigned forage value coefficients (Figure 2-7). Dispersal unit dry mass was identified as the strongest positive predictor ($\beta = 0.91 \pm 0.42$ SE, $z = 2.17$, $p = 0.030$), suggesting that species with heavier propagules were perceived by experts as more valuable forage. This likely reflects the assumption that larger reproductive units contain greater energy reserves, making them more beneficial to waterfowl nutrition. Leaf nitrogen content also demonstrated a significant positive effect ($\beta = 0.47 \pm 0.21$ SE, $z = 2.22$, $p = 0.026$), indicating that plants with higher nitrogen concentrations in their leaves were considered more valuable, potentially reflecting higher protein content and better overall nutritional quality. In contrast, seedbank duration exhibited a significant negative association with expert scores ($\beta = -0.57 \pm 0.28$ SE, $z = -2.04$, $p = 0.041$), suggesting that taxa with longer-lasting seedbanks were perceived as providing lower forage value. One possible explanation is that long-term seedbank persistence is often linked with hard-seeded taxa or seed coats that make the seed indigestible for waterfowl.

In addition to the significant predictors, several other traits exhibited relatively strong effect sizes but did not reach statistical significance (Figure 2-7). Notably, traits such as Root C/N ratio, Root C content, and Root P content demonstrated sizeable positive associations with expert-assigned forage values, yet the number of plant taxa with available data for these traits was limited ($n = 8$-9). This suggests that insufficient sample size may have constrained the ability to detect significant effects, and that expanding trait coverage for these root-related characteristics could reveal additional meaningful relationships. In contrast, for traits like Seed dry mass, where data were available for a large number of species ($n = 107$), the absence of a significant association despite high data availability suggests that this parameter is unlikely to exert a strong influence on expert evaluations of forage value. Thus, while increasing sample size may enhance the detection of certain trait effects, others appear inherently less relevant to expert forage assessments based on the patterns observed.

Expert comments often supported these general patterns, referencing a variety of plant characteristics that influenced their scoring decisions (Appendix K). In particular, experts frequently considered seed size and total seed production, the presence and abundance of tubers, overall plant biomass, and productivity. Digestibility-related traits such as seed coat hardness and thickness were also commonly mentioned as factors reducing energetic value. In addition, experts evaluated habitat accessibility, germination site, and the ease of access to different plant parts by waterfowl. For some taxa, they emphasized that nutritional value was associated with specific plant parts, such as seeds, leaves, shoots, or rootstocks.

Although some traits showed significant associations with forage value scores and plant traits, I did not find published studies that directly tested links between such traits and food use by waterfowl. Therefore, I focused on research about seed dispersal by waterbirds, which often examines which plant traits influence ingestion and digestion. These studies, although not directly about food value, help to address the knowledge gap between which seeds are eaten and how traits like size or hardness affect their survival through the digestive system.

A study by Soons et al. (2008) involving 23 wetland plant species fed to Mallards demonstrated that seed digestion was significantly influenced by seed size, hardness, and seed coat thickness, while shape and nutrient content had no consistent effect. Smaller, harder seeds with thicker coats were more likely to survive gut passage, suggesting a reduced digestibility that may impact perceived forage value. These findings were supported by a field-based study by Kleyheeg et al. (2016) that examined the digestive tracts of 100 Mallards, showing that small, round seeds with hard coats were most likely to remain intact, reinforcing the conclusion that such traits reduce digestibility and may affect perceived forage value. Additional evidence comes from a study that analyzed stomach contents of six dabbling duck species in the Netherlands, which showed that while ducks predominantly consumed small, round, hard-coated seeds, they also ingested species poorly adapted for endozoochory (seed dispersal through gut passage and excretion by animals), suggesting a broader intake but selective survival (Soons et al., 2016). Complementary findings based on controlled feeding trials with 30 seed species demonstrated that seed coat strength and volume influenced gut retention time, with harder and larger seeds persisting longer, again implying limited energy gain from these species (Kleyheeg et al., 2017).

A trait-focused study found that seed survival during gut passage was driven by seed mass, coat hardness, and phylogenetic lineage, further underscoring the digestive resistance of certain plant types (Lovas-Kiss et al., 2020). However, a contrasting analysis reported no consistent relationships between seed dispersal and traits like seed size or shape, instead emphasizing habitat structure, seasonal availability, and foraging substrate as the primary drivers - factors that were also frequently referenced by experts in this study (Sebastián-González et al., 2020).

Studies have also emphasized the importance of below-ground structures like tubers. For instance, feeding intensity by Swan Geese (*Anser cygnoides*) increased with tuber dry mass when foraging on Asian Tape Grass (*Vallisneria natans*), peaking at intermediate depths, suggesting a strong energetic payoff from heavier below-ground storage organs (Chen et al., 2019). This finding aligns with the negative association observed between expert scores and seedbank duration in my results. Further support comes from in vitro digestion simulations, which identified coat thickness, hardness, and dry mass as key predictors of digestion outcomes, whereas traits such as shape and nutrient content showed no consistent influence (Van Leeuwen et al., 2023). Additional evidence underscores the importance of environmental context in shaping seed consumption by waterfowl. For instance, in urban habitats, ducks more frequently dispersed seeds with achenes, whereas in natural environments, seeds enclosed in capsules were more common in their diet (Tóth et al., 2023). Such differences likely reflect variations in seed availability and foraging behavior depending on habitat type. This may help explain why, in my study, dispersal unit dry mass was positively associated with expert-assigned forage values.

Finally, more recent studies have expanded these findings by demonstrating that foraging guilds differ in their trait-based seed selection. Dabbling ducks were shown to disperse heavier seeds and taller plant species, while diving ducks were more commonly associated with submerged vegetation (Almeida et al., 2022, 2025). These results suggest that functional traits related to mass, size, and growth form play an important role in shaping foraging interactions, thereby supporting their integration into forage value assessment frameworks.

2.5 Conclusion

This chapter aimed to (1) develop expert-informed forage value coefficients for wetland plant species in southern Ontario to expand the applicability of the vFQI tool, and (2) assess the correspondence between these coefficients and independent measures of nutritional value.

Following the approach previously applied by Fleming et al. (2012) and Farley et al. (2022), I developed forage value coefficients for 74 wetland plant taxa, thereby expanding the spatial coverage and applicability of the vFQI to southern Ontario and the broader Great Lakes region. Importantly, confidence ratings were integrated into the expert elicitation process - a novel addition not previously utilized in forage assessments for waterfowl. This allowed for the explicit evaluation of uncertainty in expert scores. Median scores were selected to generate the final coefficients, providing a robust method of aggregation suitable for ordinal data and variable expert responses.

The resulting coefficients revealed clear differences in expert evaluations across plant taxa, which generally aligned with existing knowledge on waterfowl dietary preferences. Incorporating confidence ratings further allowed us to detect patterns whereby lower confidence was associated with both greater variability in expert scores and, in some cases, with lower forage value assignments. These patterns help to better characterize the inherent uncertainty associated with scoring certain plant groups and provide insight into which taxa may require further empirical investigation. The structured design of the expert elicitation, combined with expert comments, also offered additional ecological context and highlighted knowledge gaps that can inform future refinement of forage assessments. Comparison with the coefficients reported by Fleming et al. (2012) and Farley et al. (2022) showed high consistency across studies despite differences in regional focus. This suggests that the combined datasets may serve as a unified regional reference for vFQI applications across the Great Lakes region.

In evaluating correspondence with independent nutritional metrics, limited agreement was observed between expert-derived forage value coefficients and experimentally derived TME values, while partial associations were found with certain plant functional traits, particularly dispersal unit dry mass and leaf nitrogen content. These findings indicate that expert evaluations incorporate multiple factors in addition to intrinsic nutritional content. Instead, forage selection by waterfowl is likely shaped by a complex interplay of ecological and behavioral factors. These include seasonal changes in dietary needs (Callicutt et al., 2011; Tidwell et al., 2013), the presence of predators or human disturbance (Hagy & Kaminski, 2015; Beatty et al., 2024), the spread of invasive plant species (Monfils et al., 2014; Lishawa et al., 2020; Wersal & Getsinger, 2020), habitat accessibility (Miller et al., 2010; Miller et al., 2014), and the spatial distribution and detectability of food resources (McDuie et al., 2019; Klimas et al., 2022b). Additionally, species-specific foraging strategies, gut morphology, and learned behaviors may further influence what is consumed and prioritized in the wild (Brochet et al., 2012).

Taken together, these findings demonstrate that expert-informed forage value coefficients represent a flexible and ecologically meaningful tool for evaluating wetland habitat quality for waterfowl. By integrating expert knowledge, uncertainty estimation, and broader ecological considerations, this approach allows for more realistic vFQI applications in habitat assessment and management across southern Ontario and the Great Lakes region. Importantly, the high level of agreement between independently conducted studies further supports the robustness of the expert-elicitation approach in capturing consistent general patterns in forage value assessments. The ability to generate comprehensive coefficient datasets for a large number of plant taxa, with relatively lower resource demands compared to empirical feeding trials or laboratory-based analyses, positions this method as a practical rapid assessment tool for regional-scale wetland evaluations. Furthermore, the methodological framework developed in this study highlights future research directions, including targeted empirical evaluations for poorly characterized taxa, further integration of functional trait datasets, and refinement of expert elicitation protocols to improve confidence calibration in forage assessments.

3.0 Assessing the effects of invasive *Phragmites australis* suppression on waterfowl forage value

3.1 Introduction

The wetland ecosystems of the Laurentian Great Lakes are characterized by high diversity and richness of natural resources, providing essential habitat for nesting, migration, and feeding of numerous waterfowl species (Prince et al., 1992; Johnston et al., 2010; Lemein et al., 2017). The hydrological properties of the Great Lakes coastal zone provide for the existence of diverse wetland ecosystems with different landscapes and plant communities that have high forage value for waterfowl (Herdendorf, 1992; Timmermans et al., 2008). Among the Great Lakes, Lake Erie plays a particularly important role as a migratory stopover and staging area for millions of waterfowl traveling along the Atlantic and Mississippi Flyways (FWS, 2025). Its coastal wetlands are designated as internationally significant under the Ramsar Convention and are recognized as priority habitat under the Eastern Habitat Joint Venture (EHJV & NAWMP, 2017). These wetlands not only provide abundant foraging opportunities but also support key life history events such as molt staging and pre-breeding condition replenishment. Despite their ecological richness and global recognition, these coastal wetlands face mounting pressures from biological invasions that jeopardize their suitability for waterfowl.

3.1.1 *Phragmites australis* and waterfowl

Conditions provided by wetland ecosystems of the Great Lakes also support the invasion of Common Reed (*Phragmites australis* ssp. *australis*; hereafter *P. australis*) (Wilcox et al., 2003; Trebitz & Taylor, 2007; Tulbure & Johnston, 2010) which poses multiple threats to wetland-dependent waterfowl populations (Tozer, 2016; Robichaud & Rooney, 2017).

Phragmites australis is a tall, perennial grass native to Eurasia, characterized by rapid growth and invasive potential, particularly the non-native subspecies *P. australis* ssp. *australis* (Guo et al., 2014; Allen et al., 2017). While the native subspecies, *P. australis* ssp. *americanus*, has been present in North America for thousands of years, the invasive subspecies originated from Europe and was introduced to North America prior to 1900, expanding aggressively during the last century across the United States and Canada, notably along the Atlantic Coast and within the Great Lakes region (Wilcox et al., 2003; Lelong et al., 2007; Plut et al., 2011). In Ontario, specifically at Long Point Peninsula, the invasive subspecies showed fluctuating abundance since the 1940s but exhibited exponential growth beginning in the mid-1990s (Wilcox et al., 2003). By the late 1990s, the growth rate at Long Point Peninsula reached remarkably high levels, approximately 50% annually (Jung et al., 2020). Factors such as periods of low water depths in the Great Lakes region (Wilcox, 2012), prevalence of sandy soils (Tulbure & Johnston, 2010), extensive road development (Mazur et al., 2014), intensive agricultural practices (Trebitz & Taylor, 2007), and overall increased anthropogenic and technological pressure have facilitated this spread (Hazelton et al., 2014). Consequently, most wetlands within the Great Lakes Basin located south of the 45th parallel are now predominantly classified as experiencing high or mid-high levels of invasion (Hannah et al., 2020).

Such widespread invasion is problematic for many reasons. For example, Robichaud and Rooney (2022b) found that *P. australis* homogenizes avian communities, reducing bird diversity and facilitating the dominance of generalist species. Whyte et al. (2015) and Robichaud and Rooney (2017) both reported that *P. australis* invasion favours passerines such as Red-winged

Blackbirds (*Agelaius phoeniceus*) over wetland-obligate waterfowl. In addition, Peterson et al. (2022) demonstrated that waterfowl nests located near patches of *P. australis* faced higher predation risk, indicating lower reproductive success. Finally, Wells et al. (2008) noted that the displacement of native plant species by *P. australis* reduces the availability of suitable nesting substrates for waterfowl, further diminishing habitat quality.

In addition to these structural and reproductive impacts, Tulbure et al. (2007) and Robichaud and Rooney (2022a) demonstrated that *P. australis* extensively displaces native plant species, which in turn may compromise food availability for waterfowl. This competitive displacement is driven by several traits that make *P. australis* particularly successful as an invader. A combination of vegetative and sexual reproduction mechanisms (Duncan et al., 2017), high intensity of biomass accumulation (Yuckin & Rooney, 2019), a strong genetic capacity to tolerate stress (Oh et al., 2022), and the wide range of Canada's climatic conditions (Catling & Mitrow, 2011) allow *P. australis* to outcompete native plant species across Lake Erie's coastal wetlands. These native plants provide important food resources in the form of seeds, tubers, and vegetation that waterfowl consume and that create habitat heterogeneity important to invertebrates, which are also valuable waterfowl food.

Given these cumulative impacts on biodiversity, habitat structure, and food availability, managers often opt to suppress *P. australis* (Hazelton et al., 2014). The most widely-used approach involves treating wetlands with broad-spectrum herbicide, followed by spot treatments of areas at risk of reinvasion (Martin & Blossey, 2013; Rohal et al., 2018). Herbicide treatment often significantly suppresses the extent and density of *P. australis* (Back & Holomuzki, 2008). However, once *P. australis* is removed, native plants do not necessarily recolonize (Robichaud & Rooney, 2021). Instead, the suppression of *P. australis* may lead to the spread of other invasive plant species in the ecosystem, such as European Frog-bit (*Hydrocharis morsus-ranae*) (Bonello & Judd, 2020; Robichaud & Rooney, 2021), Hybrid Cattail (*Typh*a x *glauca*) (Whyte et al., 2009), Canada Thistle (*Cirsium arvense*), Purple Loosestrife (*Lythrum salicaria*), and others (Bonello & Judd, 2020). In addition to the adverse effects that these secondary invaders pose to plant communities, they may become dominant (e.g., Robichaud & Rooney, 2021). It is not clear whether replacing *P. australis* with secondary invasive plants improves forage value for waterfowl (e.g., Benson et al., 2019). Managers would benefit from a rapid and reliable waterfowl forage value assessment tool that could inform their invasive plant management decisions and help them track whether treatments have meaningfully improved waterfowl forage value or not.

3.1.2 Vegetation Forage Quality Index

Fortunately, there exists the vegetation forage quality index (vFQI), although it has never been applied to assess the impact of invasive plant management on waterfowl forage value. Originally developed in the Mississippi Alluvial Valley (USA), the vFQI was designed as a rapid assessment tool to estimate the forage value of wetland plant communities for waterfowl by integrating species composition and nutritional quality (Fleming et al., 2012). In the first application of the vFQI, Fleming et al. (2012) assessed how different water level management strategies influenced vegetation quality in moist-soil wetlands across the Mississippi Alluvial Valley. The index was able to distinguish between management regimes, with early drawdown sites exhibiting consistently higher forage scores compared to passive or late-managed wetlands.

In a follow-up study, Fleming et al. (2015) explored how wintering waterfowl responded to vegetation quality across wetlands with different hydrologic regimes. They found that the vFQI was positively correlated with duck density on sites with late drawdowns, suggesting that forage-rich vegetation, reflected by higher vFQI values, may attract greater numbers of waterfowl under certain management conditions.

Benson et al. (2019) applied a modified version of the vFQI to compare forage value between restored and natural wetlands in the St. Lawrence River Valley. Their findings showed that restored sites achieved vegetation quality similar to reference wetlands, while areas with higher invasive plant cover tended to have lower vFQI scores.

Farley et al. (2022) evaluated how water level management influenced vegetation quality in 30 restored wetlands in the Montezuma Wetlands Complex. vFQI scores were notably higher in sites with full or partial drawdowns, especially during autumn, suggesting that active hydrological control can enhance forage conditions for seed-eating waterfowl.

The vFQI has proven successful in evaluating the effects of wetland management strategies such as water level manipulation or vegetation restoration. I contend that it has promise as a tool to assess the outcomes of invasive species suppression. In this study, I propose a novel use of the vFQI to evaluate how efforts to suppress *P. australis* affect the forage value of wetland vegetation for waterfowl.

3.1.3 Weighted Mean Waterfowl-forage Coefficient

The Vegetation Forage Quality Index (vFQI), as described earlier, is a type of weighted-mean index where each plant species is assigned a coefficient representing its forage value for waterfowl. These coefficients are derived from expert opinion, and the index integrates species composition with these values to estimate overall habitat quality (Fleming et al., 2012; Farley et al., 2022). This approach parallels the logic of the Floristic Quality Assessment (FQA) framework, where plant taxa are assigned Coefficients of Conservatism (C-values) based on their sensitivity to anthropogenic disturbance and fidelity to natural habitats (Taft et al., 2006). In the FQA, native species that are highly disturbance-sensitive receive high values (up to 10), while tolerant or weedy species receive lower scores, and non-native or invasive species are typically assigned a score of zero (Spyreas, 2019).

Among the floristic quality assessment tools, the Weighted Mean Coefficient of Conservatism (WMCCs) has been highlighted as particularly robust, since it calculates the weighted average of C-values based on species' relative abundance (Bourdaghs et al., 2006; Bell et al., 2017). Notably, WMCCs has been found to be less sensitive to species richness than the original FQA index formulation, making it more stable across sites with varying biodiversity (Kutcher & Forrester, 2018).

Inspired by this logic, I developed a new derivative of the vFQI, named the Weighted Mean Waterfowl-forage Coefficient (WMWCs). This novel index follows the structure of the WMCCs but substitutes conservation-based C-values with the forage value coefficients developed through expert elicitation (*sensu* Fleming et al., 2012; Farley et al., 2022; Chapter 2.0). In doing so, the WMWCs retains the weighted-mean framework but shifts its interpretative focus toward nutritional value for waterfowl rather than ecological sensitivity. Because it does not directly incorporate species richness, the WMWCs is expected to be less sensitive to species richness and may thus provide a more stable benchmark for comparing the effects of invasive plant suppression on habitat quality.

Because invasion by *P. australis* is known to reduce plant diversity and the goal of *P. australis* suppression is to increase plant community richness, I anticipate that herbicide treatment will cause an increase in plant richness, which might bias the vFQI. By including the WMWCs alongside the vFQI in this study, I aim to evaluate whether these two complementary indices, respond similarly to invasive plant management and reflect meaningful ecological changes in wetland vegetation.

3.1.4 Research objectives

A case study to validate the performance of both the vFQI and the WMWCs for wetland plants common to southern Ontario was conducted by comparing vFQI scores from *P. australis*-invaded, herbicide-treated, and uninvaded reference vegetation areas with direct measures of the mass of edible seeds and tubers collected from those vegetation areas. An ideal opportunity to conduct this validation case study was in the Long Point and Big Creek National Wildlife Areas (NWAs) on the north shore of Lake Erie. These NWAs were established for the purpose of wildlife conservation, research and interpretation, according to the Canada Wildlife Act (1973). In particular, the Long Point NWA and Big Creek NWA comprise Coastal Wetland Habitat, which is one of the priority habitats under the Eastern Habitat Joint Venture for the purpose of migratory bird protection (ECCC, 2020, 2022).

To explore this opportunity and assess the broader utility of both the vFQI and WMWCs in the context of invasive species management, this chapter addresses two primary objectives: 1) to determine if *P. australis* suppression enhances waterfowl forage value in the Long Point and Big Creek NWAs by applying two forage quality indices (vFQI and WMWCs), and 2) to determine how both indices relate to food availability by comparing their values to seed mass metrics derived from edible fractions obtained through soil core sampling.

3.2 Methods

3.2.1 Study area

Fieldwork was conducted in August and September 2022, and August and October 2023. In 2022, I established 48 50-m transects in the Long Point (n = 24) and Big Creek (n = 24) NWAs (Appendix N). I established these transects in three vegetation types: *P. australis*-invaded (8 in Long Point NWA and 8 in Big Creek NWA), herbicide-treated (formerly invaded by *P. australis*; 8 in Long Point NWA, which were treated in fall 2022, and 8 in Big Creek NWA, which were treated in fall 2021), and reference with native vegetation (8 in Long Point NWA and 8 in Big Creek NWA). Each transect was constrained to its assigned vegetation type. To ensure each transect remained entirely within the assigned vegetation type, some were adjusted at the midpoint to accommodate the patch geometry.

At the beginning of the experiment, I had 16 transects of each vegetation type, including experimental controls in areas where the managers left *P. australis* intact. However, to meet their management goals, *P. australis*-invaded transects in the Long Point NWA were herbicide-treated in September 2022 (following data collection), resulting in a shingle design (Figure 3-1). This resulted in four treatment types, control (i.e., invaded by *P. australis*), reference (i.e., never invaded by *P. australis*), treated 1 year ago (i.e., formerly invaded by *P. australis*, but treated with herbicide in the fall preceding sampling), treated 2 years ago (i.e., in 2022 the transect was treated 1 year ago, so in 2023 it was treated 2 years ago).

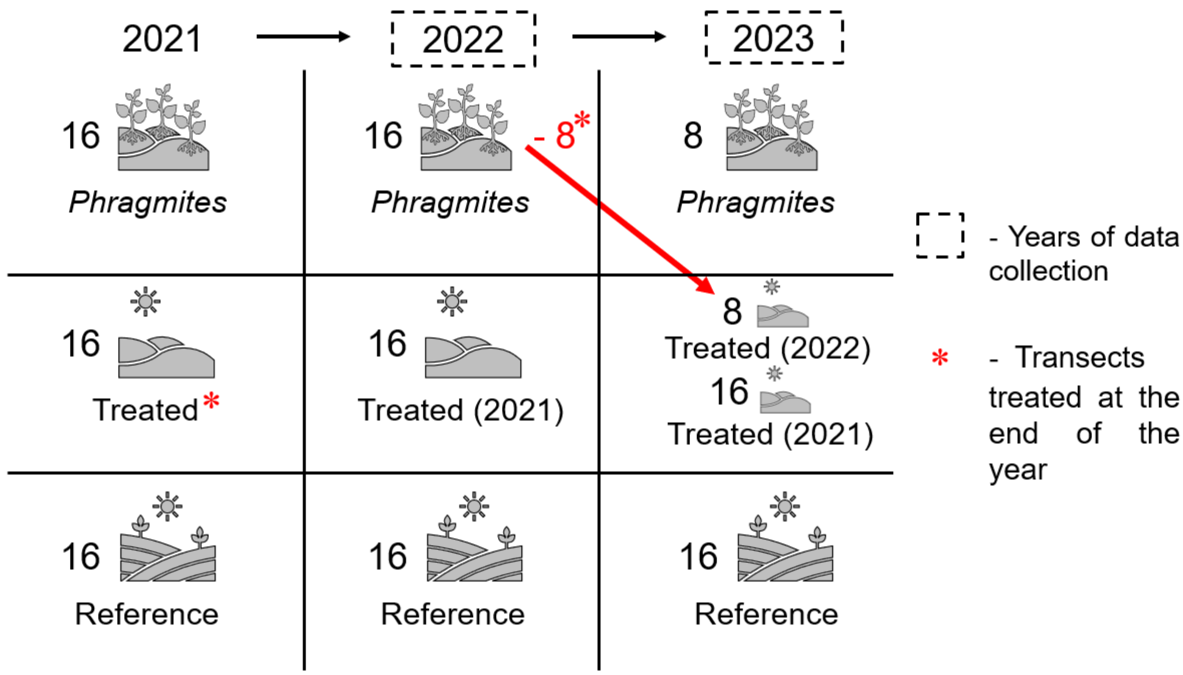

Figure 3-1. Experimental design illustrating the distribution of transects across vegetation types and years of herbicide treatment. During data and sample collection in 2022, there was an equal number of *P. australis*-dominated, treated, and reference transects. Following September 2022, eight of the *P. australis*-dominated transects remained untreated, while eight were treated via herbicide application. Therefore, in 2023, only eight transects remained *P. australis*-dominated, while the remaining eight were incorporated into the treated category. Reference transects with native vegetation remained unchanged across years.

3.2.2 Vegetation surveys

I collected plant biodiversity data from transects in August 2022 and 2023 using a point-intercept method. Briefly, I identified vegetation intersecting with the PVC pole at each 50 cm increment along each 50 m transect, yielding 101 observation points. I identified any plants touching the pole to the genus or species level, depending on the presence of diagnostic plant parts like flowers and seeds, following *The Fourth Edition of A Great Lakes Wetland Flora* (Chadde, 2019). I measured water depth at each point in both 2022 and 2023, and recorded plant canopy height only in 2023 (Appendix O).

3.2.3 Sediment core collection

In September 2023, after the seeds of most plants had ripened, I collected three surficial soil samples (12 cm deep and 5 cm in diameter) from each transect: one at the beginning of the transect (0 m), one in the middle of the transect (25 m), and one at the end of the transect (50 m) - at the locations where GPS coordinates were recorded during the transect surveys. This resulted in a total of 144 samples, which I placed into labeled sampling bags, and kept frozen until I processed them.

3.2.4 Sediment core processing

I processed the sediment core samples in the laboratory between November 2023 and January 2024. I thawed each sediment core sample in the refrigerator for 12 hours before processing. I then washed each sample through stacked sieves with mesh sizes of 4.75 mm, 1.65 mm, and 0.03 mm, and separated the contents into an edible fraction, including seeds and tubers, and an inedible fraction (detailed protocol in Appendix P). I dried the entire contents of each sample in a drying oven at 70 °C for 6 hours. I weighed each fraction, then placed them separately in labeled paper bags, according to the type of fraction and transect identification number.

3.2.5 Forage quality assessment

I calculated forage value for each transect following the approach of Fleming et al. (2012) and Farley et al. (2022), using the vFQI:

Equation 1. Vegetation Forage Quality Index (vFQI) for each transect:

$$VFQI = \sum_{i=1}^{n} \left(\frac{C_i [PO_i]}{N} \right) \times \sqrt{N}$$

Where:

Ci = mean relative forage quality coefficient for plant taxon i per wetland;

POi = proportional occurrence of plant taxon i per wetland;

N = plant taxa richness for each wetland.

I also calculated forage value for each transect using the newly developed WMWCs, a modified version of the WMCCs index (Kutcher & Forrester, 2018) that replaces conservatism scores with expert-derived forage value coefficients, following the structure of Floristic Quality Assessment methods (Bourdaghs et al., 2006; Bell et al., 2017). To accommodate the point-intercept survey method used in this study, I replaced the original proportional cover term with the number of intersections for each species and the total number of intercepts per transect. This modification allows the index to reflect both the frequency of species occurrence and the presence of unvegetated patches along the transect, which better reflects the absolute abundance of food available to waterfowl.

Equation 2. Weighted Mean Waterfowl-forage Coefficient (WMWCs) for each transect:

$$WMWC_S = \sum_{i=1}^{n} \frac{C_{i \times n_i}}{I_t}$$

Where:

C_i = mean relative forage quality coefficient for plant taxon per wetland;

n_i = number of point-intercept intersections for species *i* along the transect;

I_t = total number of intercept points per transect (i.e., 101).

Following Kutcher and Forrester (2018), the subscript 's' indicates that the $WMWC_S$ is calculated at the scale of a single sample (transect), rather than across an entire site.

3.2.6 Statistical analysis

I performed all statistical analyses in R version 4.3.1 (R Core Team, 2023) within the RStudio version 2024.12.1 environment (Posit Software, PBC, 2024). I imported and prepared data in CSV format using the base R function *read.csv*() (R Core Team, 2023), converting variables to factors with *as.factor*() and to numeric with *as.numeric*(). For visualization of vegetation-based indices by transect type and year, I employed the *ggplot2* package to generate boxplots and jittered point plots, applying *theme_minimal*() for consistent styling (Wickham et al., 2025). Model diagnostics, including residual versus fitted plots and Q-Q plots, were produced via the *car* package (Fox et al., 2024), and interaction effects were illustrated with the *effects* package using the *effect*() function to clarify Year × TransectType interactions (Ben-Shachar et al., 2025).

To test the hypothesis that *P. australis* suppression enhances waterfowl forage value in the Long Point and Big Creek NWAs, I applied a two-way ANOVA to both the vFQI and $WMWC_S$ indices. This approach allowed me to examine the main effects of vegetation type (i.e., control, reference, treated 1 year ago, treated 2 years ago) and sampling year (i.e., 2022 vs. 2023), as well as their interaction. I carried out these ANOVA tests by fitting linear models with *lm*() and summarizing with *anova*() from the *stats* package (R Core Team, 2023), and I used the base-graphics function *interaction.plot*() to create complementary interaction plots. Additionally, I assembled final summary tables of R^2 values, F-statistics, and p-values with base R data-frame functions for clear presentation.

I also assessed pairwise associations among the vFQI, the WMWCs and three sediment-derived seed metrics: total seed mass, edible fraction, and inedible fraction. Spearman's rank-order correlation coefficients were calculated using the base-R function *cor.test*() (R Core Team, 2023).

3.3 Results

In 2022, one year after herbicide treatment, the average water depth across control, reference, and treated transects in Big Creek NWA was relatively similar, measuring 17.09 cm (± 8.07), 18.87 cm (± 5.31), and 22.53 cm (± 6.21), respectively. Species richness was lowest in the control transects (8.50 taxa ± 2.98), which were dominated by Common Reed (*P. australis*), and Greater Duckweed (*Spirodela polyrhiza*). Reference transects supported the highest richness (14.37 taxa ± 2.20), with Cattail (*Typha* spp.), and Leafy Pondweed (*Potamogeton foliosus*) being the most common species. Treated transects showed intermediate richness (10.00 taxa ± 3.81) and composition, being characterized by Cattail (*Typha* spp.) like Reference transects and by Greater Duckweed (*Spirodela polyrhiza*) like *P. australis* transects. European Frogbit (*Hydrocharis morsus-ranae*) was common along all transect types.

In Long Point NWA, treated transects had deeper water levels on average (14.88 cm ± 6.33) than both control (5.09 cm ± 6.06) and reference (6.98 cm ± 6.43) transects. Species richness was comparable between treated (8.37 taxa ± 3.50) and control transects (8.62 taxa ± 4.56), while reference transects had the greatest richness (11.75 taxa ± 2.05). Control sites were dominated by Common Reed (*P. australis*), Cattail (*Typha* spp.), and European Frogbit (*Hydrocharis morsus-ranae*). Reference transects were dominated by Cattail (*Typha* spp.), Hardstem Bulrush (*Schoenoplectus acutus*), and European Frogbit (*Hydrocharis morsus-ranae*).

Perhaps because of the slightly deeper water, treated sites supported a mix of European Frogbit (*Hydrocharis morsus-ranae*), and the submersed aquatic milfoils Northern Water-milfoil (*Myriophyllum sibiricum*), and Eurasian Water-milfoil (*Myriophyllum spicatum*).

In 2023, two years after herbicide treatment, water levels increased across all sites. In Big Creek NWA, water depth remained relatively similar among transect types: 22.96 cm (± 11.02) in control, 28.05 cm (± 5.89) in treated, and 20.82 cm (± 6.41) in reference transects. Control sites had the tallest canopy height (329.46 cm ± 46.86) and the lowest species richness (5.37 taxa ± 1.77), with dominant species including Common Reed (*P. australis*) and Muskgrass (*Chara* spp.). Treated transects had the shortest vegetation (75.73 cm ± 76.87) and moderate richness (8.75 taxa ± 3.24), dominated by Cattail (*Typha* spp.) and Leafy Pondweed (*Potamogeton foliosus*). Reference sites had intermediate canopy height (96.20 cm ± 95.87), the greatest richness among vegetation types (10.62 taxa ± 1.99), and were dominated by Cattail (*Typha* spp.) and Muskgrass (*Chara* spp.). European Frogbit (*Hydrocharis morsus-ranae*) was common along all transect types, as in the previous year.

In Long Point NWA, 2023 water depth also increased across all transect types, averaging 25.80 cm (± 8.73) in treated (2021), 9.00 cm (± 8.88) in newly treated (2022; formerly control), and 15.21 cm (± 14.97) in reference transects. The shortest canopy height was observed in the treated (2021) transects (27.68 cm ± 26.16), which also had high richness (18.25 taxa ± 3.49), and were dominated by European Frogbit (*Hydrocharis morsus-ranae*), Cattail (*Typha* spp.), and Leafy Pondweed (*Potamogeton foliosus*). Transects treated in 2022 had taller vegetation (50.60 cm ± 39.03), slightly lower richness (17.62 taxa ± 7.05), and were characterized by Common Reed (*P. australis*), European Frogbit (*Hydrocharis morsus-ranae*), and Cattail (*Typha* spp.). With the *P. australis* now all treated, reference sites had the tallest vegetation (118.09 cm ±

38.20), the greatest richness (18.25 taxa ± 6.62), and were dominated by Cattail (*Typha* spp.), Touch-me-not (*Impatiens capensis*), and Hardstem Bulrush (*Schoenoplectus acutus*).

3.3.1 Vegetation indices as indicators of *P. australis* suppression

I used two-way ANOVA to evaluate how forage quality varied among vegetation types (i.e., control, reference, treated) and between sampling years (i.e., 2022 vs. 2023), based on vFQI scores derived from expert-assigned forage coefficients (Appendix O). The analysis showed no significant main effects for either vegetation type ($F = 0.93$, $p = 0.40$) or year ($F = 0.93$, $p = 0.34$), and no significant interaction between these factors ($F = 0.74$, $p = 0.48$). vFQI values were slightly higher in reference transects compared to control and treated sites (Figure 3-2). However, the variability within groups and between years was considerable. Interaction plots further demonstrated the absence of consistent trends in vFQI values over time across the three vegetation types, indicating that neither the presence of *P. australis* nor herbicide treatment led to detectable changes in forage quality over the two-year study period.

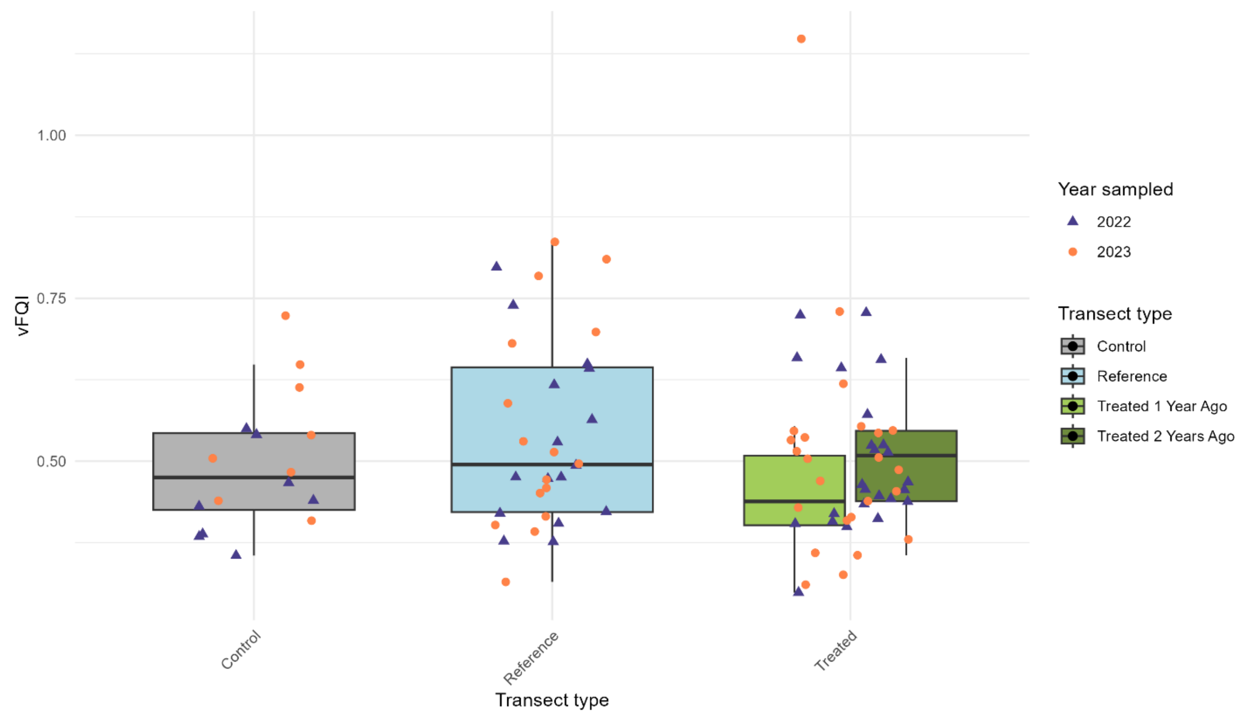

Figure 3-2. Distribution of vFQI scores across vegetation types. Each point represents a 50 m transect.

In contrast, WMWCs showed significant effects of year ($F = 26.69$, $p < 0.001$), vegetation type ($F = 21.37$, $p < 0.001$), and their interaction ($F = 6.47$, $p = 0.002$) (Appendix O). Mean forage value was highest in reference transects and lowest in treated transects (Fig. 3-3). From 2022 to 2023, the mean WMWCs increased in both reference and treated transects, but the 95% confidence interval overlapped for the reference transect (Fig. 3-4). The mean WMWCs for the control transects declined in 2023 but confidence intervals overlapped again.

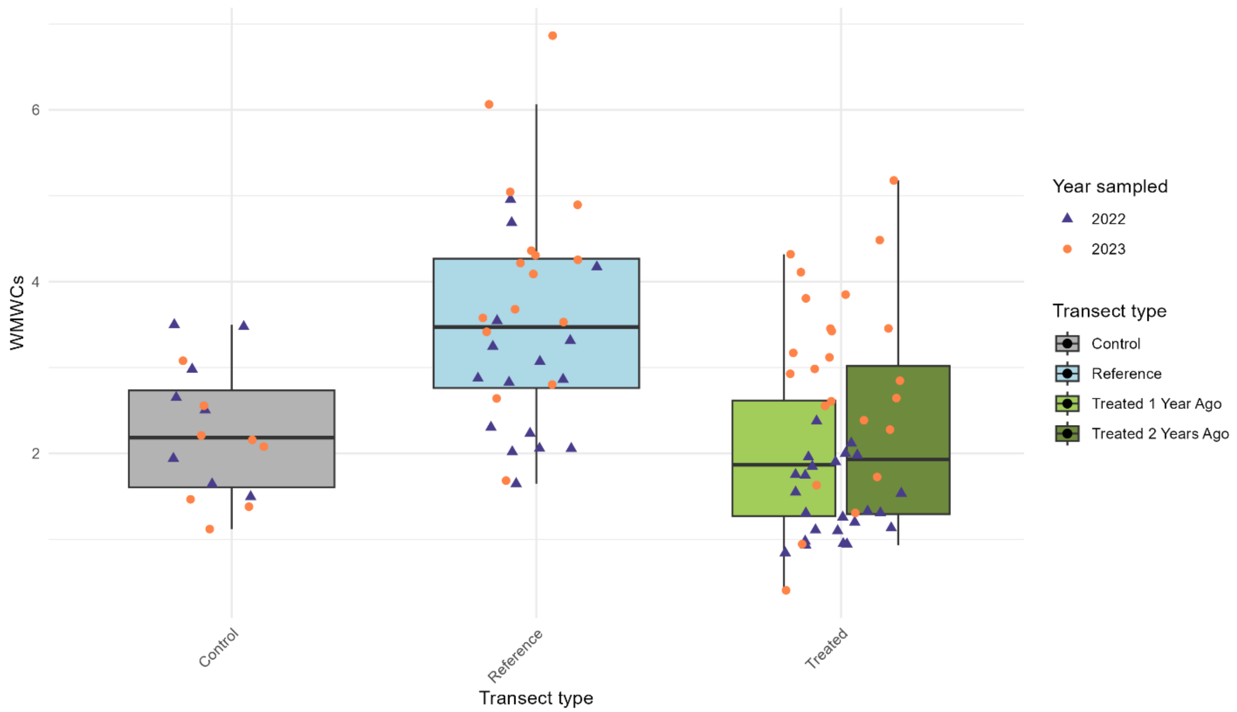

Figure 3-3. Distribution of WMWCs scores across vegetation types. Each point represents a 50 m transect.

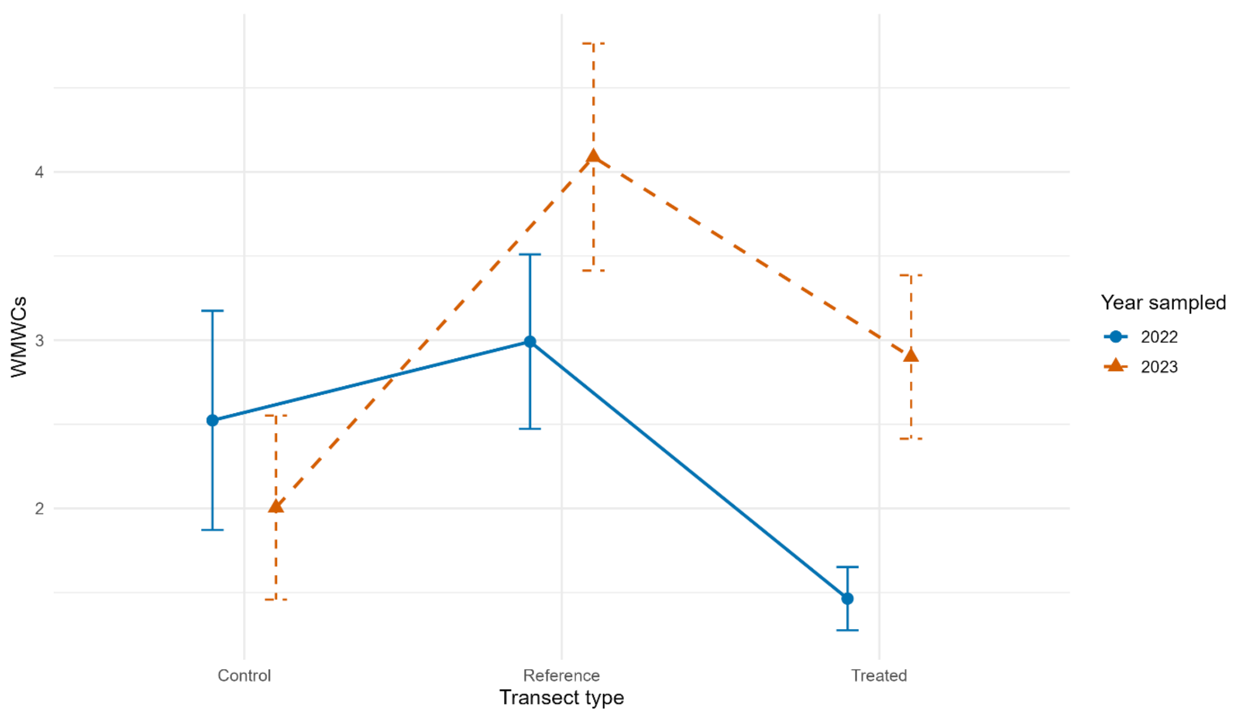

Figure 3-4. Interaction plot showing mean WMWCs values by vegetation type and sampling year.Points denote the mean for each transect type and year. Vertical error bars show 95% confidence intervals for the mean.

3.3.2 Vegetation indices and sediment core contents

To evaluate how well the vegetation-based indices reflected actual food availability for waterfowl, I tested Spearman's rank-order correlations between the index values and sediment-derived seed mass data. This included total seed mass, edible fraction mass, and inedible fraction mass, all extracted from 144 sediment cores collected in 2023 (Appendix O). Only one significant positive correlation was detected: vFQI was moderately correlated with WMWCs ($\rho = 0.32$, $p = 0.03$), likely due to shared underlying forage coefficients. No significant positive correlations were found between either index and the edible seed fraction or total seed mass. In fact, vFQI showed a significant negative correlation with the inedible fraction ($\rho = -0.31$, $p = 0.03$).

3.4 Discussion

3.4.1 Evaluating changes in waterfowl forage quality following invasive species suppression

This chapter aimed to (1) evaluate whether the suppression of *P. australis* improves the forage value of wetland vegetation for waterfowl using two vegetation-based indices (vFQI and WMWCs), and (2) determine how each index corresponds with direct empirical measures of food availability derived from sediment core sampling. Initially, I expected that both the vFQI and the WMWCs would reliably detect vegetation changes associated with *P. australis* suppression and distinguish among control, treated, and reference sites. Whether *P. australis* suppression successfully improved the forage value of wetland vegetation for waterfowl depends on which index is used to measure forage value. While the vFQI did not detect any effect of *P. australis* suppression, the WMWCs increased significantly in the treated transects between 2022 and 2023. Compared with negligible changes in control and reference transects (Figure 3-4).

Also, neither index was strongly correlated with the mass of seeds or edible plant material found in the surface sediments.

It is well-established that *P. australis* invasion reduces plant community diversity (Robichaud & Rooney, 2017; Meyerson et al., 2025) and it takes time for native vegetation to recover following herbicide treatment (Robichaud & Rooney, 2021; Jordan, 2022). Several studies have also shown that waterfowl prefer uninvaded habitats and their abundances decline after *P. australis* invasion (Whyte et al., 2015; Tozer, 2016). In response to these impacts, restoration efforts often aim to enhance plant community diversity. Consequently, I expect *P. australis* suppression to create a systematic difference in plant species richness among our experimental treatments that would result in bias in any index that is sensitive to differences in species richness (Kutcher & Forrester, 2018). However, in this study, the vFQI did not detect any difference in forage value between invaded and reference transects even though waterfowl avoid *P. australis*-dominated areas. This highlights a possible limitation of the vFQI - it may fail to detect ecologically meaningful differences in forage value when these are accompanied by substantial changes in species richness. In prior studies using the vFQI, differences in richness between wetlands with different management strategies were not as extreme, and this may have masked the index's sensitivity to richness (Fleming et al., 2015; Benson et al., 2019; Farley et al., 2022). In contrast, my study involved stark differences in richness between experimental treatments, which may have exaggerated the bias inherent to vFQI formula. Thus, while earlier applications of vFQI found utility in differentiating among management types, those conditions may have been less susceptible to the type of bias revealed here.

To demonstrate the effect of richness on index performance, I used illustrative field-based examples from three 2023 transects representing each vegetation type (Figure 3-5). The control transect consisted entirely of *P. australis*, which had a C-coefficient of 1, resulting in a vFQI value of 1. The treated transect had three plant species with C-coefficients of 1 and 2. Although this composition is intuitively more favourable from a foraging perspective, the vFQI formula divides the weighted sum by the total number of species (N), which in this case lowered the resulting score to 0.69. The reference transect supported a diverse community of 10 to 20 plant species, many of which had higher forage coefficients (e.g., 3 or 4). However, due to the same division by N, the vFQI value was further reduced, reaching just 0.55.

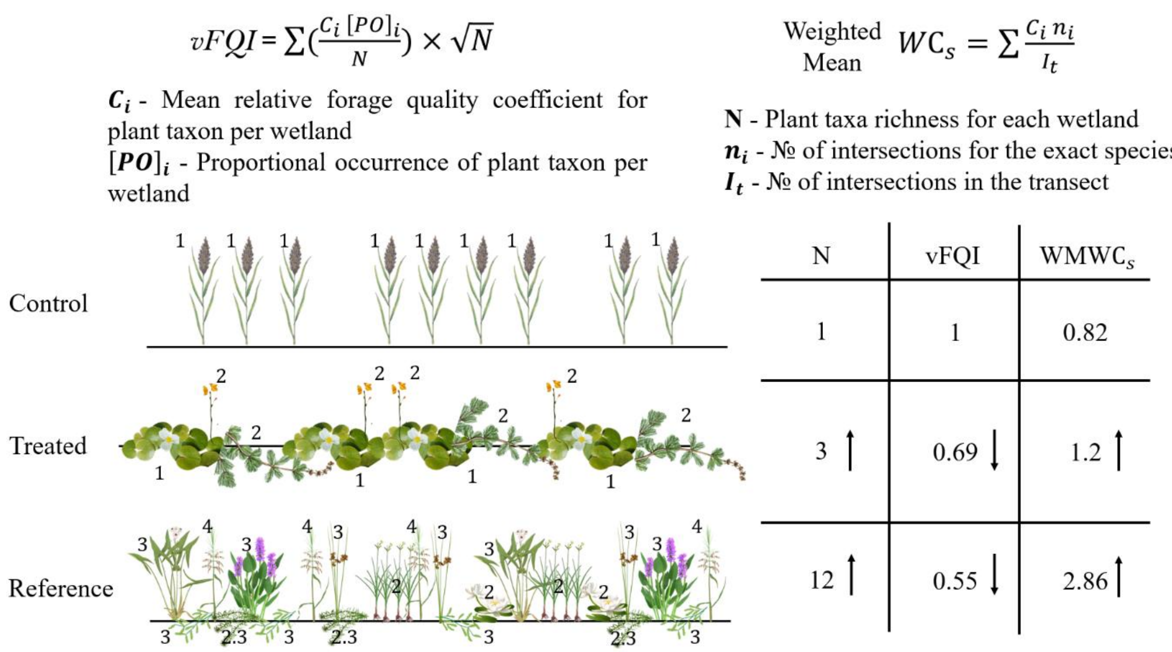

Figure 3-5. Comparison of vegetation structure, species richness (N), and forage quality index values across three vegetation types: *P. australis*-dominated, treated, and reference.The illustrations are based on real field data demonstrating the influence of richness on the vFQI and WMWCs scoring. Numbers above or below each plant illustration represent expert-derived forage coefficients (C-values). Arrows indicate the direction of change in values relative to the control transect. While vFQI values decrease with increasing species richness, WMWCs more consistently reflects changes in community composition and forage quality.

In contrast, the WMWCs index does not take species richness into account. Instead, it weights species by their relative frequency along the transect and calculates the average based on abundance rather than richness. This structure makes it less sensitive to the distorting effects of biodiversity differences between treatments. In my study, the WMWCs captured both a general between-year signal and treatment-related recovery: treated transects showed clear improvement from 2022 to 2023. These results suggest that WMWCs may be more suitable than vFQI for evaluating recovery trajectories in plant communities where species richness varies substantially across treatments.

Therefore, while the vFQI may still serve as a rapid assessment tool in wetlands with low to moderate diversity, minimal turnover, or stark compositional contrasts (such as drawdowns), it may not be appropriate in restoration contexts with variable richness. In contrast, the WMWCs offers a more resilient alternative. Its performance in this study suggests that it is well-suited to capture changes in forage value under more complex conditions. Prior studies using vFQI may not have encountered this limitation because their treatments did not produce major richness differences (Fleming et al., 2012, 2015; Benson et al., 2019; Farley et al., 2022). As such, the results here expand our understanding of when and where vegetation-based forage indices may be appropriately applied and underscore the need to consider index structure when interpreting apparent recovery trajectories in wetland plant communities.

3.4.2 Comparison with sediment core contents

The principle underlying the vFQI and WMWCs indices is that plants with higher coefficient values will provide greater forage value to waterfowl than plants with lower values (Fleming et al., 2012), and therefore a plant community comprising a higher absolute abundance of plants with higher coefficients will thus provide more food for waterfowl to consume. In chapter 2.0, I did observe a significant association between coefficient values and plants' dispersal unit dry mass, leading me to expect that locations with higher vFQI or WMWCs scores would yield sediment with a greater seed mass and potentially with a higher mass of edible plant matter. Yet, neither the vFQI nor the WMWCs was significantly correlated with the mass of edible plant material or the mass of seeds in the sediment cores (Appendix O). The only association I found was that vFQI scores were negatively correlated with the amount of inedible plant matter.

A variety of factors may explain these unexpected results. Hagy et al. (2011) showed that soil core sampling methods underestimate small seeds (≤ 4 mm^3, i.e. ~1.15 mm diameter sphere), recovering only about 75% of them compared to medium (18 – 40 mm^3) and large (≥ 80 mm^3) seeds. I used a minimum mesh size of 30 µm (Appendix P), and perhaps this contributed noise in the correlation between seed mass and index values that obscured a difference among transect types. For example, in my study, control transects dominated by *P. australis* did not show significantly lower seed mass than reference sites. However, control transects tended to have less cattail compared with reference transects, yet cattail produce numerous very small (1 to 2 mm in length, 0.2-0.3 mm in diameter; Grace & Harrison, 1986) seeds, a high proportion of which may have been lost through the sieve in processing. Control transects were conversely dominated by *P. australis*, which has slightly larger seeds (elliptical caryopses 1.2-1.5 mm long and approximately 0.6-0.75 mm wide, with the seed itself 0.8-1.0 mm in diameter; Packer et al., 2017), a larger fraction of which might have been caught in the sieve. This would potentially underestimate the seed mass of reference transects compared to controls.

Beyond sampling bias, biological differences among plant species may also influence seed availability. Van Neste et al. (2020) reported that *P. australis* monocultures in New Jersey provided high seed-based energy compared to other marsh types, despite their ecological drawbacks. However, it is important to highlight the fact that total seed mass may not capture differences in foraging value if seeds are unpalatable or inaccessible to waterfowl (Martin et al., 2022). This notion is supported by prior research demonstrating that some seeds, though abundant, offer little nutritional benefit to ducks. Hagy and Kaminski (2012a) demonstrated that excluding 17 non-consumed seed taxa from carrying capacity estimates reduced total seed availability by over 30%. Hagy et al. (2017) demonstrated that ducks' foraging behavior is influenced by seed burial depth and the availability of alternative feeding sites rather than just seed density. These results emphasize that not all seeds contribute equally to foraging value, and including low-quality seeds in energetic models may lead to overestimations of habitat quality. Despite no evidence of greater seed mass in reference transects compared to control transects dominated by *P. australis*, there may have been important differences in food quality or palatability. Chemical testing may help address nutritional quality of seeds collected from sediment, but I contend that expert-informed indices such as the vFQI and WMWCs offer a means to efficiently integrate knowledge about palatability and nutritional value because the scores in the indices take into account knowledge about the waterfowl forage value of different plants held by the experts who assigned the coefficient values that extends beyond simple seed mass (Appendix K).

In addition to seed-related factors, it is important to consider potential methodological influences on seed data. One such factor is herbicide application, which could theoretically affect seed production or retention. It is also important to clarify that herbicide use itself is unlikely to have biased seed content in sediment cores. In their study of seasonally flooded wetlands in western Tennessee, Osborn et al. (2015) examined the effects of imazapyr application for controlling Alligator weed (*Alternanthera philoxeroides*). While the herbicide significantly reduced the cover and height of this undesirable vegetation in the year of treatment, it did not affect the mass of edible seeds and tubers in the soil, nor did it alter the density or foraging behavior of American Black Ducks using the sites. These findings suggest that changes in seed availability are more closely tied to shifts in plant community composition than to the direct effects of herbicide application.

Finally, it is important to recognize that seed abundance represents only one aspect of habitat quality, and waterfowl respond to a broader range of ecological conditions. For example, Hagy and Kaminski (2015) found that seed density was a poor predictor of duck foraging habitat selection and concluded that other factors, such as perceived predation risk, food availability and handling time, and opportunity costs of foraging, played a more significant role. Olmstead et al. (2013) and Tapp et al. (2018) found that duck abundance was greater in managed wetlands despite no differences in seed biomass, likely due to improved access to food or better habitat structure. Palumbo et al. (2019) showed that habitat choice by Mallards was influenced by both food availability and hunting pressure.

Given these complex connections, it is not surprising that vegetation-based indices do not perfectly correlate with sediment seed data. Even though experts were instructed to evaluate only the value and accessibility of each plant as forage for waterfowl, they frequently mentioned additional factors in their comments (data Chapter 2.0, Appendix K). Indices like the vFQI and WMWCs therefore integrate a broader spectrum of ecological knowledge, providing a more holistic assessment of habitat quality than seed counts alone. These factors may include: (1) variation in the energetic or nutritional quality of seeds and selective foraging preferences of waterfowl (Hagy & Kaminski, 2012a; Martin et al., 2022); (2) the potential of certain plant species to support invertebrate communities that serve as alternative or complementary food sources for ducks (Matuszak et al., 2014; Keljo, 2022); and (3) vegetation structure that enhances cover or reduces predation risk, thereby increasing site attractiveness (Wells et al., 2008; Peterson et al., 2022). Further research is needed to clarify how well these indices reflect actual waterfowl foraging activity and foraging habitat quality under different seasonal conditions and management regimes. As an immediate next step, a behavioural validation study should test whether plots with higher WMWCs scores receive more foraging use by waterfowl (e.g., greater duck-use days and higher time budgets spent feeding).

3.5 Conclusion

This study offered an opportunity to test the vFQI and WMWCs indices using new coefficients that I developed in Chapter 2.0 to evaluate whether management effort to suppress the aggressive invasive grass *P. australis* resulted in rapid improvements in habitat quality for waterfowl. Specifically, whether the plant community that assembled following *P. australis* suppression with herbicide offered enhanced forage value for waterfowl.

The original formulation of this index, the vFQI (Equation 1), did not evidence to detect any change in forage value associated with the *P. australis* suppression within the first two years after herbicide application. Long term monitoring results (Tulbure et al., 2007; Robichaud & Rooney, 2022a; Jordan, 2022) indicate that several years are required for the vegetation community to recover following *P. australis* suppression, and so perhaps as succession proceeds, the vFQI would eventually detect an effect of suppression. However, my results highlight an aspect of the vFQI calculation that may limit its utility in monitoring the outcomes of invasive plant management. Due to its sensitivity to differences in plant community richness, vFQI may be inappropriate for use when comparing vegetation communities recovering from dominance by an invasive plant. The alternate index I developed based on the floristic quality assessment literature is more robust to differences in plant community richness. Therefore, I suggest that wetland managers seeking to evaluate the outcomes of invasive *P. australis* management efforts for waterfowl apply the WMWCs, which I developed based on the WMCCs (Kutcher & Forrester, 2018). This alternative index will show less bias where management actions yield a change in plant community richness.

Neither index showed strong correlations with the mass of seeds or edible plant material collected from sediment cores. However, I contend that this does not invalidate the indices. Rather, I suggest that the coefficients derived from expert opinion (Chapter 2.0) might incorporate aspects of waterfowl foraging preferences beyond seed yield. For example, that seeds from some plants might have greater energetic or nutritional value and hence be preferred (Hagy & Kaminski, 2012a; Martin et al., 2022), that waterfowl may select habitat for foraging using criteria beyond simply the total amount of seeds provided, like protection from predators (Peterson et al., 2022) or accessibility (Hagy et al., 2017), or that waterfowl may be cueing in on certain plants because of the macroinvertebrate food resources associated with them (Keljo, 2022). Future work should seek to measure waterfowl foraging preferences and habitat selection to directly test whether vegetation with higher vFQI or WMWCs scores is preferred by waterfowl or supports higher densities of foraging birds.

4.0 Conclusions and recommendations

4.1 Thesis overview

The coastal wetlands of the Laurentian Great Lakes, particularly those along Lake Erie, provide critical habitat for migratory waterfowl by offering abundant foraging and breeding opportunities (Prince et al., 1992; Timmermans et al., 2008; Johnston et al., 2010). These ecosystems are globally recognized for their ecological value (EHJV & NAWMP, 2017) but have been increasingly threatened by the invasion of *Phragmites australis* ssp. *australis* (hereafter *P. australis*), a highly aggressive perennial grass (Wilcox et al., 2003; Guo et al., 2014). In southern Ontario, including the Long Point and Big Creek National Wildlife Areas (NWAs), *P. australis* has rapidly displaced native plant communities, leading to structural habitat simplification and potential reductions in food availability for wetland-dependent bird species (Whyte et al., 2015; Robichaud & Rooney, 2017).

While *P. australis* suppression through herbicide treatment has become a common management strategy (Martin & Blossey, 2013; Rohal et al., 2018), it remains unclear whether such efforts restore the nutritional quality of wetland vegetation for waterfowl. Effective tools for evaluating post-treatment vegetation recovery in terms of forage value are therefore urgently needed. Although one index, the vFQI, was originally developed to estimate the nutritional value of wetland plant communities for waterfowl (Fleming et al., 2012), its regional applicability and performance in the context of invasive species suppression have not been previously tested in the Great Lakes region.

In my thesis, I address these knowledge gaps by adapting the vFQI to reflect the most common wetland plant species in southern Ontario and by introducing a new index, the WMWCs, to evaluate forage value in diverse, dynamic plant communities. The study was conducted in the Long Point and Big Creek NWAs, where two years of vegetation and one year of sediment data were collected from transects representing *P. australis*-invaded, treated, and native vegetation zones. The research objectives were to: (1) develop regionally-calibrated forage value coefficients through expert elicitation; (2) evaluate how the vFQI and Weighted Mean Waterfowl-forage Coefficient (WMWCs) reflect vegetation changes following *P. australis* suppression and assess how these indices correlate with empirical seed availability data from sediment cores. Together, these efforts aim to inform more effective monitoring and management of wetland habitat quality for migratory waterfowl in the face of ongoing invasive species pressure.

4.2 Thesis summary

In the first data chapter, I developed regionally-calibrated forage value coefficients for 74 wetland plant taxa common to southern Ontario by conducting an expert elicitation survey. This survey gathered input from 22 experts across governmental, non-governmental, and academic institutions. Experts evaluated the forage value of each taxon for waterfowl and rated their confidence in each score, allowing for the quantification of uncertainty in responses. Median values were used to calculate final coefficients, ensuring robustness in the face of ordinal data and variable expert agreement. Patterns in expert scores were analyzed in relation to confidence levels, existing literature, and available functional trait data. While moderate associations were found between expert scores and certain plant traits (e.g., dispersal unit dry mass, leaf nitrogen content), limited agreement was observed with direct energetic metrics such as True Metabolizable Energy (TME), reinforcing the idea that waterfowl forage preferences are shaped by ecological, morphological, and behavioral factors. This chapter demonstrated that expert elicitation is a practical, cost-effective method for generating forage value coefficients, and that these coefficients can serve as a standardized resource for evaluating habitat quality across the Great Lakes region.

In the second data chapter, I evaluated how forage quality varies across wetland vegetation types affected by *P. australis* invasion and suppression in the Long Point and Big Creek NWAs. I applied the vFQI and a newly developed index, the WMWCs, to compare forage value across 48 transects representing reference, *P. australis*-invaded, and treated (1-year and 2-years post-herbicide) sites. While the vFQI did not detect significant differences among vegetation types, the WMWCs revealed a clear pattern of ecological recovery: the lowest scores were observed in *P. australis*-dominated sites, followed by gradual improvement in treated sites, with the highest scores found in reference vegetation. This suggests that the vFQI may underestimate forage quality in biodiverse or transitional communities due to its sensitivity to species richness, while the WMWCs provides a more stable and interpretable measure of forage value under dynamic restoration conditions. Additionally, I compared both index values to empirical data from soil core samples collected in 2023 but found no strong correlation with seed and tuber mass. This may be due to seasonal variation, localized seed dispersal, and limitations in core sampling resolution. Nonetheless, this chapter demonstrated that the WMWCs offers a promising tool for tracking management outcomes in wetland ecosystems where plant composition is changing and biodiversity varies.

4.3 Research implications and applications

The findings of my research have direct applications for wetland managers, conservation practitioners, and policy makers involved in invasive species control and habitat restoration. In particular, the indices allow managers to evaluate whether treated areas are improving habitat value for waterfowl from a foraging perspective, using standard vegetation monitoring protocols that are already widely implemented (Gray et al., 2013; Gonsalves & Cartwright, 2016).

More broadly, these results align waterfowl habitat assessment with cross-system practice in rangelands, ungulate ecology, and fisheries, where expert elicitation is routinely paired with field data to turn incomplete evidence into management-ready guidance. The southern Ontario coefficients and the WMWCs lower barriers to adoption in Great Lakes wetlands and offer a template for region-specific extensions beyond southern Ontario.

I make several key contributions to the development, interpretation, and application of vegetation-based indices for assessing waterfowl forage value in wetland ecosystems. First, by integrating expert confidence scores into the elicitation process, this research advances our understanding of how experts assign forage value to plants. The observed relationships between forage scores, confidence levels, and plant traits indicate that expert assessments are shaped not solely by direct energetic content, but also by a broader suite of ecological and functional attributes. Based on expert comments, these include seed accessibility, seasonal availability, and prevalence in known waterfowl diets. As such, this research strengthens the argument that the forage indices based on expert-derived coefficients, such as the vFQI, may capture more than energetic potential alone, potentially offering a richer, ecologically informed metric for evaluating plant community quality for waterfowl.

Second, I address important limitations in the original vFQI formulation (Fleming et al., 2012; Farley et al., 2022). Specifically, I demonstrate that the index's formulation makes it sensitive to species richness, which can under-represent scores in more diverse systems or during early successional stages of post-treatment recovery. This may partially explain why past studies using the vFQI sometimes reported weak or inconsistent correlations with food availability or waterfowl use (e.g., Fleming et al., 2015). The alternative index introduced in this thesis, the WMWCs, was designed to reduce this sensitivity by applying a weighted-abundance approach modeled after the Floristic Quality Assessment framework (Kutcher & Forrester, 2018; Spyreas, 2019). In doing so, it was more sensitive to differences in vegetation between *P. australis*-invaded and reference sites and was able to detect changes in vegetation following herbicide treatment. By comparing the performance of both indices, this research contributes to a more nuanced understanding of which tools are appropriate in different wetland management contexts. In particular, the findings suggest that the vFQI may remain useful in ecosystems with stable plant community richness, but where management actions are anticipated to increase biodiversity, using the vFQI may under-detect improvements in waterfowl forage value. In these cases, I recommend the WMWCs as a more robust alternative for evaluating dynamic and transitional communities undergoing active restoration.

One more benefit of these indices is that they are based on plant frequency data, which is commonly collected in routine monitoring, making them easy to integrate with existing long term monitoring programs. They can even be calculated retroactively on historic vegetation datasets. This also opens the possibility of retrospective analyses that evaluate changes in forage value over time, using historical data collected before and after management interventions. Furthermore, these indices provide a more accessible and cost-effective alternative to sediment core sampling or energetic laboratory analyses, which often require specialized equipment, expertise, and financial resources (Dugger et al., 2007; Hagy et al., 2011; Lancaster et al., 2019).

4.4 Future work and recommendations

This thesis addresses key knowledge gaps in the assessment of wetland forage quality for waterfowl by refining existing tools (e.g., the vFQI) and introducing a new vegetation-based index (WMWCs) shaped to southern Ontario's wetland plant communities. It also demonstrates how expert opinions can capture both direct (e.g., seed mass, palatability) and indirect (e.g. accessibility, predation risk) dimensions of forage value. These insights lay the foundation for a more robust and ecologically-grounded assessment of wetland habitat quality under invasive species pressure. At the same time, the work opens several promising avenues for future research and tool development.

First, I tried comparing coefficients to TME scores, plant traits, and even empirically relating index scores to seed mass available for foraging waterfowl, but the best validation of this approach to waterfowl forage evaluation would be to directly compare index scores to waterfowl foraging behavior. So, future studies could integrate field observations of waterfowl presence or foraging behavior in each vegetation type, similar to what Fleming et al., (2015) have done in the past. Comparing bird use data with vegetation indices and sediment core contents would offer a more direct link between habitat quality metrics and actual waterfowl habitat use, strengthening the ecological validity of both vFQI and WMWCs.

Second, a larger dataset of experimentally-derived TME values would allow for a more thorough comparison between expert-derived forage coefficients and direct measures of energetic value. At present, the limited availability of TME data, and the variability in TME between different plant parts (e.g., seed, tubers, and leaves), make it difficult to identify a strong agreement. Future work could explore how best to aggregate or select TME values, whether using averages across plant parts or focusing solely on the most relevant edible components for waterfowl.

Third, this study suggests that further exploration of associations between forage coefficients and plant functional traits is warranted. In particular, additional data on traits related to belowground biomass, such as tuber mass, would be valuable, given the importance of tubers for waterfowl foraging. Currently, such traits are underrepresented in trait databases, limiting their inclusion in broader analyses (Laliberte, 2017).

Fourth, extended, multi-year monitoring of vegetation on all vegetation types (i.e., reference, *P. australis*-invaded, and treated) would help assess how biodiversity and forage value change over longer recovery timelines (e.g., 5–10 years). Similarly, sediment core sampling could be repeated across multiple seasons and years to evaluate temporal variation in seed abundance and composition. These repeated measurements could clarify how vegetation indices like vFQI and $WMWC_S$ reflect actual food availability over time and under different management strategies, and whether seedbank composition shifts following herbicide treatment and associated changes in plant community structure.

Fifth, identification analysis of seeds extracted from sediment cores would provide insight into whether different vegetation types support distinct seed assemblages, and how these assemblages change following *P. australis* suppression. This could also serve as a validation tool for expert-derived coefficients, particularly if the most abundant seeds in cores align with high-ranking forage species.

Finally, further testing of the WMWCs in other types of wetlands, particularly those under different management regimes (e.g., drawdowns, planting programs, passive restoration), would help assess its broader utility. If the WMWCs consistently captures meaningful differences across management types, it may prove valuable as a decision-support tool for wetland managers aiming to optimize forage value in support of migratory waterfowl conservation.

While further research is needed to fill these knowledge gaps and continue refining vegetation-based forage indices, the work presented in this thesis represents an important step toward improving how we assess habitat quality for waterfowl in restored and invaded wetlands. By combining expert elicitation, vegetation surveys, and sediment core analysis, this research advances our understanding of how plant communities contribute to food availability and offers practical tools to support more adaptive and evidence-based wetland management across southern Ontario and beyond.

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Appendices

Appendix A

Table A1-1. Seeds or vegetative parts of plant taxa consumed by 18 waterfowl species and one secretive marsh bird, identified through 13 peer-reviewed studies (1992–2022)

№	Genus	Waterfowl species (Breeding/Migration/Wintering)		Status	Types of food	
		Scientific name	**Common name**		**Scientific name**	**Common name**
1	*Anas*	*Anas acuta*	Northern Pintail	Migration [a]	*Bolboschoenus maritimus* [g] *Echinochloa walteri* [f] *Leersia oryzoides* [f] *Lepidium latifolium* [g] *Polygonum pensylvanicum* [f]	Alkali Bulrush Walter's Millet Rice Cutgrass Perennial Pepperweed Pennsylvania Smartweed
2	*Anas*	*Anas crecca*	Green-winged Teal	Migration [a]	*Ammannia* spp. [m] *Ceratophyllum demersum* [h] *Cyperus* spp. [m] *Echinochloa crus-galli* [c] *Eleocharis* spp. [e] *Eleocharis palustris* [c] *Elodea canadensis* [h] *Lepidium latifolium* [g] *Leptochloa* spp. [m] *Najas flexilis* [h] *Panicum* spp. [m] *Persicaria amphibia* [c] *Persicaria bicornis* [c] *Polygonum aviculare* [c] *Polygonum* spp. [m] *Potamogeton* spp. [h, m] *Rumex obtusifolius* [c]	Redstems Hornwort Flat Sedge Barnyard Grass Spikerush Common Spikerush Canadian Waterweed Perennial Pepperweed Sprangletops Slender Naiad Panic Grass Water Smartweed Pink Smartweed Common Knotgrass Knotweed or Knotgrass Pondweed Bitter Dock

					Vallisneria americana [h] *Zizania* spp. [h]	Wild Celery Wild Rice
3	*Anas*	*Anas platyrhynchos*	Mallard	Breeding [a] Migration [a] Wintering [a]	*Abutillon* spp. [l] *Alisma* spp. [l] *Amaranthus* spp. [l] *Bidens* spp. [l] *Bolboschoenus maritimus* [g] *Carex* spp. [l] *Cephalanthus* [l] *Ceratophyllum* spp. [l] *Ceratophyllum demersum* [h, k] *Chara* spp. [h, l] *Chenopodium album* [g] *Cyperus* spp. [l] *Echinochloa* spp. [i] *Echinochloa colona* [f] *Echinochloa crus-galli* [f, l] *Echinochloa walteri* [f] *Eleocharis* spp. [e, i, l] *Eleocharis palustris* [g] *Elodea canadensis* [h, k] *Eragrostis* spp.[l] *Helenium* spp.[l] *Leersia* spp. [i, l] *Leersia oryzoides* [f] *Lemna* spp. [l] *Lepidium latifolium* [g] *Myriophyllum spicatum* [k] *Najas* spp. [l] *Najas flexilis* [h] *Najas guadalupensis* [k] *Panicum* spp. [l] *Panicum dichotomiflorum* [f] *Polygonum* spp. [i, l] *Polygonum lapathifolium* [f] *Polygonum pensylvanicum* [f]	Indian Mallow Water-plantains Amaranth Beggarticks Alkali Bulrush Sedge Buttonbush Coontail or Hornwort Hornwort Muskgrass Lamb's Quarters Nutsedge Barnyard Grass Deccan Grass Barnyard Grass Walter's Millet Spikerush Common Spikerush Canadian Waterweed Lovegrass Sneezeweed Cutgrass Rice Cutgrass Duckweed Perennial Pepperweed Water Milfoil Naiad Slender Naiad Common Water Nymph Panic Grass Fall Panic Grass Knotweed or Knotgrass Pale Persicaria Pennsylvania Smartweed

					Potamogeton spp. [h,i,l] *Rumex* spp.[l] *Sagittaria* spp.[l] *Sagittaria latifolia* [f] *Schoenoplectus maritimus* [g] *Setaria* spp.[l] *Setaria lutescens* [f] *Sparganium* spp. [i,l] *Stuckenia pectinata* [k] *Vallisneria americana* [h,k] *Vitis* spp. [l] *Zizania* spp. [h]	Pondweed Dock Duck Potato Duck Potato Alkali Bulrush Foxtail Pigeon Grass Burreed Sago Pondweed Wild Celery Grapevine Wild Rice
4	*Anas*	*Anas rubripes*	American Black Duck	Breeding [a] Migration [a] Wintering [a]	*Ceratophyllum demersum* [h] *Chara* spp. [h] *Elodea canadensis* [h] *Najas flexilis* [h] *Potamogeton* spp. [h] *Vallisneria americana* [h] *Zizania* spp. [h]	Hornwort Muskgrass Canadian Waterweed Slender Naiad Pondweed Wild Celery Wild Rice
5	*Aythya*	*Aythya affinis*	Lesser Scaup	Migration [a]	*Amaranthus* spp. [l] *Bidens* spp. [l] *Cephalanthus* [l] *Ceratophyllum* spp.[l] *Chara* spp. [h] *Chenopodium* spp. [l] *Cyperus* spp. [l] *Echinochloa crus-galli* [l] *Eragrostis* spp.[l] *Ipomea* spp.[l] *Juncus* spp.[l] *Leersia* spp.[l] *Ludwigia* spp.[l] *Myriophyllum spicatum* [l] *Najas* spp. [l] *Polygonum* spp. [l] *Potamogeton* spp. [l]	Amaranth Beggartick Buttonbush Coontail or Hornwort Muskgrass Goosefoot Nutsedge Barnyard Grass Lovegrass Morning Glory Rushes Cutgrass Primrose or Seedbox Water Milfoil Naiad Knotweed or Knotgrass Pondweed

					Stuckenia pectinata [h] *Vallisneria americana* [h]	Sago Pondweed Wild Celery
6	*Aythya*	*Aythya americana*	Redhead	Migration [a]	*Bidens* spp. [b] *Chara* spp. [h] *Leersia* spp. [b] *Polygonum* spp. [b] *Potamogeton* spp. [b] *Scirpus* spp. [b] *Stuckenia pectinata* [h] *Vallisneria americana* [a, h]	Beggarticks Muskgrass Cutgrass Knotweed or Knotgrass Pondweed Bulrush Sago Pondweed Wild Celery
7	*Aythya*	*Aythya collaris*	Ring-necked Duck	Migration [a]	*Amaranthus* spp. [l] *Ceratophyllum* spp.[l] *Chara* spp. [l] *Cyperus* spp. [l] *Echinochloa crus-galli* [l] *Leersia* spp.[l] *Ludwigia* spp.[l] *Myriophyllum* spp. [h] *Najas* spp. [l] *Panicum* spp. [l] *Polygonum* spp. [l] *Potamogeton* spp. [l] *Wolffia* spp.[l] *Vallisneria americana* [l]	Amaranth Coontail or Hornwort Muskgrass Nutsedge Barnyard Grass Cutgrass Primrose or Seedbox Water Milfoil Naiad Panic Grass Knotweed or Knotgrass Pondweed Watermeal Wild Celery
8	*Aythya*	*Aythya marila*	Greater Scaup	Migration[a]	*Stuckenia pectinata* [h] *Vallisneria americana* [h]	Sago Pondweed Wild Celery
9	*Aythya*	*Aythya valisineria*	Canvasback	Migration [a]	*Chara* spp. [h] *Stuckenia pectinata* [h] *Vallisneria americana* [a, h]	Muskgrass Sago Pondweed Wild Celery
10	*Aix*	*Aix sponsa*	Wood Duck	Breeding [a] Migration [a]	*Ceratophyllum demersum* [h] *Chara* spp. [h] *Elodea canadensis* [h] *Najas flexilis* [h] *Potamogeton* spp. [h] *Vallisneria americana* [h] *Zizania* spp. [h]	Hornwort Muskgrass Canadian Waterweed Slender Naiad Pondweed Wild Celery Wild Rice

11	*Branta*	*Branta canadensis interior*	Canada Goose	Migration [a]	Crops: corn, winter wheat, and small grains [a] *Cyperus esculentus* [f]	 Yellow Nutsedge [f]
12	*Branta*	*Branta canadensis maxima*	Giant Canada Goose	Breeding [a] Migration [a]	Crops: corn, winter wheat, and small grains [a] *Cyperus esculentus* [f]	 Yellow Nutsedge [f]
13	*Cygnus*	*Cygnus columbianus*	Tundra Swan	Migration [a]	*Elodea canadensis* [d] *Najas* spp. [d] *Vallisneria americana* [d]	Canadian Waterweed Naiad Wild Celery
14	*Cygnus*	*Cygnus olor*	Mute Swan	Breeding [a] Wintering [a]	*Callitriche hermaphroditica* [h] *Ceratophyllum demersum* [h] *Chara vulgaris* [h] *Elodea canadensis* [h] *Lemna minor* [h] *Lemna trisulca* [h] *Myriophyllum* spp. [h] *Najas flexilis* [h] *Nitella* spp. [h] *Potamogeton pusillus* [h] *Potamogeton richardsonii* [h] *Potamogeton zosteriformis* [h] *Potamogeton* spp. [h] *Sagittaria* spp. [h] *Scirpus* spp. [h] *Sparganium* spp. [h] *Stuckenia pectinata* [h] *Zizania* spp. [h]	Water Starwort Hornwort Common Stonewort Canadian Waterweed Lesser Duckweed Star Duckweed Water Milfoil Slender Naiad Stonewort Small Pondweed Richardson's Pondweed Flat-stem Pondweed Pondweed Arrowhead Bulrush Burreed Sago Pondweed Wild Rice
15	*Mareca*	*Mareca americana*	American Wigeon	Migration [a]	*Ceratophyllum demersum* [h] *Chara* spp. [h] *Eleocharis* spp. [e] *Elodea canadensis* [h] *Najas flexilis* [h] *Potamogeton* spp. [h] *Stuckenia pectinata* [h] *Vallisneria americana* [h] *Zizania* spp. [h]	Hornwort Muskgrass Spikerush Canadian Waterweed Slender Naiad Pondweed Sago Pondweed Wild Celery Wild Rice

16	*Mareca*	*Mareca strepera*	Gadwall	Migration [a]	*Alisma* spp. [l] *Amaranthus* spp. [l] *Carex* spp. [l] *Cephalanthus* spp. [l] *Ceratophyllum* spp. [l] *Ceratophyllum demersum* [k] *Chara* spp. [l] *Cyperus* spp. [l] *Digitaria* spp. [l] *Echinochloa crus-galli* [l] *Eleocharis* spp. [l] *Elodea canadensis* [k] *Eragrostis* spp. [l] *Helenium* spp. [l] *Leersia* spp. [l] *Lemna* spp. [l] *Ludwigia* spp. [l] *Myriophyllum* spp. [l] *Myriophyllum spicatum* [k] *Najas* spp. [l] *Najas guadalupensis* [k] *Panicum* spp. [l] *Poaceae* spp. [l] *Polygonum* spp. [l] *Potamogeton* spp. [l] *Rhynchospora* spp. [l] *Sagittaria* spp. [l] *Stuckenia pectinata* [k] *Vallisneria americana* [k] *Wolffia* spp. [l]	Water-plantains Amaranth Sedge Buttonbush Coontail or Hornwort Hornwort Muskgrass Nutsedge Crabgrass Barnyard Grass Spikerush Canadian Waterweed Lovegrass Sneezeweed Cutgrass Duckweed Primrose or Seedbox Milfoil Water Milfoil Naiad Common Water Nymph Panic Grass Grasses Knotweed or Knotgrass Pondweed Beak-rush Duck Potato Sago Pondweed Wild Celery Watermeal
17	*Oxyura*	*Oxyura jamaicensis*	Ruddy Duck	Migration [a]	*Eleocharis* spp. [e]	Spikerush
18	*Spatula*	*Spatula discors*	Blue-winged Teal	Breeding [a] Migration [a]	*Alisma* spp. [l] *Amaranthus* spp. [l] *Bidens* spp. [f, l] *Carex* spp. [i, l] *Cephalanthus* spp. [l]	Water-plantains Amaranth Beggartick Sedge Buttonbush

					Chenopodium spp. [l]	Goosefoot
					Cyperus spp. [i, l]	Nutsedge
					Digitaria spp. [l]	Crabgrass
					Echinochloa spp. [i]	Barnyard Grass
					Echinochloa crus-galli [f, l]	Barnyard Grass
					Echinochloa walteri [f]	Walter's Millet
					Eleocharis spp. [i, l]	Spikerush
					Eragrostis spp. [l]	Lovegrass
					Leersia spp. [i, l]	Cutgrass
					Lemna spp. [l]	Duckweed
					Ludwigia spp. [l]	Primrose or Seedbox
					Panicum spp. [l]	Panic Grass
					Panicum dichotomiflorum [f]	Fall Panic Grass
					Polygonum spp. [i, l]	Knotweed or Knotgrass
					Polygonum pensylvanicum [f]	Pennsylvania Smartweed
					Potamogeton spp. [i, l]	Pondweed
					Rhynchospora spp. [l]	Beak-rush
					Rumex spp. [l]	Dock
19	*Porzana*	*Porzana carolina*	Sora [n, a]		*Polygonum* spp. [j]	Knotweed or Knotgrass

[a] - (Prince et al., 1992);
[b] - (Kenow & Rusch, 1996);
[c] - (Anderson et al., 2000);
[d] - (Petrie et al., 2002);
[e] - (Bogiatto & Karnegis, 2007);
[f] - (Brasher et al., 2007);
[g] - (Dugger et al., 2007);
[h] - (Bailey et al., 2008);
[i] - (Tidwell et al., 2013);
[j] - (Wilson et al., 2018);
[k] - (Gross et al., 2020);
[l] - (Hitchcock et al., 2021);
[m] - (Klimas et al., 2022b);

Appendix B

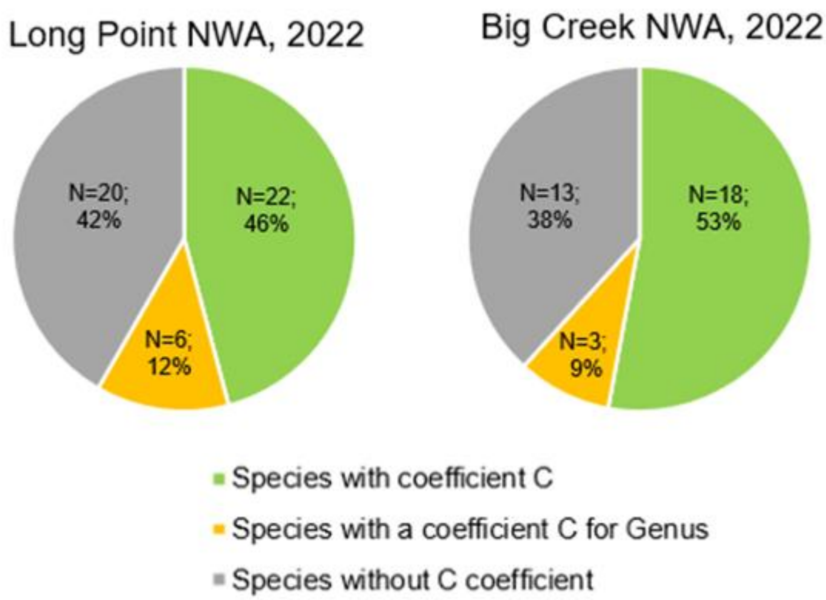
Table B1-1. List of available TME index values found during the literature search.

Scientific name	Common name	№ Birds	TME (kcal/g)						SE	p	Author
			Mallard	Blue-winged teal	Northern Pintail	Canada goose	Gadwall	Lesser Scaup			
Amaranthus spp.	Pigweed	7	2.97	-	-	-	-	-	0.19	$p > 0.05$	(Checkett et al., 2002)
Echinochloa crus-galli	Wild Millet	4	2.61	-	-	-	-	-	0.12	$p > 0.05$	
Digitaria ischaemum	Little Hairy Crabgrass	5	3.1	-	-	-	-	-	0.07	$p > 0.05$	
Digitaria sanguinalis	Hairy Crabgrass	7	3.09	-	-	-	-	-	0.02	$p > 0.05$	
Panicum dichotomum	Fall Panicum	7	2.75	-	-	-	-	-	0.10	$p > 0.05$	
Rumex crispus	Curly Dock	7	2.68	-	-	-	-	-	0.05	$p > 0.05$	
Setaria lutescens	Yellow Foxtail	7	2.88	-	-	-	-	-	0.06	$p > 0.05$	
Paspalum laeve	Paspalum	6	1.57	-	-	-	-	-	0.08	$p < 0.05$	
Polygonum lapathifolium	Nodding Smartweed	7	1.52	-	-	-	-	-	0.09	$p < 0.05$	
Rhynchospora corniculata	Beakrush	7	1.86	-	-	-	-	-	0.07	$p < 0.05$	
Chenopodium album	Lamb's Quarters	7	2.52	-	-	-	-	-	0.08	-	(Dugger et al., 2007)
Eleocharis palustris	Spike Rush	7	0.5	-	-	-	-	-	0.08	-	
Lepidium latifolium	Perennial Pepperweed	5	1.31	-	-	-	-	-	0.09	-	
Schoenoplectus maritimus	Alkali Bulrush	7	0.65	-	-	-	-	-	0.08	-	
Bidens spp.	Beggartick	-	-	0.55	-	-	-	-	-	-	(Brasher et al., 2007)
Cyperus esculentus (tubers)	Yellow Nutsedge	-	-	-	-	4.03	-	-	-	-	
Echinochloa colonum	Jungle Rice	-	2.54	-	-	-	-	-	-	-	
Echinochloa crus-galli	Wild Millet	-	2.61	2.67	-	-	-	-	-	-	
Echinochloa walteri	Coast Barnyard Grass	-	2.86	2.67	2.82	-	-	-	-	-	
Leersia oryzoides	Rice Cutgrass	-	3.00	-	2.82	-	-	-	-	-	
Panicum dichotomiflorum	Fall Panic Grass	-	2.75	2.54	-	-	-	-	-	-	
Polygonum lapathifolium	Pale Persicaria	-	1.52	-	-	-	-	-	-	-	
Polygonum pensylvanicum	Pennsylvania Smartweed	-	1.08	1.59	1.25	-	-	-	-	-	
Sagittaria latifolia (tubers)	Duck Potato	-	3.06	-	-	-	-	-	-	-	
Setaria lutescens	Yellow Foxtail	-	2.88	-	-	-	-	-	-	-	
Ruppia maritima [a]	Widgeongrass	6	1.23	-	-	-	-	-	0.38	-	(Lancaster et al., 2019)
Ruppia maritima [b]	Widgeongrass	6	0.83	-	-	-	-	-	0.32	-	
Ceratophyllum demersum	Coontail	M-8; G-18	1.51	-	-	-	0.55	-	M-0.28;G-0.28	-	(Gross et al., 2020)
Elodea canadensis	Canadian Waterweed	M-13; G-20	1.66	-	-	-	0.7	-	M-0.26;G-0.31	-	

Myriophyllum spicatum	Eurasian Watermilfoil	M-12; G-19	-0.13	-	-	-	0.77	-	M-0.42;G-0.32	-	
Najas guadalupensis	Southern Naiad	M-14; G-17	1.37	-	-	-	-0.61	-	M-0.39;G-0.34	-	
Stuckenia pectinata	Sago Pondweed	M-14; G-20	0.50	-	-	-	-1.07	-	M-0.22;G-0.33	-	
Vallisneria americana	Wild Celery	M-11; G-20	0.05	-	-	-	-0.98	-	M-0.42;G-0.39	-	
Echinochloa crus-galli	Wild Millet	125	-	-	-	-	-	2.20	-	-	(Larson et al., 2024)

[a] - Basin Collection Method; [b] - Harness Collection Method; For Gross et al., 2020 in column “№ Birds” M - Mallard, G - Gadwall.

Appendix C



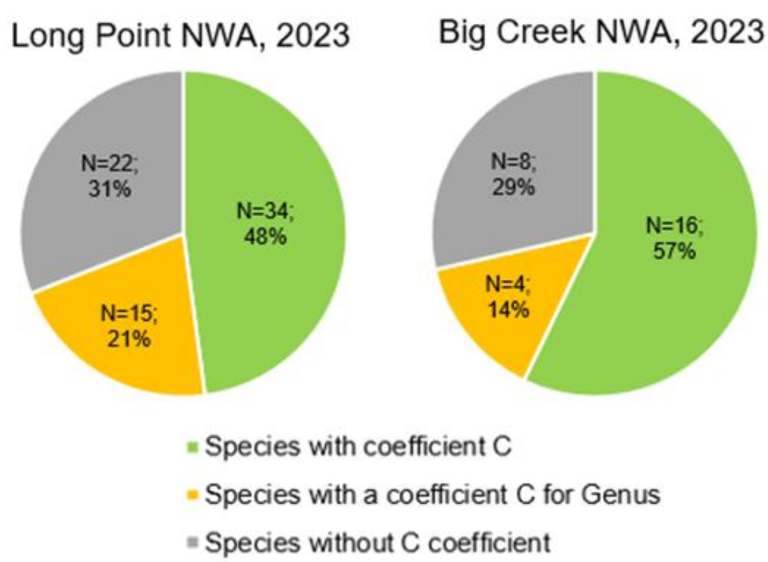
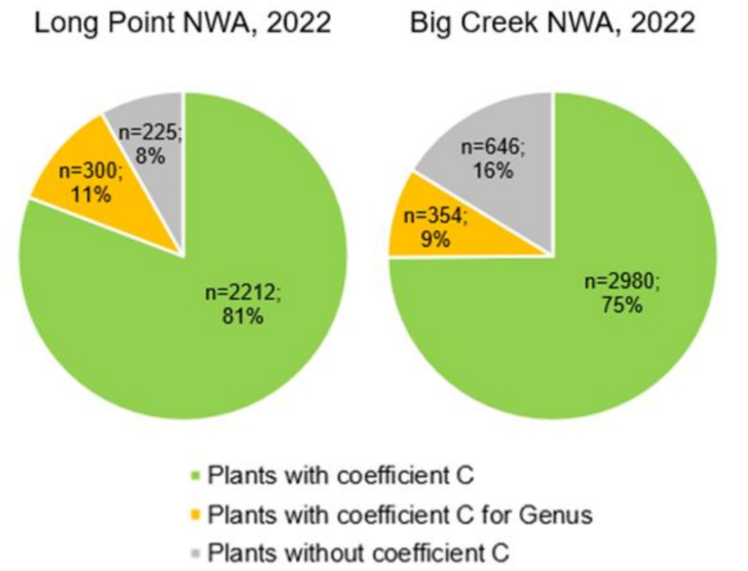


Figure C1-1., C1-2. Ratio of vFQI coefficient data availability for plant species observed in Big Creek NWA or Long Point NWA in August 2022 and 2023. 'N' indicates the number of species observed.



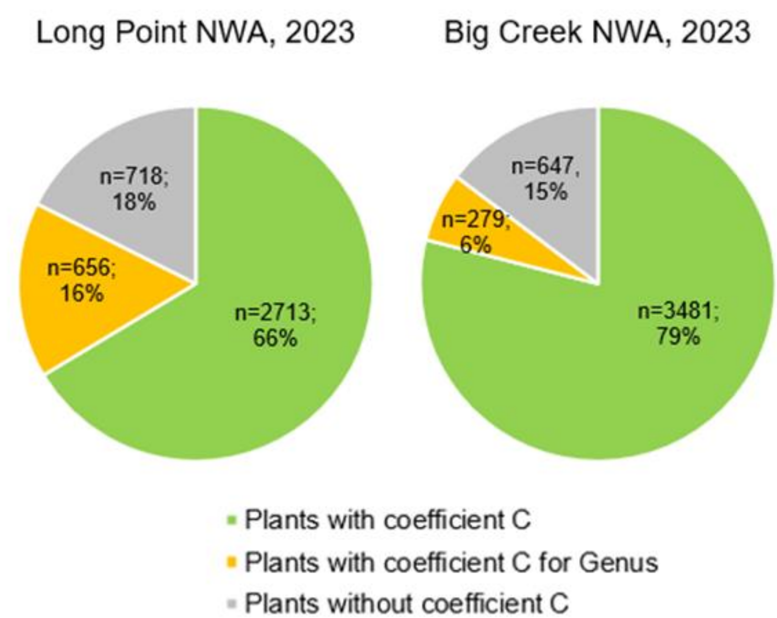


Figure C2-1., C2-2. Ratio of plants with vFQI coefficients compared to those without or those where I could apply a coefficient for another species within the same genus from August 2022 and 2023. 'n' indicates the number of interceptions of that species on the transects.

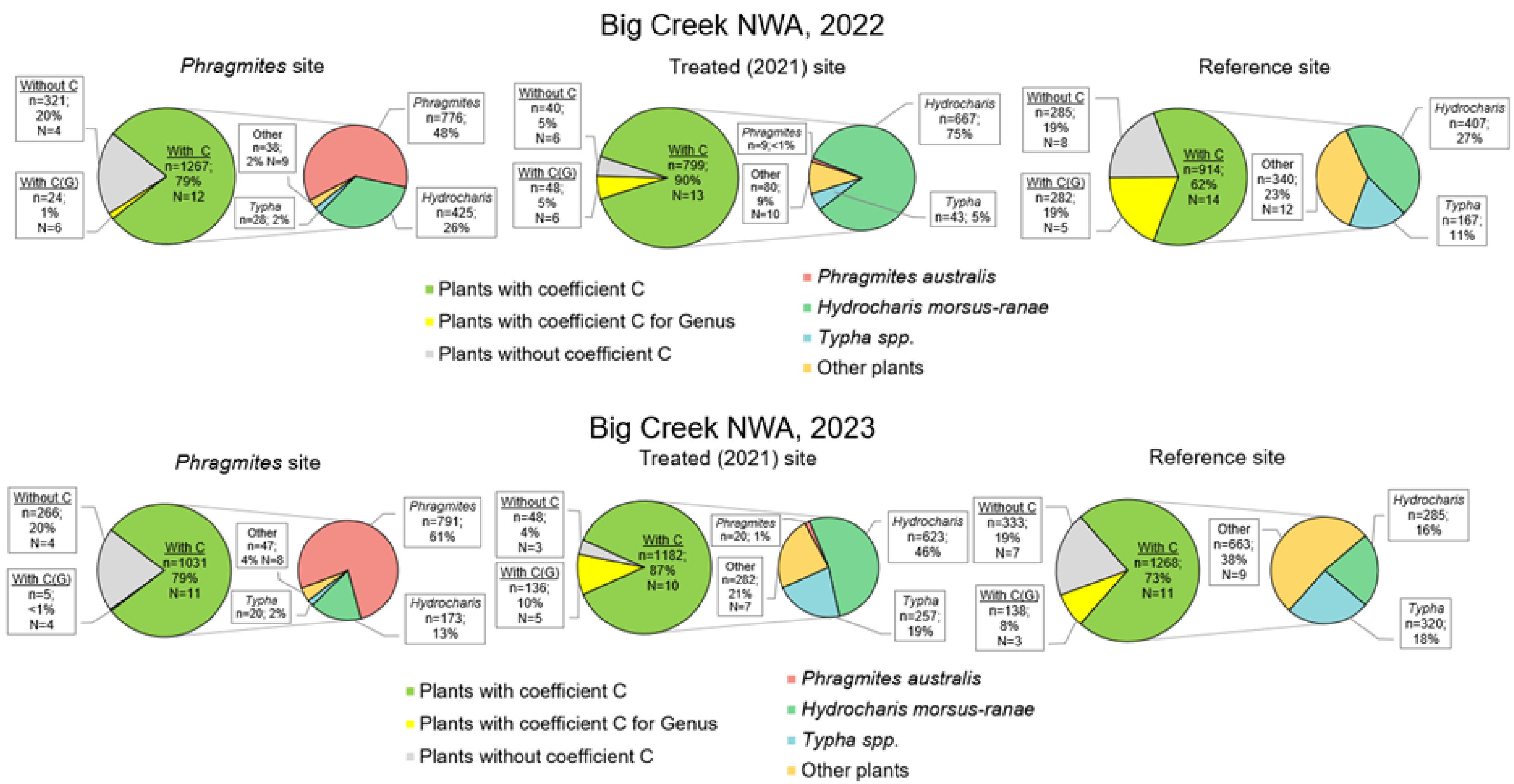


Figure C3-1. Plants intercepted on vegetation transects in Big Creek NWA in August 2022 (top) and 2023 (bottom). Species are divided into three groups: those with vFQI coefficients, those with no coefficients, and those for which coefficients from other species in the same genus could be applied. Individual plants with coefficients are further broken down by species to illustrate the proportion of individual plants with coefficients that are invasive or native. 'n' indicates the cumulative number of plants intercepted belonging to each category. All percentages refer to the relative abundance of the category on the transect, including the percentages from the inset pie (i.e., percentages in the inset pies do not add up to 100%, rather they add up to the total % in the class of individuals intercepted that possess a coefficient value).

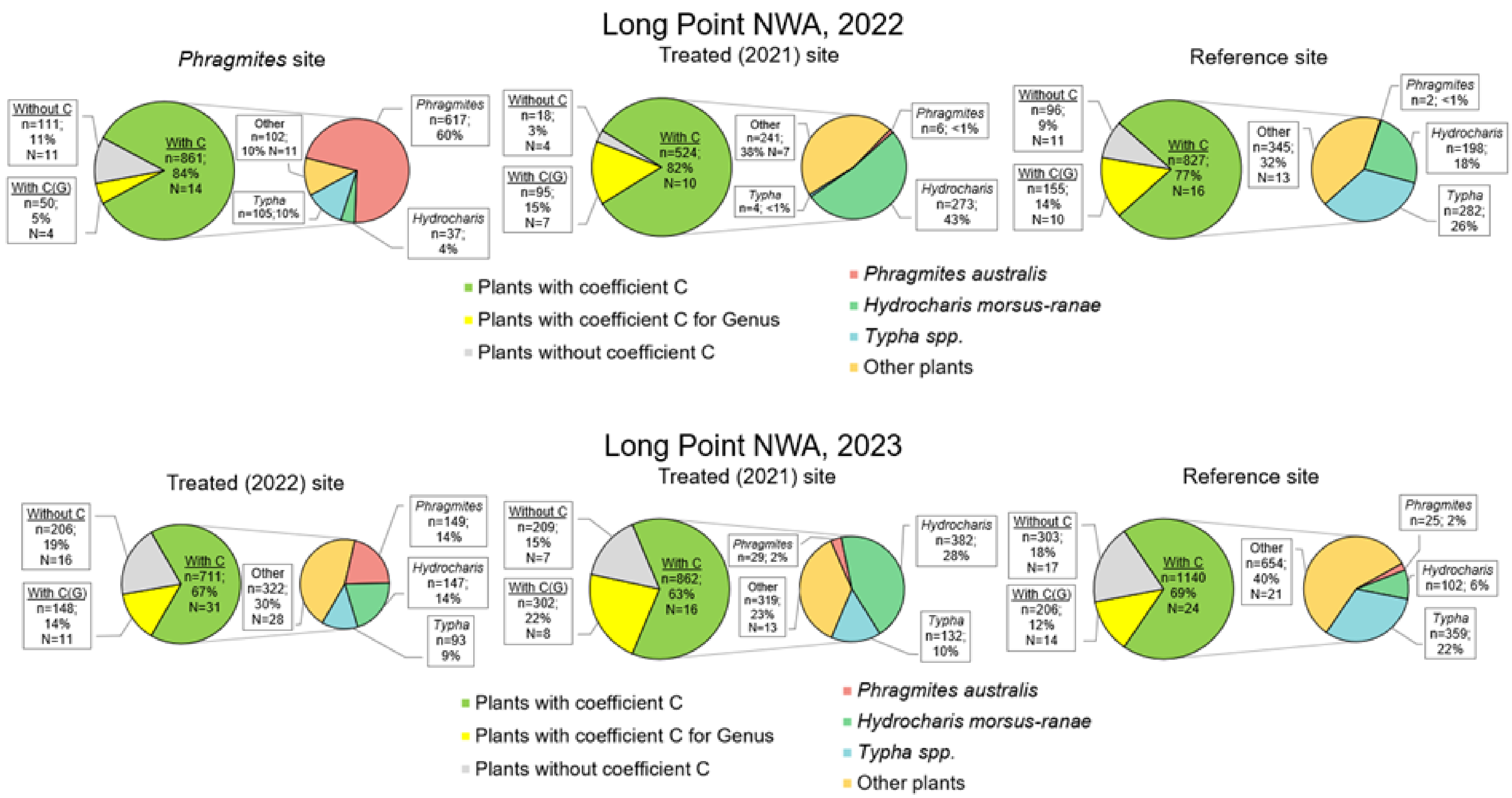


Figure C4-1. Plants intercepted on vegetation transects in Long Point NWA in August 2022 (top) and 2023 (bottom). Species are divided into three groups: those with vFQI coefficients, those with no coefficients, and those for which coefficients from other species in the same genus could be applied. Individual plants with coefficients are further broken down by species to illustrate the proportion of individual plants with coefficients that are invasive or native. 'n' indicates the cumulative number of plants intercepted belonging to each category. All percentages refer to the relative abundance of the category on the transect, including the percentages from the inset pie (i.e., percentages in the inset pies do not add up to 100%, rather they add up to the total % in the class of individuals intercepted that possess a coefficient value).

Appendix D

To identify which plant taxa to include in my expert elicitation survey, I compiled a list of the 74 most common taxa from 46 coastal marsh monitoring sites (Table D1-1) belonging to the Great Lakes Coastal Wetland Monitoring Program (CWMP). These 46 sites were distributed across Ecoregion 7E of Ontario (Figure D1-1).

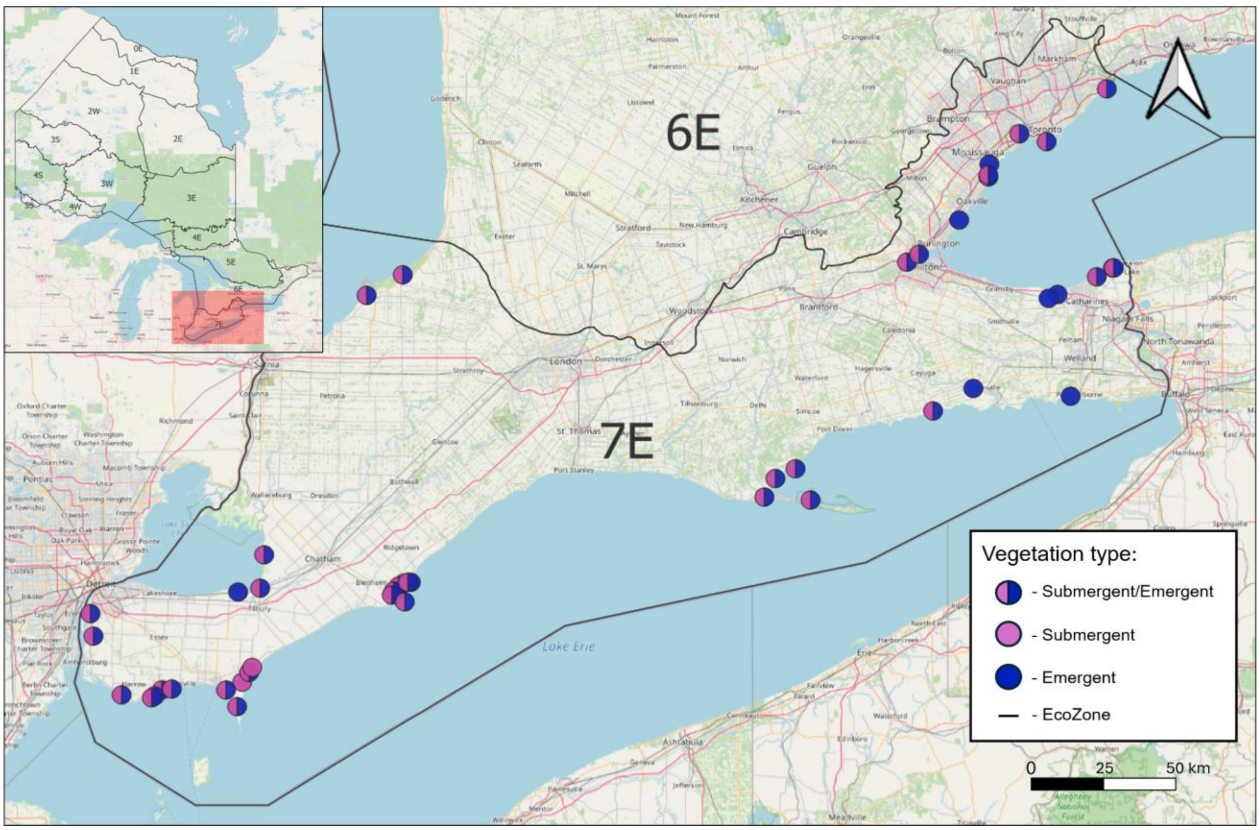


Figure D1-1. Locations of the 46 monitoring sites within Ecoregion 7E (Crins et al., 2009) in which the CWMP surveyed plant community composition. This map was created in qGIS3.8 (2023).

Most of these wetlands have been surveyed multiple times by the CWMP, which aims to resurvey 1014 coastal wetlands on the Laurentian Great Lakes at least once every five years on a rotating schedule (EPA, 2017). While the entire protocol for these surveys is detailed in the CWMP Standard Operating Procedures, I will describe them in brief. Vegetation surveys are conducted in early summer (June or July) to ensure optimal plant identification, minimize seasonal variability, and account for stable water levels. This period also aligns with peak growth for many species, allowing for reliable assessments of foliar cover, species composition, and invasive plant presence. According to Great Lakes CWMP Standard Operating Procedures, vegetation surveys are conducted along three transects which extend parallel to the water depth gradient and cross major wetland vegetation zones including wet meadow, emergent vegetation, and submerged vegetation, when present (CWMP, 2018). Each transect begins at a randomly selected starting point at the upland or swamp forest edge, with quadrats systematically placed along the transect to ensure representative sampling of plant communities. Fifteen 1 m^2 quadrats are surveyed per vegetation zone, resulting in a minimum of 15 quadrats per transect (when only one vegetation zone is present) and a maximum of 45 (when all three zones occur along the transect). All quadrats are positioned 2 m to the right of the transect line to minimize disturbance from field crew movement. Within each quadrat, the foliar cover of all plant species is visually estimated, and additional environmental variables, such as water depth and substrate type, are recorded. To standardize sampling, all field personnel receive training and certification before data collection begins each year, ensuring consistency and accuracy across study sites. Additionally, quality control measures, such as duplicate quadrat sampling and expert verification, are implemented to validate species identifications and percent cover estimates.

I requested data from CWMP for all research sites within Ecoregion 7E where vegetation biodiversity had been surveyed since 2015, ensuring that the dataset was representative. Some requested sites were excluded from the final dataset due to not meeting the criteria for Great

Lakes coastal wetlands (e.g., insufficient lake connectivity, small size, etc.) or logistical constraints, such as limited access for vegetation survey teams or sampling capacity restrictions. I refined the dataset by excluding sites with data only for the wet meadow zone, resulting in 30 riverine wetlands, 12 lacustrine (coastal) wetlands, and 4 barrier (protected) wetlands. Among these 46 sites, 37 were emergent and submerged, 7 were emergent, and 2 were submerged. These emergent and submerged vegetation zones are critical for waterfowl, especially for dabbling ducks, as they provide key foraging habitats. These seasonally or permanently flooded areas support diverse aquatic vegetation and invertebrates, forming essential food resources (Merendino et al., 1995; NRCS & USFWS, 2007; Ezell et al., 2009). They also contribute to seedbank formation and serve as vital habitat, ensuring suitable conditions for feeding and survival. For each site, I selected biodiversity data from the most recent year of observations (Figure D2-1) to account for potential shifts in plant diversity driven by external factors such as climate change, invasive species encroachment, wetland management practices, and herbicide application.

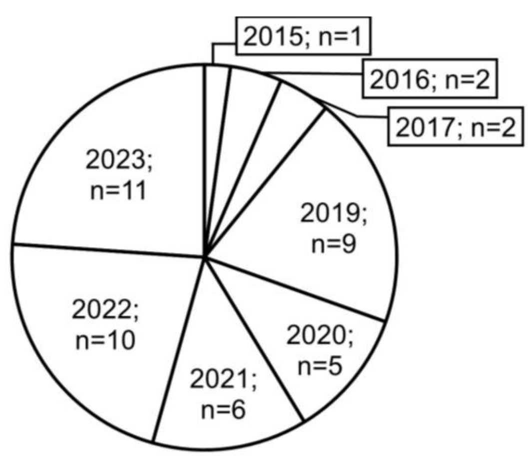


Figure D2-1. Distribution of 46 CMWP research sites (n) by most recent plant diversity survey year.

I aggregated plant biodiversity data separately for submerged and emergent vegetation zones across all sites. To determine the most common species within each zone, I calculated the frequency for each plant taxon, which was defined as the number of quadrats in which the taxon occurred in all transects. Additionally, for each taxon, I determined the maximum percent cover - the greatest percent coverage for the taxon among all quadrats in all transects. I chose this approach instead of calculating the total percent cover in all quadrats of all transects for each taxon to avoid distortion of the data due to a high presence of invasive species. I then used the taxon's frequency and percent cover to calculate a combined index for each plant taxon:

$$combined\ index_i = \frac{a_i\ + b_i}{2}$$

Where:

a_i- Normalized frequency, or ratio of the occurrence frequency of an individual taxon to the highest frequency in the dataset;

b_i- Normalized percentage of cover, or ratio of the maximum percentage of cover of an individual taxon to the maximum highest percentage of cover in the dataset.

The value of the combined index ranges from 0 to 1, where the closer the value is to 1, the more widespread and abundant the plant taxon is. I ranked a total of 153 plant taxa for the emergent vegetation zone, and 65 taxa for the submerged zone according to their widespread presence using the combined index. I filtered and excluded most taxa with previously-established coefficients for calculating the vFQI, except for 30 plants that I used to determine the correlation between my results and coefficients from previous studies (i.e., Fleming et al., 2012; Farley et al., 2022). Instead, I added species or genera without coefficients that the Waterloo Wetland Laboratory observed during fieldwork in the Long Point and Big Creek NWAs in August 2022 and 2023 that were not already reflected in the CWMP data (Table D1-1).

Table D1-1. List of species or genera of plants encountered in the Big Creek NWA or the Long Point NWA in August 2022 and 2023 that lack vFQI coefficients. Y (yes) **-** indicates that the plant taxon was observed in Big Creek NWA or Long Point NWA field surveys in the given year; and N (no) - indicates it was not observed during the field surveys. * - indicates that the value of the coefficient of the genus to which the species belongs was used for this species.

Genus	**Species**	**Common name**	**2022**	**2023**	**Coefficient**
Alisma	*subcordatum*	American Water Plantain	N	Y	2.43
Ambrosia	spp.	Ragweed	N	Y	1.86
Apocynum	spp.	Dogbane	Y	Y	
Asclepias	spp.	Milkweed	N	Y	1
Barbarea	*vulgaris*	Wintercress	Y	N	
Bidens	spp.	Beggartick	Y	Y	2.86
Boehmeria	*cylindrica*	False Nettle	Y	Y	
Bolboschoenus	*fluviatilis*	River Bulrush	N	Y	1.6
Butomus	*umbellatus*	Flowering Rush	N	Y	
Calystegia	*sepium*	Hedge Bindweed	N	Y	1.33*
Carex	*spp.*	Sedge	Y	Y	2.29
Ceratophyllum	*demersum*	Coontail	Y	Y	2.57
Chara	spp.	Stonewort	Y	Y	2.5
Cicuta	*douglasii*	Water Hemlock	N	Y	
Convolvulus	spp.	Bindweed	Y	N	
Cuscuta	spp.	Dodder	N	Y	
Cyperus	spp.	Nutsedge	Y	Y	2.54
Decodon	*verticillatus*	Swamp Loosestrife	Y	Y	
Eleocharis	spp.	Spike-rush	Y	N	2.57
Eleocharis	*acicularis*	Needle Spikerush	Y	N	2.57*
Eleocharis	*obtusa*	Blunt Spikerush	N	Y	2.57*
Elodea	*canadensis*	Canadian Waterweed	Y	Y	2.33
Epilobium	spp.	Willowherb	Y	Y	1
Erechtites	spp.	Burnweed	N	Y	
Eupatorium	*perfoliatum*	Common Boneset	Y	Y	1.17
Heteranthera	*dubia*	Water Stargrass	Y	Y	
Hydrocharis	*morsus-ranae*	European Frog-bit	Y	Y	1
Impatiens	*capensis*	Spotted Jewelweed	Y	Y	1
Iris	spp.	Iris	Y	Y	
Juncus	spp.	Rush	N	Y	2.14
Lathyrus	*palustris*	Marsh Pea	Y	N	

Genus	Species	Common name	2022	2023	Coefficient
Leersia	*oryzoides*	Rice Cutgrass	N	Y	3.57
Lemna	*minor*	Lesser Duckweed	Y	N	1.77*
Lycopus	*americanus*	American Water Horehound	Y	Y	1.33*
Lythrum	*salicaria*	Purple Loosestrife	N	Y	1
Melilotus	spp.	Sweet Clover	N	Y	
Mimulus	*ringens*	Allegheny Monkeyflower	N	Y	1
Myriophyllum	*sibiricum*	Northern Milfoil	Y	Y	2
Myriophyllum	*spicatum*	Eurasian Milfoil	Y	Y	1.67
Myriophyllum	*verticillatum*	Whorled Milfoil	N	Y	2*
Najas	*flexilis*	Nodding Waternymph	Y	Y	
Nuphar	*lutea*	Yellow Pond Lily	Y	Y	
Nymphaea	*odorata*	American White Water Lily	Y	Y	
Persicaria	spp.	Knotweeds	Y	N	2.14
Persicaria	*amphibia*	Water Smartweed	Y	Y	2.14*
Persicaria	*hydropiper*	Water Pepper	N	Y	3.43*
Persicaria	*lapathifolia*	Pale Smartweed	N	Y	3.43*
Persicaria	*maculosa*	Lady's Thumb	Y	N	3.43*
Phragmites	*americanus*	Common Reed	Y	N	
Phragmites	*australis*	Common Reed	Y	Y	1
Pilea	*pumila*	Canadian Clearweed	Y	N	
Pontederia	*cordata*	Pickerelweed	Y	Y	2.29
Potamogeton	*crispus*	Curly leaf Pondweed	Y	Y	2.86
Potamogeton	*foliosus*	Leafy Pondweed	Y	Y	2.62*
Potamogeton	*natans*	Floating Pondweed	Y	Y	2.62*
Potamogeton	*richardsonii*	Richardson's Pondweed	N	Y	2.62*
Potamogeton	*zosteriformis*	Flat-stem Pondweed	Y	Y	2.62*
Rorippa	spp.	Yellowcress	N	Y	
Rorippa	*palustris*	Bog Yellowcress	N	Y	
Sagittaria	spp.	Arrowhead	Y	Y	3.43
Sagittaria	*latifolia*	Broadleaf Arrowhead	Y	N	3.43*
Salix	spp.	Willow	Y	Y	1.17
Schoenoplectus	*acutus*	Hardstem Bulrush	Y	Y	2.17
Schoenoplectus	*pungens*	Three-square Bulrush	Y	Y	
Schoenoplectus	*tabernaemontani*	Softstem Bulrush	Y	Y	
Scirpus	spp.	Bulrush	Y	N	2.08
Scutelleria	spp.	Skullcap	Y	N	1.20
Solanum	*dulcamara*	Bittersweet Nightshade	N	Y	1
Solidago	spp.	Goldenrod	Y	Y	1
Sparganium	spp.	Burreed	Y	Y	3.14
Sparganium	*eurycarpum*	Giant Bur-reed	Y	N	3.14
Spartina	spp.	Cordgrass	N	Y	
Spartina	*maritima*	Small Cordgrass	Y	N	
Spartina	*pectinata*	Prairie Cordgrass	Y	N	
Spirodela	*polyrhiza*	Common Duckweed	Y	N	

Genus	Species	Common name	2022	2023	Coefficient
Strophostyles	*helvola*	Trailing Fuzzybean	N	Y	
Stuckenia	*pectinata*	Sago Pondweed	Y	Y	3.57
Thelypteris	*palustris*	Marsh Fern	Y	Y	
Typha	spp.	Cattail	Y	Y	1
Utricularia	*minor*	Lesser Bladderwort	N	Y	1.83
Utricularia	*vulgaris*	Greater Bladderwort	Y	Y	1.83
Valisneria	*americana*	Eelgrass	Y	Y	3.57
Verbena	*hastata*	Blue Vervain	Y	Y	1.2
Zizania	*palustris*	Northern Wild Rice	Y	Y	

Appendix E

To assemble the expert panel, I engaged experts recommended by governmental and non-governmental organizations related to waterfowl conservation and habitat preservation, as well as authors of studies that have been involved in developing the assessment of the nutritional value of wetland plants as food for waterfowl (Table E1-1).

Table E1-1. List of governmental, non-governmental and international institutions that were contacted regarding potential experts.

Category	Country	Name
Governmental institutions	Canada	- Canadian Wildlife Service (CWS) - Environment and Climate Change Canada (ECCC) - Parks Canada - Ministry of Natural Resources and Forestry Ontario (MNRF) - Ministry of Environment Conservation and Parks Ontario (MECP) - Ontario Parks
	USA	- Natural Resources Conservation Service (NRCS) - Environmental Protection Agency USA (EPA) - Fish and Wildlife Service (FWS) - United States Geological Survey (USGS) - Foreign Agricultural Service (USDA FAS)
Non-governmental institutions	Canada	- Ducks Unlimited Canada - Nature Canada - Nature Conservancy of Canada (NCC) - Birds Canada - Canadian Wildlife Federation (CWF) - Long Point Waterfowlers' Association - Wildlife Habitat Canada (WHC) - Delta Waterfowl Foundation Canada - Conservation Ontario - Eastern Habitat Joint Venture - Society of Canadian Ornithologist (SCO-SOC)
	USA	- Ducks Unlimited US - Partners in Flight - National Audubon Society - American Birding Association (ABA) - Cornell Lab of Ornithology - American Bird Conservancy (ABC) - Ornithological Council

Category	Country	Name
		- Delta Waterfowl Foundation US - The National Wildlife Federation (NWF) - Sustain Our Great Lakes (SOGL) - Ornithology Exchange - Upper Mississippi/Great Lakes Joint Venture
International non-governmental institutions	-	- North American Bird Conservation Initiative (NABCI) - Commission for Environmental Cooperation (CEC)

In total, the overall response rate during the expert search phase was approximately 40% for governmental organizations, 47% for non-governmental organizations, and 74% for scientific authors (Table E2-1). While contacting these institutions, I also relied on a snowball sampling strategy, anticipating that initial contacts would refer me to additional qualified experts.

Table E2-1. Response rate during the expert identification phase

Category	Country	Contacted	Replied	No Reply	Response Rate (%)
Governmental Institutions	Canada	10	4	6	40
	USA	21	8	13	38
Non-Governmental Organizations	Canada	9	6	3	67
	USA	13	4	9	31
Thematic Publication Authors	-	27	20	7	74

For governmental institutions in Canada and the United States of America, USA, the number of contacts exceeded the number of organizations listed in Table E1-1 because I reached out not only to federal or provincial ministries, but also to specific branches and departments directly involved in wetland and waterfowl management. In the USA, this included contacting both the central departments and regional offices of relevant agencies in each of the Great Lakes states. Responses from governmental institutions typically fell into several categories: some informed me that they had shared my message internally with relevant staff; others stated that they did not have appropriate experts; several redirected me to other government bodies; a number referred me to non-governmental experts or organizations; and some replies contained no actionable information. Replies from non-governmental organizations were generally more targeted: they often provided contacts for other experts or organizations and suggested relevant forums or thematic networks. Among authors of relevant publications, most responses were constructive - some offered expert referrals, while others explained that they were retired or not currently engaged in this area of research.

Following the expert identification phase, I compiled a list of 38 experts considered suitable for the survey, and the questionnaire was sent to all of them. Of those contacted, 29 replied to the initial message, and 22 eventually completed the survey, resulting in a 58% participation rate (Table E3-1).

Table E3-1. Response outcomes during the expert elicitation phase

Stage	Number of Experts (n)	Total Identified Experts (%)
Identified as potential experts	38	100
Responded to email	29	76
Completed the survey	22	58

The primary reasons for non-participation included retirement, a busy schedule during the survey period, or a perceived lack of expertise in the relevant plant taxa. Nevertheless, many of those who declined to participate still contributed valuable input by referring other potential experts or providing relevant literature and methodological resources.

Appendix F

I designed the expert survey to establish forage value coefficients for common wetland plants in southern Ontario's Great Lakes coastal wetlands and conducted it using Google Forms (Google LLC., n.d.). Experts were asked to evaluate plant taxa based on their nutritional contribution to waterfowl forage, including considerations of metabolizable energy, plant abundance, and accessibility for waterfowl. At the beginning of the survey, participants provided their name and email address to allow for tracking completion. However, all responses remained confidential. I presented each plant taxon on a separate page with a corresponding photo (Figure F1-1). I asked experts to assess each plant's nutritional and energetic value, as well as its availability to waterfowl (e.g., productivity, abundance), using a four-point scale: 1 - poor, 2 - fair, 3 - good, and 4 - excellent. Additionally, experts were asked to indicate confidence in their assessment on a separate scale, where 1 - not confident, 2 - slightly confident, 3 - fairly confident, and 4 - completely confident. This was necessary to examine whether expert ratings varied with confidence levels and whether certain plant taxa tended to receive lower confidence ratings due to factors such as limited research availability or genuinely low forage value for waterfowl. Additionally, an optional comment box was included on each page, allowing experts to provide remarks or insights. To assist experts in their evaluations, a link to the USDA Plants Database was provided (USDA & NRCS, 2024), enabling them to access detailed species descriptions, morphological traits, and habitat information. However, no additional information or specific characteristics of each plant species were provided within the survey itself to avoid influencing expert judgment. This approach ensured that experts could focus on the aspects they deemed most relevant in their assessments. Experts were also encouraged to use any additional sources they found relevant.

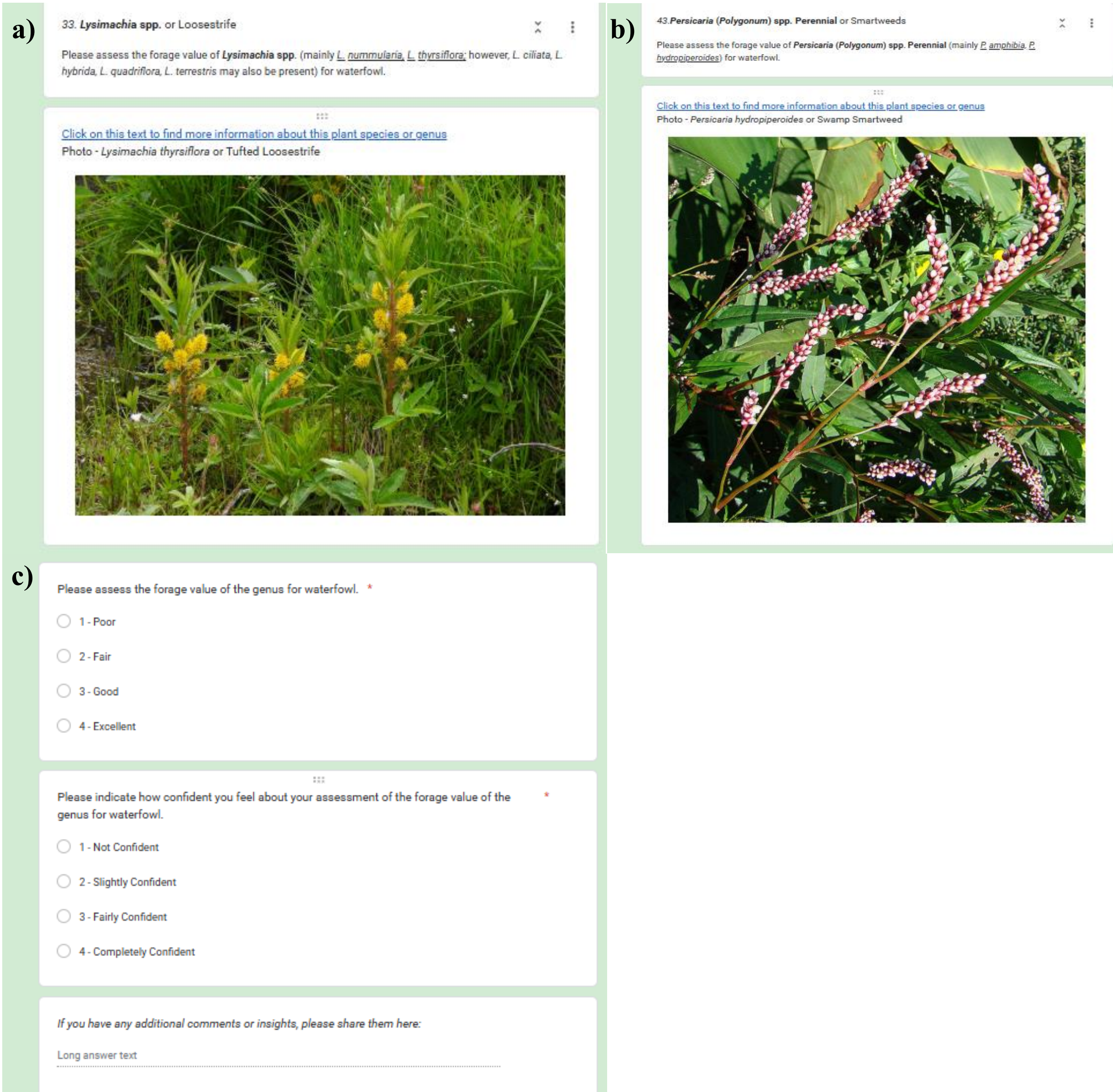


Figure F1-1. Screenshots from the expert survey in Google Forms, including a) taxon description, photograph and link to the USDA Plants Database for *Lythrum salicaria*; b) taxon description, photograph and link to the USDA Plants Database for *Persicaria hydropiper*; c) scoring interface and comments field used by experts to assign forage-value ratings.

For genus-level assessments, a list of example species was provided to guide the evaluation. For example:

"Please assess the waterfowl forage value of Lysimachia *spp. (mainly* L. nummularia*,* L. thyrsiflora*, however,* L. ciliata*,* L. hybrida*,* L. quadriflora*,* L. terrestris *may also be present)."*

The distinction between "main" and "potential" species was determined using data from the CWMP, direct field observations, and distribution records from *The Fourth Edition of A Great Lakes Wetland Flora* (Chadde, 2019) and the USDA Plants Database (USDA & NRCS, 2024). I listed example species to guide experts toward the most representative and commonly occurring species within each genus in the region, helping them focus on relevant taxa rather than considering the full diversity of species within the genus. To reduce the length of the survey and optimize experts' time, species were grouped at the genus level when they shared similar key traits. This grouping was based on similarities in seed quantity, the presence and abundance of tubers, and life cycle type (e.g., annual vs. perennial), as determined using the USDA Plants Database (USDA & NRCS, 2024) and *A Great Lakes Wetland Flora, Fourth Edition* (Chadde, 2019). For example, *Echinochloa walteri, Echinochloa muricata,* and *Echinochloa crus-galli* were merged into *Echinochloa* spp., as their forage value for waterfowl was expected to be approximately the same. This approach simplified the survey by reducing the need for experts to distinguish between closely-related species within a constrained scoring system, ultimately streamlining the survey to 74 species or genera and decreasing the time required for completion, allowing experts to focus their efforts more efficiently (Table F1-1).

Table F1-1. List of plant taxa included in the expert survey.

№	Scientific Name	Common Name	Notes
1	*Acer saccharinum*	Silver Maple	
2	*Amphicarpaea bracteata*	American Hog Peanut	
3	*Apios americana*	Common Groundnut	
4	*Apocynum* spp.	Dogbane	*A. cannabinum, A. androsaemifolium*
5	*Barbarea* spp.	Winter Cress	Mainly *B. vulgaris*; however, *B. orthoceras, B. stricta, B. verna* may also be present
6	*Bidens* spp.	Beggartick	*B. cernua, B. connata, B. discoidea, B. frondosa, B. beckii*
7	*Boehmeria cylindrica*	False Nettle	
8	*Butomus umbellatus*	Flowering Rush	
9	*Calamagrostis* spp.	Reedgrass	Mainly *C. canadensis*; however, *C. stricta* may also be present
10	*Carex* spp.	Sedges	Mainly *C. lacustris, C. comosa*; however, *C. aquatilis, C. brunnescens* may also be present
11	*Ceratophyllum demersum*	Coontail	
12	*Chara* spp.	Muskgrass or Stonewort	
13	*Cirsium* spp.	Thistles	Mainly *C. muticum*; however, *C. arvense, C. palustre* may also be present
14	*Cuscuta* spp. (annual)	Dodder	Mainly *C. gronovii*; however, *C. cephalanthi, C. epilinum* may also be present
15	*Cyperus* spp. (perennial)	Perennial Sedge	Mainly *C. esculentus, C. strigosus*; however, *C. houghtonii, C. lupulinus* may also be present
16	*Cyperus* spp. (annual)	Flat Sage	Mainly *C. erythrorhizos, C. diandrus, C. engelmannii*; however, *C. acuminatus, C. bipartitus, C. flavescens, C. squarrosus* may also be present
17	*Daucus carota*	Queen Anne's Lace	
18	*Decodon verticillatus*	Swamp Loosestrife	
19	*Echinochloa* spp.	Wild Millets	Mainly *E. walteri, E. muricata, E. crus-galli*; however, *E. frumentacea* may also be present
20	*Eleocharis* spp.	Spike Rushes	Mainly *E. erythropoda, E. acicularis, E. palustris, E. intermedia, E. obtusa*;

№	Scientific Name	Common Name	Notes
			however, *E. compressa, E. elliptica* may also be present
21	*Erechtites* spp.	Fireweeds	Mainly *E. hieraciifolius*
22	*Eutrochium maculatum*	Joe-Pye Weed	
23	*Glyceria* spp.	Mannagrass	Mainly *G. grandis, G. maxima*; however, *G. canadensis, G. borealis, G. septentrionalis, G. striata* may also be present
24	*Heteranthera dubia*	Water Star-grass	Mainly *H. dubia*; however *H. limosa* may also be present
25	*Hibiscus* spp.	Rose Mallow	Mainly *H. moscheutos*; however, *H. laevis, H. trionum* may also be present
26	*Hydrocharis morsus-ranae*	European Frog-bit	
27	*Iris* spp.	Iris or Flag	Mainly *I. pseudacorus, I. versicolor*; however, *I. virginica* may also be present
28	*Juncus* spp.	Rushes	Mainly *J. effusus, J. torreyi*; however, *J. albescens, J. compressus*, etc. may also be present
29	*Justicia americana*	American Water-willow	
30	*Lathyrus* spp. (perennial)	Wild Pea	Mainly *L. palustris*
31	*Leersia oryzoides*	Rice Cutgrass	
32	*Lemna* spp.	Duckweed	Mainly *L. minor, L. trisulca*; however, *L. perpusilla, L. valdiviana* may also be present
33	*Lysimachia* spp.	Loosestrife	Mainly *L. nummularia, L. thyrsiflora*; however, *L. ciliata, L. hybrida, L. quadriflora, L. terrestris* may also be present
34	*Melilotus* spp. (annual)	Melilot	Mainly *M. alba*; however, *M. officinalis* may also be present
35	*Myosotis* spp.	Forget-me-not	Mainly *M. arvensis, M. scorpioides*; however, *M. laxa* may also be present
36	*Najas* spp.	Water-nymphs	Mainly *N. flexilis, N. guadalupensis, N. minor*; however, *N. marina, N. gracillima* may also be present
37	*Nitellopsis obtusa*	Starry Stonewort	
38	*Nuphar* spp. (perennial)	Yellow Water Lily	Mainly *N. advena, N. lutea*; however, *N. variegata* may also be present
39	*Nymphaea* spp.	Water Lily	Mainly *N. odorata*; however, *N. leibergii* may also be present

№	Scientific Name	Common Name	Notes
40	*Panicum* spp.	Panic Grass	Mainly *P. miliaceum, P. virgatum*
41	*Parthenocissus quinquefolia*	Virginia Creeper	
42	*Persicaria (Polygonum)* spp. (annual)	Smartweed	Mainly *P. punctata, P. sagittata, P. lapathifolia, P. maculosa*
43	*Persicaria (Polygonum)* spp. (perennial)	Smartweed	Mainly *P. amphibia, P. hydropiperoides*
44	*Phalaris arundinacea*	Reed Canarygrass	
45	*Phragmites* spp.	Common Reed	
46	*Pilea* spp.	Clearweed	Mainly *P. pumila*; however, *P. fontana* may also be present
47	*Pontederia cordata*	Pickerelweed	
48	*Potamogeton* spp.	Pondweed	Mainly *P. crispus, P. foliosus, P. friesii, P. gramineus. P. illinoensis, P. natans, P. nodosus, P. pusillus, P. richardsonii, P. zosterifirmis;* however, *P. alpinus, P. perfoliatus*, etc. may also be present
49	*Potentilla* spp.	Cinquefoil	Mainly *P. anserina;* however, *P. fruticosa, P. palustris, P. paradoxa, P. rivalis* may also be present
50	*Quercus* spp.	Oak	Mainly *Q. alba*
51	*Ribes* spp.	Gooseberry	Mainly *R. americanum;* however, *R. cinosbati, R. glandulosum, R. hirtellum* may also be present
52	*Riccia fluitans*	Floating Crystalwort	
53	*Ricciocarpos natans*	Fringed Heartwort	
54	*Rorippa* spp. (annual)	Yellow-Cress	Mainly *R. palustris;* however, *R. sessiliflora* may also be present
55	*Sagittaria* spp.	Arrowhead/ Duck Potato	Mainly *S. latifolia, S. rigida, S. graminea;* however, *S. cuneata* may also be present
56	*Salix* spp.	Willow	Mainly *S. nigra, S. exigua,*
57	*Schoenoplectus* spp.	Bulrush	Mainly *S. pungens, S. tabernaemontani, S. acutus;* however, *S. heterochaetus, S. subterminalis, S. torreyi* may also be present
58	*Scirpus* spp.	Bulrush	Mainly *S. atrovirens, S. cyperinus, S. expansus, S. microcarpus, S. pendulus;* however*, S. atrocinctus, S. pallidus,* may also be present
59	*Setaria* spp.	Foxtail	Mainly *S. faberi, S. italica, S. viridis, S. verticillata*
60	*Sparganium* spp.	Bur Reed	Mainly *S. eurycarpum;*

№	Scientific Name	Common Name	Notes
			however, *S. americanum, S. androcladum, S. angustifolium, S. emersum, S. fluctuans, S. glomeratum, S. natans* may also be present
61	*Spartina* spp.	Cordgrass	Mainly *S. maritima, S. pectinata;* however, *S. gracilis, S. patens* may also be present
62	*Spirodela polyrhiza*	Greater Duckweed	
63	*Stachys* spp.	Hedge-nettle	Mainly *S. palustris, S. tenuifolia, S. hispida;* however, *S. hyssopifolia* may also be present
64	*Strophostyles* spp. (annual)	Wild Bean	Mainly *S. helvola;* however, *S. leiosperma* may also be present
65	*Strophostyles* spp. (perennial)	Wild Bean	Mainly *S. umbellata*
66	*Stuckenia pectinata*	Sago Pondweed	
67	*Teucrium canadense*	Canada Germander	
68	*Thelypteris palustris*	Eastern Marsh Fern	
69	*Triadenum* spp.	Marsh St. Johnswort	Mainly *T. fraseri;* however, *T. virginicum* may also be present
70	*Typha* spp.	Cattail	
71	*Utricularia* spp.	Bladderwort	Mainly *U. gibba, U. minor, U. vulgaris*
72	*Vallisneria americana*	Wild Celery	
73	*Wolffia* spp.	Watermeal	Mainly *W. columbiana;* however, *W. borealis* may also be present
74	*Zizania* spp.	Wild Rice	Mainly *Z. palustris, Z. aquatica*

To encourage experts to complete the full survey - which required a significant time commitment - I incorporated gamification elements into the Google Forms interface. Specifically, after every ten evaluated plant taxa, participants encountered a motivational screen featuring the silhouettes of "hungry ducks." Each set of ten evaluations “unlocked” a silhouette and revealed the image of a specific duck species. In total, experts could unlock eight species, culminating in a congratulatory screen with full-colour illustrations upon completing all 74 evaluations (Figure F2-1).

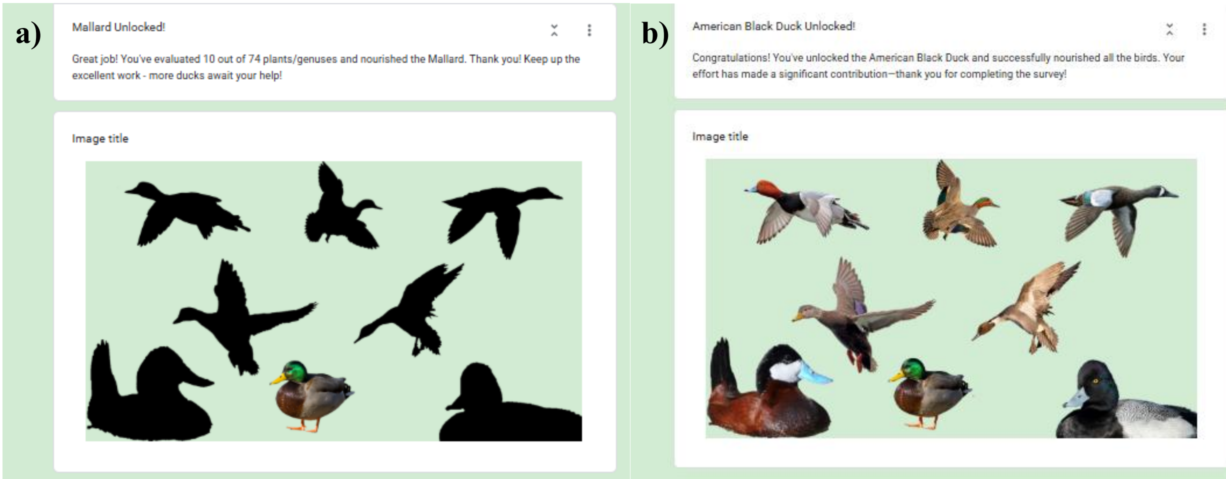


Figure F2-1. Gamification elements in the expert elicitation survey, shown as two panels: a) motivational slide displayed after completing ten plant evaluations, unlocking the Mallard image; b) final congratulatory slide displayed upon completing all 74 evaluations, unlocking the American Black Duck image.

Appendix G

Initially, I prepared a request for trait data from the TRY database for 135 plant species representing 72 genera. These species were selected based on their inclusion in the expert survey above. To maximize the likelihood of obtaining trait data for each genus, where possible, I included multiple species per genus. The species list comprised plants observed during fieldwork, previously identified through the CWMP data analysis, or reported as present in the region according to *A Great Lakes Wetland Flora, Fourth Edition* (Chadde, 2019). Additionally, I identified 50 functional traits relevant to the nutritional value of plant species and their specific organs, such as leaves, roots, and belowground structures (Table G1-1).

Table G1-1. List of functional traits potentially reflecting the forage value of wetland plants for waterfowl, used for data requests to the TRY Global Database.

Trait group	Trait ID	Trait
Dispersal	193	Dispersal distance
	890	Dispersal frequency
	28	Dispersal syndrome
	233	Dispersal unit dry mass
Leaf	13	Leaf carbon (C) content per leaf dry mass
	146	Leaf carbon/nitrogen (C/N) ratio
	151	Leaf carbon/phosphorus (C/P) ratio
	14	Leaf nitrogen (N) content per leaf dry mass
	56	Leaf nitrogen/phosphorus (N/P) ratio
	15	Leaf phosphorus (P) content per leaf dry mass
	3857	Leaf protein content per area: proteomic analysis
Root/Rhizome	84	Root carbon (C) content per root dry mass
	1055	Root carbon/nitrogen (C/N) ratio
	3517	Root carbon/phosphorus (C/P) ratio
	80	Root nitrogen (N) content per root dry mass
	687	Root nitrogen/phosphorus (N/P) ratio
	683	Root phosphorus (P) content per root dry mass
Seeds	2944	Seed carbon (C) content per dry mass (or diaspore)
	3636	Seed coat thickness
	26	Seed dry mass
	2945	Seed nitrogen (N) content per dry mass (or diaspore)
	336	Seed number per ramet

Trait group	Trait ID	Trait
	138	Seed number per reproduction unit
	1105	Seed number per square meter
	1106	Seed number per stem
	96	Seed oil content per seed mass
	3000	Seed phosphorus (P) content per dry mass (or diaspore)
	97	Seed protein content per seed mass
	3585	Seed starch content per seed dry mass
	1111	Seedbank density
	2808	Seedbank thickness of top layer
	2809	Seedbank duration
	2807	Seedbank number of layers
	775	Shoot carbon (C) content per shoot dry mass
	409	Shoot carbon/nitrogen (C/N) ratio
Shoot	339	Shoot nitrogen (N) content per shoot dry mass
	340	Shoot phosphorus (P) content per shoot dry mass
	3609	Shoot protein content per dry mass
	407	Stem carbon (C) content per stem dry mass
	165	Stem carbon/nitrogen (C/N) ratio
Stem	406	Stem nitrogen (N) content per stem dry mass
	686	Stem nitrogen/phosphorus (N/P) ratio
	616	Stem phosphorus (P) content per stem dry mass
	2551	Belowground plant organ carbon (C) content per belowground plant organ dry mass
	2569	Belowground storage organ dry mass
	2563	Belowground plant organ dry mass per ground area
Belowground plant organ	2547	Belowground plant organ nitrogen (N) content per belowground plant organ dry mass
	2561	Belowground plant organ phosphorus (P) content per belowground plant organ dry mass
	3590	Belowground storage organ dry mass
	3591	Belowground storage organ total non-structural carbohydrate content per dry mass

After retrieving the data, I grouped them by Trait ID (Table G2-1). Traits with insufficient observations, or traits for which data were only available for a few requested species were discarded. I further processed the remaining data to ensure consistency in measurement units. I retained observations (StdValue) only if they were recorded in standardized units (StdUnit), as researchers often use different original units (OrigUnitStr) in experiments. This led to cases where data for a single trait (OrigValueStr) were reported in multiple units. For example, for the trait Seed dry mass, the standardized unit was *mg*, but original units included values in *mg*, *1/kg*, *1/pound*, *g*, *g/1000 seeds*, *g/100 seeds*, *mg/seed*, and others. I filtered out observations lacking standardized unit conversions for all traits. Next, I filtered standardized values, where I retained only observations classified as single, mean, or best estimation, while I excluded extreme values (e.g., maximum, minimum, upper/lower quartile). This ensured that extreme values did not distort the calculation of mean trait values for each species. I sorted each trait and grouped them by species. I then calculated the mean value for each species based on the remaining observations.

Table G2-1. Description of selected TRY column headers used in trait data processing

Data column header	**Definition**
Trait ID	Unique identifier for traits (only if the record is a trait)
StdValue	Standardized value: available for standardized traits
StdUnit	Standard unit: available for standardized traits
OrigUnitStr	Original unit as text string
OrigValueStr	Original value as text string
ValueKindName	Value kind (single measurement, mean, median, etc.)

In cases where certain traits were not present in the database, I estimated additional values using available data. Specifically, I calculated the Leaf carbon/nitrogen ratio (Trait 146) for *Butomus umbellatus, Calamagrostis canadensis, Hibiscus moschatus,* and *Myosotis scorpioides* using data from Leaf carbon content per leaf dry mass (Trait 13) and Leaf nitrogen content per leaf dry mass (Trait 14). Similarly, I determined the Leaf nitrogen/phosphorus ratio (Trait 56) for *Cirsium arvense, Cirsium palustre, Daucus carota, Iris pseudacorus, Persicaria amphibia, Sagittaria latifolia, Scirpus cyperinus, Setaria viridis, Stachys palustris, Stuckenia pectinata, Thelypteris palustris,* and *Utricularia vulgaris* by using values from Leaf nitrogen content per leaf dry mass (Trait 14) and Leaf phosphorus content per leaf dry mass (Trait 15).

Additionally, I estimated the Root carbon/nitrogen ratio (Trait 1055) for *Apocynum androsaemifolium* using data from Root carbon content per root dry mass (Trait 84) and Root nitrogen content per root dry mass (Trait 80). In addition, I merged Seed number per ramet (Traits 336) and Seed number per stem (1106) into a single trait, as no species in the dataset had multiple stems per ramet. I derived data for *Butomus umbellatus* from Seed number per stem (Trait 1106), while for *Stachys palustris*, I averaged the values from both traits. For Shoot carbon/nitrogen (C/N) ratio (Trait 409), I numerically converted categorical values (i.e., low, medium, high) to 1, 2, and 3, respectively, for statistical analysis. Finally, I excluded taxa without any trait data and obtained the final dataset. In accordance with the Intellectual Property Guidelines for the TRY Plant Trait Database, in addition to the general reference to the TRY Global Database (Kattge et al., 2011) listed in the References section, I also provide citations for individual data contributors and original studies from which trait records were obtained for each plant feature. The data I processed from the TRY Global Database and citations can be found at: https://doi.org/10.6084/m9.figshare.29497607.v1

Appendix H

Since my data consist of ordinal median scores on a 1-4 scale, I chose a cumulative link regression model (CLM) as a primary inferential method. Nevertheless, to maintain consistency with previous studies, I also calculated Spearman's rank correlations between obtained expert-derived coefficients and those published by Fleming et al. (2012) and Farley et al. (2022). The results showed strong, statistically significant correlations with both Fleming et al. (2012) ($\rho = 0.86$, $p < 0.001$) and Farley et al. (2022) ($\rho = 0.91$, $p < 0.001$), demonstrating broad agreement in forage value rankings (Figure H1-1).

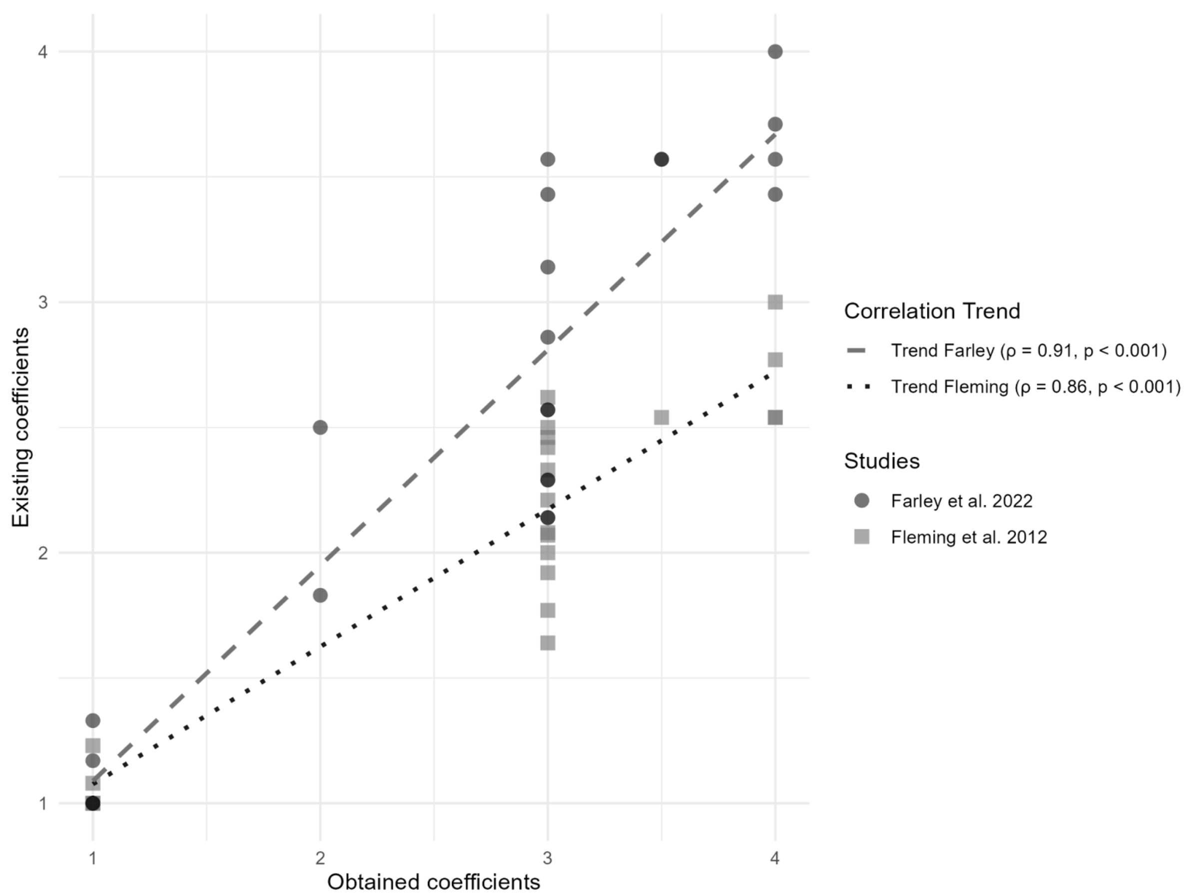


Figure H1-1. Agreement between expert-derived forage value coefficients from this study (X-axis) and those reported by Fleming et al. (2012) and Farley et al. (2022) (Y-axis).

When I compare these Spearman results to my CLM findings, both approaches point to the same pattern of agreement: the Farley et al. (2022) predictor shows a clear, significant association ($\beta = 7.43 \pm 2.71$ SE, $z = 2.74$, $p = 0.006$; 95% CI 3.63-14.95), while the Fleming et al. (2012) predictor yields a positive but marginal effect ($\beta = 18.23 \pm 10.69$ SE, $z = 1.71$, $p = 0.088$; 95% CI 6.15-54.47). Although the CLM is more appropriate for ordinal data with unequal steps, given my data and sample size, both its results and those of Spearman's rank correlation show the same trends in my data.

Appendix I

Table I1-1. Comprehensive list of feed quality coefficients (C) derived from expert opinions for 248 plant species and genera in the studies by Fleming et al. (2012), Farley et al. (2022), and my survey. For Fleming et al. (2012), and Farley et al. (2022), the table presents average ranked values for each plant, while for my study, the coefficients are represented by median values. Blank values indicate that the taxon was not evaluated in the study.

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
1	*Abutilon theophrasti* Medik.	Velvetleaf		1.67	
2	*Acer rubrum* L.	Red Maple	1.69		
3	*Acer saccharinum* L.	Silver Maple			1.50
4	*Achillea millefolium* L.	Yarrow		1.00	
5	*Agrostis* spp. L.	Bent Grass	1.20		
6	*Alisma subcordatum* Raf.	Water Plantain		2.43	
7	*Allium canadense* L.	Meadow Garlic	1.27		
8	*Alnus incana* ssp. *Rugosa* (Du Roi) R.T. Clausen	Speckled Alder		1.00	
9	*Alternanthera philoxeroides* (Mart.) Griseb.	Alligator Weed	1.10		
10	*Althaea officinalis* L.	Marshmallow		1.17	
11	*Amaranthus palmeri* S. Watson	Carelessweed	1.75		
12	*Amaranthus retroflexus* L.	Redroot Pigweed	2.15		
13	*Amaranthus* spp. L.	Pigweed		3.29	
14	*Ambrosia artemisiifolia* L.	Common Ragweed	1.77		
15	*Ambrosia* spp. L.	Ragweed		1.86	
16	*Ambrosia trifida* L.	Great Ragweed	1.50		
17	*Ammannia coccinea* Rottb.	Toothcup	2.00		
18	*Amphicarpaea bracteata* (L.) Fernald	American Hog Peanut			1.00
19	*Ampelopsis arborea* (L.) Koehne	Peppervine	1.00		
20	*Andropogon* spp. L.	Bluestem	1.15		
21	*Apios americana* Medik.	Common Groundnut			1.00
22	*Apocynum cannabinum* L.	Dogsbane	1.00	1.00	
23	*Apocynum* spp. L.	Dogbane			1.00
24	*Arrhenatherum* spp. P. Beauv.	Oatgrass	1.00		
25	*Asclepias incarnata* L.	Swamp Milkweed		1.00	
26	*Asclepias* spp. L.	Milkweed	1.00		
27	*Aster* spp. L.	Aster	1.08		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
28	*Atropa* spp. L.	Belladonna	1.00		
29	*Baccharis halimifolia* L.	Eastern Baccharis	1.00		
30	*Bacopa rotundifolia* (Michx.) Wettst.	Waterhyssop	1.60		
31	*Barbarea* spp. W.T. Aiton	Winter Cress			1.00
32	*Bidens cernua* L.	Nodding Beggartick	2.38		
33	*Bidens* spp. L.	Beggartick		2.86	3.00
34	*Boehmeria cylindrica* (L.) Sw.	False Nettle			1.00
35	*Bolboschoenus fluviatilis* (Torr.) Soják	River Bulrush		1.60	
36	*Boltonia* spp. L'Hér.	Doll's Daisy	1.27		
37	*Brasenia schreberi* J.F. Gmel.	Watershield	2.09		
38	*Brunnichia ovata* (Walter) Shinners	Redvine	1.25		
39	*Butomus umbellatus* L.	Flowering Rush			2.00
40	*Calamagrostis* spp. Adans.	Reedgrass			2.00
41	*Callitriche heterophylla* Pursh	Water-starwort	1.33		
42	*Cardiospermum halicacabum* L.	Balloon Vine	1.25		
43	*Carex* spp. L.	Sedge	2.33	2.29	3.00
44	*Celtis laevigata* Willd.	Sugarberry	1.40		
45	*Celtis occidentalis* L.	Hackberry		1.50	
46	*Cephalanthus occidentalis* L.	Common Buttonbush	1.54		
47	*Ceratophyllum demersum* L.	Coontail	2.07	2.57	3.00
48	*Chamaecrista fasciculata* (Michx.) Greene	Partridge Pea	1.60		
49	*Chara* spp. Linnaeus, 1763	Muskgrass		2.50	2.00
50	*Chenopodium album* L.	Goosefoot		2.83	
51	*Cirsium arvense* (L.) Scop.	Canadian Thistle		1.00	
52	*Cirsium* spp. Mill.	Thistle			1.00
53	*Convolvulus* spp. L.	Morning Glory		1.33	
54	*Conyza canadensis* (L.) Cronquist	Horseweed	1.10		
55	*Coreopsis* spp. L.	Tickseed	1.33		
56	*Cornus* spp. L.	Dogwood		1.40	
57	*Croton* spp. L.	Croton	1.82		
58	*Cuscuta gronovii* Willd. ex Schult.	Common Dodder		1.25	
59	*Cuscuta* spp. (annual) L.	Dodder			1.00

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
60	*Cynodon dactylon* (L.) Pers.	Bermudagrass	1.23		
61	*Cynosciadium digitatum* DC.	Finged Dog Shade	1.00		
62	*Cyperus erythrorhizos* Muhl.	Redroot Flatsedge		3.43	
63	*Cyperus esculentus* L.	Yellow Nutsedge	2.79	3.71	
64	*Cyperus* spp. (annual) L.	Flat sedge	2.54		4.00
65	*Cyperus* spp. (perennial) L.	Sedge	2.50		3.00
66	*Daucus carota* L.	Queen Anne's Lace	1.00	1.00	1.00
67	*Decodon verticillatus* (L.) Elliott	Swamp Loosestrife			2.00
68	*Desmanthus illinoensis* (Michx.) MacMill. ex B.L. Rob. & Fernald	Bundleflower	1.25		
69	*Dichanthelium ensifolium* (Baldwin ex Elliott) Gould	Cypress Panic Grass	1.88		
70	*Digitaria* spp. Haller	Crabgrass	2.00		
71	*Diodia virginiana* L.	Buttonweed	1.60		
72	*Diospyros virginiana* L.	Common Persimmon	1.18		
73	*Echinochloa* spp. P. Beauv.	Wild Millet or Barnyard Grass	3.00	4.00	4.00
74	*Echinodorus cordifolius* (L.) Griseb.	Burhead	1.78		
75	*Eclipta prostrata* (L.) L.	False Daisy	1.00		
76	*Eleocharis* spp. R. Br.	Spike Rush	2.21	2.57	3.00
77	*Elodea canadensis* Michx.	Canadian Waterweed		2.33	
78	*Elymus canadensis* L.	Wild Rye		2.50	
79	*Epilobium* spp. L.	Willowherb		1.00	
80	*Equisetum* spp. L.	Horsetail		1.25	
81	*Eragrostis* spp. Wolf	Lovegrass	2.14		
82	*Erechtites* spp. Raf.	Fireweed			1.00
83	*Eryngium prostratum* Nutt. ex DC.	Creeping Eryngo	1.29		
84	*Eupatorium capillifolium* (Lam.) Small	Dogfennel	1.00		
85	*Eupatorium perfoliatum* L.	Common Boneset		1.17	
86	*Eupatorium serotinum* Michx.	Boneset	1.08		
87	*Euphorbia humistrata* Engelm. ex A. Gray	Spreading Sandmat	1.00		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
88	*Eutrochium maculatum* (L.) E.E. Lamont	Joe-Pye Weed		1.00	1.00
89	*Forestiera acuminata* (Michx.) Poir.	Swampprivet	1.25		
90	*Fraxinus* spp. L.	Ash	1.62		
91	*Galium* spp. L.	Bedstraw		1.00	
92	*Gamochaeta purpurea* (L.) Cabrera	Purple Everlast	1.00		
93	*Glyceria canadensis* (Michx.) Trin.	Rattlesnake Mannagrass		1.67	
94	*Glyceria* spp. R. Br.	Mannagrass			2.00
95	*Gratiola neglecta* Torr.	Hedgehyssop	1.00		
96	*Heliopsis helianthoides* (L.) Sweet	Sweet Oxeye	1.50		
97	*Heliotropium* spp. L.	Helioptrope	1.25		
98	*Heteranthera dubia* (Jacq.) MacMill.	Water Star-grass			2.00
99	*Heteranthera limosa* (Sw.) Willd.	Mudplantain	1.85		
100	*Hibiscus* spp. L.	Rosemallow	1.08		1.00
101	*Hydrocharis morsus-ranae* L.	European Frogbit		1.00	1.00
102	*Hydrolea ovata* Nutt. ex Choisy	False Fiddleleaf	1.00		
103	*Ilex decidua* Walter	Holly	1.00		
104	*Impatiens capensis* Meerb.	Touch-me-not		1.00	
105	*Ipomoea* spp. L.	Morning Glory	1.15		
106	*Iris* spp. L.	Iris or Flag			1.00
107	*Iva annua* L.	Marsh Elder	1.00		
108	*Juncus effusus* L.	Soft Rush		1.71	
109	*Juncus* spp. L.	Rushes	1.92	2.14	3.00
110	*Justicia americana* (L.) Vahl	American Water-willow			1.50
111	*Lathyrus hirsutus* L.	Caley Pea	1.00		
112	*Lathyrus* spp. (perennial) L.	Wild Pea			2.00
113	*Leersia oryzoides* (L.) Sw.	Rice Cutgrass	2.54	3.57	3.50
114	*Lemna* spp. L.	Duckweed	1.77		3.00
115	*Lepidium campestre* (L.) W.T. Aiton	Field Peppercress		1.33	
116	*Leptochloa panicea* ssp. *brachiata* (Steud.) N.W. Snow	Sprangletop	2.69		
117	*Lespedeza* spp. Michx.	Lespedeza	1.50		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
118	*Liatris* spp. Gaertn. ex Schreb.	Blazingstar	1.18		
119	*Lindernia dubia* (L.) Pennell	False Pimpernel	1.22		
120	*Lolium* spp. L.	Ryegrass	1.69		
121	*Lonicera japonica* Thunb.	Honeysuckle	1.17		
122	*Ludwigia palustris* (L.) Elliott	Marsh Seedbox		1.60	
123	*Ludwigia* spp. L.	Primrose or Seedbox	1.54		
124	*Lycopus* spp. L.	Water Horehound		1.33	
125	*Lysimachia* spp. L.	Loosestrife			1.00
126	*Lythrum alatum* Pursh	Winged Loosestrife	1.27		
127	*Lythrum alatum var. lanceolatum* (Elliott) Torr. & A. Gray ex Rothr.	Southern Winged Loosestrife	1.09		
128	*Lythrum salicaria* L.	Purple Loosestrife		1.00	
129	*Melilotus alba* Medik.	White Sweet Clover	1.11		
130	*Melilotus* spp. (annual) Mill.	Melilot			1.50
131	*Mentha* spp. L.	Wild Mint		1.00	
132	*Mikania scandens* (L.) Willd.	Hemp Vine	1.18		
133	*Mimulus ringens* L.	Monkeyflower		1.00	
134	*Myosotis* spp. L.	Forget-me-not			1.00
135	*Myriophyllum aquaticum* (Vell.) Verdc.	Parrot Feather	1.70		
136	*Myriophyllum spicatum* L.	Eurasian Water - milfoil		1.67	
137	*Myriophyllum* spp. L.	Water-milfoil	2.00		
138	*Najas* spp. L.	Water-nymph			3.00
139	*Nelumbo lutea* Willd.	American Lotus	1.38		
140	*Nitellopsis obtusa* (N.A. Desvaux) J. Groves	Starry Stonewort			2.00
141	*Nuphar variegata* Durand	Yellow Pond Lily		2.00	
142	*Nuphar* spp. (perennial) Sm.	Yellow Water Lily			2.00
143	*Nymphaea* spp. L.	Water Lily			2.00
144	*Nymphoides cordata* (Elliott) Fernald	Floating Heart		1.50	
145	*Oenothera biennis* L.	Evening Primrose	1.17		
146	*Onoclea sensibilis* L.	Sensitive Fern		1.00	
147	*Osmundaceae* Martinov	Royal Fern		1.00	
148	*Panicum* spp. L.	Panic Grass	2.54	3.71	4.00
149	*Panicum virgatum* L.	Switchgrass	2.08		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
150	*Parthenocissus quinquefolia* (L.) Planch.	Virginia Creeper			1.00
151	*Paspalum* spp. L.	Bull Grass	2.46		
152	*Passiflora incarnata* L.	Purple Passionflower	1.00		
153	*Pastinaca sativa* L.	Wild Parsnip		1.00	
154	*Peltandra virginica* (L.) Schott	Arrow Arum		2.33	
155	*Penthorum sedoides* L.	Ditch Stonecrop		1.00	
156	*Persicaria pensylvanica* (L.) M. Gómez	Knotweed	2.43		
157	*Persicaria* spp. (annual) (L.) Mill.	Smartweed	2.42	3.43	3.00
158	*Persicaria* spp. (perennial) (L.) Mill.	Perennial Smartweed	2.00	2.14	3.00
159	*Phalaris arundinacea* L.	Reed Canarygrass	1.23	1.33	1.00
160	*Phragmites australis* (Cav.) Trin. ex Steud.	Common Reed		1.00	1.00
161	*Phyla lanceolata* (Michx.) Greene	Lanceleaf Fogfruit	1.50		
162	*Physalis angulata* L.	Ground Cherry	1.00		
163	*Phytolacca americana* L.	American Pokeweed	1.00		
164	*Pilea* spp. Lindl.	Clearweed			1.00
165	*Platanus occidentalis* L.	American Sycamore	1.08		
166	*Pluchea camphorata* (L) DC	Camphor Pluchea	1.00		
167	*Poa* spp. L.	Grass		1.67	
168	*Pontederia cordata* L.	Pickerelweed	1.64	2.29	3.00
169	*Populus deltoides* W. Bartram ex Marshall	Cottonwood		1.17	
170	*Populus* spp. L.	Cottonwood	1.08		
171	*Populus tremuloides* Michx.	Quaking Aspen		1.00	
172	*Potamogeton crispus* L.	Curly Leaf Pondweed	2.21	2.86	
173	*Potamogeton diversifolius* Raf.	Waterthread	2.36		
174	*Potamogeton nodosus* Poir.	Longleaf Pondweed		2.86	
175	*Potamogeton* spp. L.	Pondweed	2.62		3.00
176	*Potentilla* spp. L.	Cinquefoil			1.00
177	*Prunus* spp. L.	Plum	1.18		
178	*Quercus phellos* L.	Willow Oak	2.69		
179	*Quercus texana* Buckley	Nuttall Oak	2.55		
180	*Quercus* spp. L.	Oak			4.00
181	*Ranunculus* spp. L.	Buttercup	1.27		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
182	*Rhynchospora* spp. Vahl	Beak Rush	1.92		
183	*Ribes* spp. L.	Gooseberry			2.00
184	*Riccia fluitans* L.	Floating Crystalwort			1.00
185	*Ricciocarpos natans* (L.) Corda	Fringed Heartwort			1.50
186	*Robinia pseudoacacia* L.	Black Locust	1.08		
187	*Rorippa* spp. (annual) Scop.	Yellow-Cress			2.00
188	*Rosa palustris* Marshall	Swamp Rose	1.18		
189	*Rubus* spp. L.	Blackberry	1.09		
190	*Rudbeckia hirta* L.	Black-eyed Susan		1.20	
191	*Rumex crispus* L.	Curly Dock	2.15		
192	*Rumex* spp. L.	Dock		2.57	
193	*Saccharum giganteum* (Walter) Pers.	Plumegrass	1.20		
194	*Sagittaria* spp. L.	Arrowhead or Duck Potato	2.77	3.43	4.00
195	*Salix* spp. L.	Willow	1.00	1.17	1.00
196	*Schoenoplectus acutus* (Muhl. ex Bigelow) Á. Löve & D. Löve	Hardstem Bulrsuh		2.17	
197	*Schoenoplectus* spp. (Rchb.) Palla	Bulrush			3.00
198	*Scirpus atrovirens* Willd.	Dark Green Bulrush		2.17	
199	*Scirpus cyperinus* (L.) Kunth	Woolgrass	1.46	1.67	
200	*Scirpus* spp. L.	Bulrush	2.08		3.00
201	*Scutellaria* spp. L.	Skullcap		1.20	
202	*Senna obtusifolia* (L.) H.S. Irwin & Barneby	Sicklepod	1.09		
203	*Sesbania macrocarpa* Muhl. ex Raf.	Coffeeweed	1.31		
204	*Setaria* spp. P. Beauv.	Foxtail	2.46	3.57	3.00
205	*Sida spinosa* L.	Prickly Sida	1.27		
206	*Sinapis arvensis* L.	Wild Mustard		1.25	
207	*Solanum* spp. L.	Nightshade		1.00	
208	*Solidago gigantea* Aiton	Giant Golden Rod	1.00		
209	*Solidago* spp. L.	Goldenrod		1.00	
210	*Sorghum bicolor* (L.) Moench	Milo	2.92		
211	*Sorghum vulgare*	Sudan Grass	1.69		
212	*Sorghum halepense* (L.) Pers.	Johnson Grass	1.58		
213	*Sparganium* spp. L.	Burreed		3.14	3.00
214	*Spartina* spp. Schreb.	Cordgrass			2.00

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
215	*Spirodela polyrhiza* (L.) Schleid.	Greater Duckweed			2.00
216	*Stachys* spp. L.	Hedge-nettle			1.00
217	*Strophostyles* spp. (annual) Elliott	Wild Bean			2.00
218	*Strophostyles* spp. (perennial) Elliott	Wild Bean			2.00
219	*Stuckenia pectinata* (L.) Börner	Sago Pondweed		3.57	3.50
220	*Symphyotrichum* spp. Nees	American Aster		1.00	
221	*Taraxacum officinale* F.H. Wigg.	Dandelion		1.20	
222	*Taxodium distichum* (L.) Rich.	Bald Cypress	1.08		
223	*Teucrium canadense* L.	Canada Germander			1.00
224	*Thelypteris palustris* Schott	Eastern Marsh Fern			1.00
225	*Toxicodendron radicans* (L.) Kuntze	Poison Ivy		1.00	
226	*Triadenum* spp. Raf.	Marsh St. Johnswort			1.00
227	*Tridens strictus* (Nutt.) Nash	Grease Grass	1.56		
228	*Typha angustifolia* L.	Narrowleaf Cattail		1.00	
229	*Typha latifolia* L.	Broadleaf Cattail		1.00	
230	*Typha* spp. L.	Cattail	1.08		1.00
231	*Typha x glauca* Godr. (pro sp.)	Hybrid Cattail		1.00	
232	*Ulmus americana* L.	American Elm	1.71		
233	*Ulmus rubra* Muhl.	Slippery Elm	1.50		
234	*Urochloa platyphylla* (Munro ex C. Wright) R.D. Webster	Signalgrass	1.29		
235	*Urtica dioica* L.	Stinging Nettle		1.00	
236	*Utricularia* spp. L.	Bladderwort		1.83	2.00
237	*Vallisneria americana* Michx.	Wild Celery		3.57	4.00
238	*Vallisneria* spp. L.	Eelgrass	2.85		
239	*Verbena hastata* L.	Blue Vervain		1.20	
240	*Verbena* spp. L.	Verbena	1.25		
241	*Vicia* spp. L.	Vetch		1.00	
242	*Vitis palmata* Vahl	Grape	1.08		
243	*Vitis riparia* Michx.	Riverbank Wild Grape		1.80	
244	*Wolffia* spp. Horkel ex Schleid.	Watermeal			2.00
245	*Xanthium* spp. L.	Cocklebur	1.08		

№	Scientific Name	Common Name	(Fleming et al., 2012)	(Farley et al., 2022)	Kramarenko & Rooney
246	*Xyris* spp. L.	Yelloweyed Grass	1.25		
247	*Zea* spp. L.	Corn	2.85		
248	*Zizania* spp. L.	Wild Rice			4.00

This table also can be found here:

https://doi.org/10.6084/m9.figshare.29497682.v1

Appendix J

Table AJ1-1. Cumulative link model results testing the association between expert-assigned vFQI coefficients and 21 plant functional traits.

Trait	**Estimate (β)**	**Standard error**	**z-value**	**p-value**
Root C/N ratio (n = 8)	1.03066395	0.9472412	1.08806914	0.27656459
Seed oil content (n = 18)	-0.94140683	0.5598627	-1.68149588	0.09266664
Dispersal unit dry mass (n = 24)	0.90921959	0.4187396	2.17132477	0.02990663
Seed dry mass (n = 107)	0.59095833	0.6249377	0.94562753	0.34433860
Seedbank duration (n = 47)	-0.57477315	0.2810764	-2.04489973	0.04086474
Root C content (n = 8)	0.51208604	0.7721347	0.66320814	0.50719726
Leaf C/N ratio (n = 60)	-0.50129726	0.4714395	-1.06333319	0.28763088
Leaf N content (n = 80)	0.46837365	0.2106247	2.22373615	0.02616620
Seed protein content (n = 15)	-0.42590900	0.5528819	-0.77034361	0.44109610
Leaf P content (n = 45)	0.41022583	0.3123900	1.31318504	0.18912060
Shoot C/N ratio (n = 50)	-0.39617413	0.2799607	-1.41510632	0.15703733
Seed number per reproduction unit (n = 45)	-0.34079857	0.4080432	-0.83520207	0.40360394
Leaf N/P ratio (n = 45)	-0.20422994	0.2928896	-0.69729321	0.48561932
Seed number per square meter (n = 23)	-0.13404030	0.4367237	-0.30692241	0.75890243
Seedbank number of layers (n = 49)	-0.13169849	0.2746944	-0.47943631	0.63162827
Root N content (n = 15)	-0.12091615	0.4336553	-0.27883007	0.78037523
Root P content (n = 9)	0.10740219	0.5794593	0.18534896	0.85295534
Seedbank thickness of top layer (n = 41)	-0.06735398	0.3031885	-0.22215213	0.82419546
Seed number per ramet (n = 22)	-0.05149045	0.4272983	-0.12050235	0.90408522
Leaf C content (n = 60)	-0.04635392	0.2435059	-0.19036060	0.84902657
Seedbank density (n = 39)	0.01418125	0.2534307	0.05595713	0.95537596

Appendix K

Table K1-1. Clustering of wetland plant taxa based on expert-assigned forage value scores, confidence levels, and thematic analysis of expert comments.

№	Group	Scientific Name	Common Name	Median Score	Median Confidence	№ of comments	№ of experts	Comments
1	Plant taxa with median confidence from 1.0 to 2.0 and median score 1.0	*Apios americana*	Common Groundnut	1.0	1.0	25	4	Experts: often noted that they were unfamiliar with the plant taxa or did not associate the taxon as a wetland plant or as food for waterfowl; were not sure about the energy value for waterfowl; were not sure about accessibility as a food source; noted the low productivity of seeds or their low energy value; pointed out that some species may have some nutritional value (and are not a priority for consumption) for only a few species of waterfowl: *Potentilla* spp. can be consumed by geese; *Apios americana* and *Amphicarpaea bracteata* may have some nutritional value due to the presence of tubers, although in general the species were not associated with waterfowl food.
		Cuscuta spp. (annual)	Dodder	1.0	1.0			
		Riccia fluitans	Floating Crystalwort	1.0	1.0			
		Amphicarpaea bracteata	American Hog Peanut	1.0	1.5			
		Teucrium canadense	Canada Germander	1.0	1.5			
		Barbarea spp.	Winter Cress	1.0	2.0			
		Boehmeria cylindrica	False Nettle	1.0	2.0			
		Erechtites spp.	Fireweeds	1.0	2.0			
		Eutrochium maculatum	Joe-Pye Weed	1.0	2.0			
		Lysimachia spp.	Loosestrife	1.0	2.0			
		Myosotis spp.	Forget-me-not	1.0	2.0			
		Pilea spp.	Clearweed	1.0	2.0			
		Potentilla spp.	Cinquefoils	1.0	2.0			
		Stachys spp.	Hedge-nettle	1.0	2.0			
		Triadenum spp.	Marsh St. Johnswort	1.0	2.0			
2	Plant taxa with median confidence from 1.5 to	*Justicia americana*	American Water-willow	1.5	2.0	21	6	Experts: sometimes noted that they were unfamiliar with the plant taxa or did not associate the taxon as a wetland plant or as food for waterfowl;
		Melilotus spp. (annual)	Melilot	1.5	2.0			

№	Group	Scientific Name	Common Name	Median Score	Median Confidence	№ of comments	№ of experts	Comments
	2.0 and median score from 1.5 to 2.0	*Ricciocarpos natans*	Fringed Heartwort	1.5	2.0			were not sure about the energy value for waterfowl; were not sure about accessibility as a food source (including through place of germination); positive nutritional value of certain parts of plant species (in particular leaves) was noted; for some species it was noted that the seeds were too hard to digest effectively; pointed out that some species may have some nutritional value (but not always a priority) for only a few species of waterfowl: *Acer saccharinum* can be valuable for Wood ducks (*Aix sponsa*).
		Acer saccharinum	Silver Maple	1.5	3.0			
		Strophostyles spp. (annual)	Wild Bean	2.0	1.5			
		Heteranthera dubia	Water Star-grass	2.0	2.0			
		Lathyrus spp. (perennial)	Wild Pea	2.0	2.0			
		Ribes spp.	Gooseberry	2.0	2.0			
		Rorippa spp. (annual)	Yellow-Cress	2.0	2.0			
		Strophostyles spp. (perennial)	Wild Bean	2.0	2.0			
3	Plant taxa with median confidence from 2.5 to 4.0 and median score 1.0	*Iris* spp.	Iris/Flag	1.0	2.5	13	5	Experts: were confident about the poor energetic value, and also noted additional negative reasons that affect waterfowl nutrition for some plant taxa due to their large coverage area; were confident that plants or seeds are not accessible to waterfowl (including through seed propagation techniques); pointed out that some species may have some nutritional value (but not always a priority) for only a few species of waterfowl: *Typha* spp. can be valuable for swans; *Iris* spp. can be consumed by geese; *Hydrocharis morsus-ranae*, despite the agreement among experts on its low nutritional quality, can be consumed by swans and some duck species. Although experts noted that they
		Apocynum spp.	Dogbane	1.0	3.0			
		Cirsium spp.	Thistles	1.0	3.0			
		Daucus carota	Queen Anne's Lace	1.0	3.0			
		Hibiscus spp.	Rose Mallow	1.0	3.0			
		Hydrocharis morsus-ranae	European Frog-bit	1.0	3.0			
		Parthenocissus quinquefolia	Virginia Creeper	1.0	3.0			
		Phalaris arundinacea	Reed Canarygrass	1.0	3.0			
		Salix spp.	Willow	1.0	3.0			
		Thelypteris palustris	Eastern Marsh Fern	1.0	3.0			
		Typha spp.	Cattail	1.0	3.0			
		Phragmites spp.	Common Reed	1.0	4.0			

№	Group	Scientific Name	Common Name	Median Score	Median Confidence	№ of comments	№ of experts	Comments
								were not sure about the protein content or total energy value indicated that their assessments were based either on data from the literature or on their own observations.
4	Plant taxa with median confidence from 2.5 to 3.0 and median score 2.0	*Nitellopsis obtusa*	Starry Stonewort	2.0	2.5	19	9	Experts: were confident about some energy value, noting that some taxa are not well studied or their nutritional value is not very significant compared to other plants; noted some potential nutritional value, but lowered the score due to the limited availability of taxa as food; noted low seed productivity and small seed sizes; indicated some nutritional value of leaves or young shoots for some taxa; pointed out that some species have nutritional value for some groups of waterfowl: *Spartina* spp. was noted by many experts as a valuable taxon for grazing species; *Chara* spp. was noted as being consumed by many waterfowl species, not for its own energy value, but because it is an excellent home for invertebrates and is consumed with them. indicated that their assessments were based on their own observations, field research, literature and data from USDA plant database.
		Butomus umbellatus	Flowering Rush	2.0	3.0			
		Calamagrostis spp.	Reedgrass	2.0	3.0			
		Chara spp.	Muskgrass or Stonewort	2.0	3.0			
		Decodon verticillatus	Swamp Loosestrife	2.0	3.0			
		Glyceria spp.	Mannagrass	2.0	3.0			
		Nuphar spp. (perennial)	Yellow Water Lily	2.0	3.0			
		Nymphaea spp.	Water Lily	2.0	3.0			
		Spartina spp.	Cordgrass	2.0	3.0			
		Spirodela polyrhiza	Greater Duckweed	2.0	3.0			
		Utricularia spp.	Bladderwort	2.0	3.0			
		Wolffia spp.	Watermeal	2.0	3.0			
5	Plant taxa with median confidence	*Carex* spp.	Sedges	3.0	3.0	38	9	Experts: were confident of the significant energy value;
		Eleocharis spp.	Spike Rushes	3.0	3.0			
		Juncus spp.	Rushes	3.0	3.0			

№	Group	Scientific Name	Common Name	Median Score	Median Confidence	№ of comments	№ of experts	Comments
	from 3.0 to 4.0 and median score from 3.0 to 3.5	*Lemna* spp.	Duckweed	3.0	3.0			made conclusions based on TME values, amount of seeds produced by the taxon, the complexity of seed digestion, total cover/distribution (the amount of food produced by the taxon), carbohydrate content; separately pointed out the energy value for certain parts of plants, in particular seeds, shoots, leaves, rootstock, tubers; most plant taxa (*Eleocharis* spp., *Lemna* spp., *Setaria* spp., *Bidens* spp., *Ceratophyllum demersum*, *Leersia oryzoides*) were noted as being widely consumed by many different waterfowl species, additionally: *Najas* spp. is noted by experts as a species widely consumed by waterfowl based on observations, although there is limited information about it in the literature; *Ceratophyllum demersum* is noted as valuable both in terms of seeds and vegetative parts, and due to its excellent habitat for invertebrates; for species of *Cyperus* spp., the value depends on the presence of tubers. However, the challenge is that it is only possible to distinguish between species with and without tubers during a certain period of time/season. These species can also be easily confused, which can lead to an error in assessing the food quality of the environment. noted the difference between annual and perennial species, noting that annuals are
		Najas spp.	Water-nymphs	3.0	3.0			
		Persicaria (Polygonum) spp.	Annual Smartweeds	3.0	3.0			
		Pontederia cordata	Pickerelweed	3.0	3.0			
		Potamogeton spp.	Pondweed	3.0	3.0			
		Schoenoplectus spp.	Bulrush	3.0	3.0			
		Scirpus spp.	Bulrush	3.0	3.0			
		Setaria spp.	Foxtail	3.0	3.0			
		Sparganium spp.	Bur Reed	3.0	3.0			
		Bidens spp.	Beggartick	3.0	3.5			
		Ceratophyllum demersum	Coontail	3.0	4.0			
		Cyperus spp. (annual)	Flat Sage Annual	3.0	4.0			
		Persicaria (Polygonum) spp.	Perennial Smartweeds	3.0	4.0			
		Leersia oryzoides	Rice Cutgrass	3.5	3.5			
		Stuckenia pectinata	Sago Pondweed	3.5	4.0			

№	Group	Scientific Name	Common Name	Median Score	Median Confidence	№ of comments	№ of experts	Comments
								preferable for birds due to higher seed production (some experts). indicated that their assessments were based on their own observations, field research (including waterfowl stomach contents) and literature data.
6	Plant taxa with median confidence 4.0 and median score 4.0	*Cyperus* spp. (perennial)	Perennial Sedge	4.0	4.0	20	7	Experts: noted high energy values, often referring to TME values; pointed out the importance of germination site as a limiting factor in availability; put a lot of emphasis on tubers, number of seeds per seed head, number of seeds produced per plant, number of seeds produced per hectare, and carbonate content
		Echinochloa spp.	Wild Millets	4.0	4.0			
		Panicum spp.	Panic Grass	4.0	4.0			
		Quercus spp.	Oak	4.0	4.0			
		Sagittaria spp.	Arrowhead / Duck Potato	4.0	4.0			
		Vallisneria americana	Wild Celery	4.0	4.0			
		Zizania spp.	Wild Rice	4.0	4.0			

Appendix L

As part of a follow-up analysis, I investigated whether forage value scores and expert confidence levels differed between introduced and native plant species. To address this, I used the VasCan database (Brouillet et al., 2016), which compiles expert-verified information on the nativity status of vascular plant taxa for each Canadian province. The VasCan taxonomy is maintained by the Canadensys network and built from authoritative regional floras, herbarium records, and botanical literature, providing a reliable source for distinguishing native, introduced, and invasive taxa (Brouillet et al., 2016).

In designing the expert survey, species were grouped at the genus level when closely-related taxa shared similar key traits relevant to forage quality. This approach was chosen because, for several taxonomic groups, plant species within a genus often exhibit comparable functional characteristics such as seed production, presence and abundance of tubers, and life cycle type (e.g., annual vs. perennial), as determined using the USDA Plants Database (USDA & NRCS, 2024) and *A Great Lakes Wetland Flora, Fourth Edition* (Chadde, 2019) (see Appendix F for full details). Moreover, expert feedback also supported this approach, as several experts noted that even when some species within a genus may differ in forage quality (e.g., due to the presence of tubers), distinguishing between them can be difficult in the field, as in the case of *Cyperus* spp. (Appendix K). As a result, experts were asked to evaluate forage quality and assign confidence ratings primarily at the genus level rather than for individual species.

However, this structure presents a substantial challenge for assessing non-nativity. Upon reviewing the genera included in the survey using VasCan, I found that many genera classified as native to Ontario, and broadly represented by native species, also include several introduced species (Table L1-1). In such cases, a single forage value coefficient was applied uniformly across all species within a genus, regardless of whether some members of that genus are considered introduced.

Table L1-1. Status of plant genera included in the expert survey, with representative introduced species listed according to the VasCan database for Ontario

Genus	Species		Status in VasCan
	Scientific Name	**Common Name**	
Barbarea spp.*		Winter Cress	Native
	Barbarea stricta	Small-flowered Winter Cress	Introduced
	Barbarea vulgaris	Bitter Winter Cress	Introduced
Bidens spp.*		Beggartick	Native
	Bidens aristosa	Bearded Beggarticks	Introduced
	Bidens pilosa	Hairy Beggarticks	Introduced
Butomus spp.		Flowering Rush	Introduced
	*Butomus umbellatus**	Flowering Rush	Introduced
Calamagrostis spp.*		Reedgrass	Native
	Calamagrostis epigejos	Chee Reedgrass	Introduced
Carex spp.*		Sedges	Native
	Carex acutiformis	Lesser Pond Sedge	Introduced
	Carex disticha	Two-ranked Sedge	Introduced
	Carex divulsa	Grassland Sedge	Introduced
	Carex flacca	Heath Sedge	Introduced
	Carex hirta	Hammer Sedge	Introduced
	Carex hookeriana	Hooker's Sedge	Introduced
	Carex leersii	Leers' Sedge	Introduced
	Carex muricata	Lesser Prickly Sedge	Introduced
	Carex nigra	Smooth Black Sedge	Introduced
	Carex praegracilis	Clustered Field Sedge	Introduced
	Carex spicata	Spiked Sedge	Introduced
	Carex sylvatica	European Woodland Sedge	Introduced
Cirsium spp.*		Thistles	Native
	Cirsium arvense	Canada Thistle	Introduced
	Cirsium palustre	Marsh Thistle	Introduced
	Cirsium vulgare	Bull Thistle	Introduced
Cyperus spp.*		Perennial Sedge	Native
	Cyperus fuscus	Brown Flatsedge	Introduced
Daucus spp.		Wild Carrot	Introduced
	*Daucus carota**	Queen Anne's Lace	Introduced
Echinochloa spp.*		Wild Millets	Native
	Echinochloa Crus-galli	Large Barnyard Grass	Introduced
Glyceria spp.*		Mannagrass	Native

	Glyceria maxima	Rough Mannagrass	Introduced
Hibiscus spp.*		Rose Mallow	Native
	Hibiscus trionum	Flower-of-an-hour	Introduced
Hydrocharis spp.		Frog-bit	Introduced
	*Hydrocharis morsus-ranae**	European Frog-bit	Introduced
Iris spp.*		Iris or Flag	Native
	Iris × germanica	German Iris	Introduced
	Iris ensata	Japanese Iris	Introduced
	Iris pallida	Sweet Iris	Introduced
	Iris prismatica	Slender Blue Iris	Introduced
	Iris pseudacorus	Yellow Iris	Introduced
	Iris pumila	Dwarf Iris	Introduced
	Iris sibirica	Siberian Iris	Introduced
Juncus spp.*		Rushes	Native
	Juncus compressus	Compressed Rush	Introduced
	Juncus conglomeratus	Compact Rush	Introduced
	Juncus inflexus	Incurved Rush	Introduced
Lathyrus spp.*		Wild Pea	Native
	Lathyrus latifolius	Everlasting Pea	Introduced
	Lathyrus pratensis	Meadow Vetchling	Introduced
	Lathyrus sativus	Cultivated Vetchling	Introduced
	Lathyrus sylvestris	Narrow-leaved Vetchling	Introduced
	Lathyrus tuberosus	Tuberous Vetchling	Introduced
Lysimachia spp.*		Loosestrife	Native
	Lysimachia arvensis	Scarlet Pimpernel	Introduced
	Lysimachia nummularia	Creeping Yellow Loosestrife	Introduced
	Lysimachia punctata	Spotted Yellow Loosestrife	Introduced
	Lysimachia vulgaris	Garden Yellow Loosestrife	Introduced
Melilotus spp.*		Melilot	Introduced
	Melilotus albus	White Sweet-clover	Introduced
	Melilotus altissimus	Tall Yellow Sweet-clover	Introduced
	Melilotus officinalis	Yellow Sweet-clover	Introduced
Myosotis spp.*		Forget-me-not	Native
	Myosotis arvensis	Field Forget-me-not	Introduced
	Myosotis discolor	Yellow-and-blue Forget-me-not	Introduced
	Myosotis scorpioides	True Forget-me-not	Introduced
	Myosotis stricta	Upright Forget-me-not	Introduced
	Myosotis sylvatica	Woodland Forget-me-not	Introduced
Najas spp.*		Water-nymphs	Native
	Najas marina	Spiny Naiad	Introduced
	Najas minor	Brittle-leaved Naiad	Introduced
Panicum spp.*		Panic Grass	Native
	Panicum dichotomiflorum	Fall Panicgrass	Introduced
	Panicum miliaceum	Proso Millet	Introduced

Persicaria (*Polygonum*) spp.*		Smartweeds	Native
	Persicaria hydropiper	Marsh-pepper Smartweed	Introduced
	Persicaria longiseta	Long-bristled Smartweed	Introduced
	Persicaria maculosa	Spotted Lady's-thumb	Introduced
	Persicaria minor	Small Water-pepper	Introduced
	Persicaria orientalis	Oriental Smartweed	Introduced
Phragmites spp.*		Reed	Native
	Phragmites australis	Common Reed	Introduced
Potamogeton spp. *		Pondweed	Native
	Potamogeton crispus	Curly-leaved Pondweed	Introduced
Potentilla spp.*		Cinquefoils	Native
	Potentilla argentea	Silvery Cinquefoil	Introduced
	Potentilla inclinata	Ashy Cinquefoil	Introduced
	Potentilla indica	Mock Strawberry	Introduced
	Potentilla intermedia	Downy Cinquefoil	Introduced
	Potentilla recta	Sulphur Cinquefoil	Introduced
	Potentilla reptans	Creeping Cinquefoil	Introduced
	Potentilla verna	Spring Cinquefoil	Introduced
Quercus spp.*		Oak	Native
	Quercus robur	English Oak	Introduced
Ribes spp.*		Gooseberry	Native
	Ribes alpinum	Alpine Currant	Introduced
	Ribes aureum	Golden Currant	Introduced
	Ribes missouriense	Missouri Gooseberry	Introduced
	Ribes nigrum	European Black Currant	Introduced
	Ribes rubrum	European Red Currant	Introduced
	Ribes uva-crispa	European Gooseberry	Introduced
Rorippa spp.*		Yellow-Cress	Native
	Rorippa curvipes	Blunt-leaved Yellowcress	Introduced
	Rorippa sinuata	Spreading Yellowcress	Introduced
	Rorippa sylvestris	Creeping Yellowcress	Introduced
Salix spp.*		Willow	Native
	Salix × pendulina	Pendulous Willow	Introduced
	Salix × rubens	Hybrid White Willow	Introduced
	Salix × sepulcralis	Golden Weeping Willow	Introduced
	Salix alba	White Willow	Introduced
	Salix atrocinerea	Rusty Willow	Introduced
	Salix caprea	Goat Willow	Introduced
	Salix cinerea	Ashy Willow	Introduced
	Salix daphnoides	Violet Willow	Introduced
	Salix elaeagnos	Elaeagnus Willow	Introduced
	Salix fragilis	Crack Willow	Introduced
	Salix matsudana	Corkscrew Willow	Introduced
	Salix myrsinifolia	Dark-leaved Willow	Introduced
	Salix pentandra	Laurel Willow	Introduced

	Salix purpurea	Purple Willow	Introduced
	Salix triandra	Almond Willow	Introduced
	Salix viminalis	Basket Willow	Introduced
Setaria spp.*		Foxtail	Introduced
	Setaria faberi	Giant Foxtail	Introduced
	Setaria pumila	Yellow Foxtail	Introduced
	Setaria verticillata	Bristly Foxtail	Introduced
	Setaria viridis	Green Foxtail	Introduced
Spartina spp. (*Sporobolus*)*		Cordgrass	Native
	Sporobolus pumilus	Saltmeadow Cordgrass	Introduced
	Sporobolus schoenoides	Swamp Pricklegrass	Introduced
Stachys spp.*			Native
	Stachys byzantina	Woolly Hedge-nettle	Introduced
	Stachys germanica	German Hedge-nettle	Introduced
	Stachys palustris	Marsh Hedge-nettle	Introduced
	Stachys sylvatica	Woodland Hedge-nettle	Introduced
Triadenum spp. (*Hypericum*)*		Marsh St. Johnswort	Native
	Hypericum hirsutum	Hairy St. John's-wort	Introduced
	Hypericum maculatum	Dotted St. John's-wort	Introduced
	Hypericum perforatum	Perforate St. John's-wort	Introduced
	Hypericum tetrapterum	Four-winged St. John's-wort	Introduced
Typha spp.*		Cattail	Native
	Typha angustifolia	Narrow-leaved Cattail	Introduced

* - indicates a species or genus that was included in the expert elicitation

Thus, while the idea of comparing expert assessments of native versus introduced or invasive species is conceptually sound, the genus-level resolution of the survey design precludes such an analysis. Any attempt to apply a binary native/introduced or invasive classification across genera would oversimplify the taxonomic and ecological complexity of the surveyed taxa and could lead to misleading conclusions. Although a small number of taxa in the survey, such as Flowering Rush (*Butomus umbellatus*) and European Frog-bit (*Hydrocharis morsus-ranae*), are well-documented invasive species with no native congeners in Ontario, the total number of such clearly invasive entries is too small for meaningful statistical comparison. Additionally, an open question remains as to whether invasive species from otherwise native genera negatively affect the diet quality of waterfowl. For example, Wild Millets (*Echinochloa* spp.) received the highest forage quality rating and confidence from experts and are known to be highly valuable as food for waterfowl. The genus *Echinochloa* has a native status in VasCan, yet it includes Large Barnyard Grass (*Echinochloa crus-galli*), which is listed as introduced. It is likely that ducks consume this species just as readily as others in the genus, and its invasiveness may not substantially affect its value as food.

In contrast, Cattail (*Typha* spp.) and Common Reed (*Phragmites* spp.) are genera considered native in VasCan, yet Narrow-leaved Cattail (*Typha angustifolia*) and Common Reed (*Phragmites australis*) or their hybrids are among the most widespread and dominant invasive wetland plants in the region (Wilcox et al., 2003; Whyte et al., 2015; Lishawa et al., 2020). Even native species within these genera are of limited dietary value to waterfowl, and invasive members are unlikely to offer superior forage benefits. In such cases, if ducks avoid native *Typha* species, they are unlikely to benefit from invasive *Typha* either. Conversely, European Frog-bit (*Hydrocharis morsus-ranae*) is an example of a plant that is invasive and ecologically disruptive (Zhu et al., 2018), even if it may be occasionally consumed by certain species of waterfowl. Its rapid spread displaces native species and alters ecosystem structure, demonstrating that not all invasive plants, regardless of edibility, should be regarded neutrally in habitat assessments.

Taken together, this analysis suggests that comparing “invasiveness” or native status in isolation from site context and successional stage is not appropriate. Only by assessing changes in biodiversity over time at specific locations, such as through long-term habitat monitoring or management interventions, can we begin to evaluate whether plant community shifts (including introductions and invasions) represent improvements or declines in ecosystem function from the perspective of waterfowl habitat quality.

Appendix M

In Table M1-1, I indicate how filtering out coefficient scores with a confidence value below different thresholds would influence both the percentage of experts whose opinion would inform the species or genus coefficient score and how the resulting median score would change. First, I observed that filtering out low-confidence scores significantly reduced the available data. When excluding all scores with a confidence level of 1, 6 out of 74 plant taxa lost 50% or more of their available scores. When excluding scores with confidence levels of 1 and 2, 26 taxa lost 50% or more of their scores. Finally, when excluding all scores with confidence levels of 1, 2, and 3 except those with the highest confidence level (confidence = 4), 63 out of 74 taxa lost 50% or more of their scores. The outcome of filtering the data to remove coefficients without high confidence would consequently be that some species would be excluded from assessments with the vFQI tool.

Table M1-1. Effect of confidence threshold filtering on expert-derived forage value coefficients

Scientific Name	Total number of scores	% Scores remaining (Confidence ≥ 2)	% Scores remaining (Confidence ≥ 3)	% Scores remaining (Confidence = 4)	Median score (All responses)	Median score (Confidence ≥ 2)	Median score (Confidence ≥ 3)	Median score (Confidence = 4)
Acer saccharinum	22	90.9	77.3	18.2	1.5	1.5	1	1
Amphicarpaea bracteata	22	50	18.2	0	1	1	1	0
Apios americana	22	45.5	9.1	0	1	1	1	0
Apocynum spp.	22	86.4	54.5	13.6	1	1	1	1
Barbarea spp.	22	77.3	36.4	4.5	1	1	1	1
Bidens spp.	22	95.5	90.9	50	3	3	3	3

Scientific Name	Total number of scores	% Scores remaining (Confidence ≥ 2)	% Scores remaining (Confidence ≥ 3)	% Scores remaining (Confidence = 4)	Median score (All responses)	Median score (Confidence ≥ 2)	Median score (Confidence ≥ 3)	Median score (Confidence = 4)
Boehmeria cylindrica	22	81.8	40.9	0	1	1	1	0
Butomus umbellatus	22	81.8	63.6	9.1	2	2	2	1
Calamagrostis spp.	22	90.9	72.7	4.5	2	2	1	1
Carex spp.	22	100	95.5	40.9	3	3	3	3
Ceratophyllum demersum	22	100	100	59.1	3	3	3	3
Chara spp.	22	95.5	86.4	36.4	2	2	2	2
Cirsium spp.	22	81.8	68.2	13.6	1	1	1	1
Cuscuta spp. (annual)	22	45.5	31.8	4.5	1	1	1	1
Cyperus spp. (annual)	22	95.5	95.5	54.5	3	3	3	3.5
Cyperus spp. (perennial)	22	100	95.5	63.6	4	4	4	4
Daucus carota	22	95.5	63.6	13.6	1	1	1	1
Decodon verticillatus	22	95.5	68.2	9.1	2	2	2	1
Echinochloa spp.	22	100	95.5	81.8	4	4	4	4
Eleocharis spp.	22	100	100	36.4	3	3	3	2
Erechtites spp.	22	59.1	27.3	0	1	1	1	0
Eutrochium maculatum	22	77.3	40.9	13.6	1	1	1	1
Glyceria spp.	22	95.5	54.5	4.5	2	2	2	2
Heteranthera dubia	22	77.3	36.4	4.5	2	2	2	1
Hibiscus spp.	22	100	54.5	13.6	1	1	1	1

Scientific Name	Total number of scores	% Scores remaining (Confidence ≥ 2)	% Scores remaining (Confidence ≥ 3)	% Scores remaining (Confidence = 4)	Median score (All responses)	Median score (Confidence ≥ 2)	Median score (Confidence ≥ 3)	Median score (Confidence = 4)
Hydrocharis morsus-ranae	22	86.4	63.6	18.2	1	2	2.5	1.5
Iris spp.	22	95.5	50	9.1	1	1	1	1
Juncus spp.	22	100	95.5	36.4	3	3	3	3
Justicia americana	22	59.1	18.2	4.5	1.5	1	1	1
Lathyrus spp. (perennial)	22	54.5	31.8	0	2	2	2	0
Leersia oryzoides	22	95.5	90.9	50	3.5	4	4	4
Lemna spp.	22	100	100	45.5	3	3	3	3
Lysimachia spp.	22	72.7	36.4	9.1	1	1	1	1
Melilotus spp. (annual)	22	72.7	36.4	4.5	1.5	1.5	1	1
Myosotis spp.	22	59.1	40.9	13.6	1	1	1	1
Najas spp.	22	100	72.7	13.6	3	3	3	3
Nitellopsis obtusa	22	81.8	50	4.5	2	2	2	2
Nuphar spp. (perennial)	22	100	86.4	31.8	2	2	2	3
Nymphaea spp.	22	90.9	72.7	18.2	2	2	2.5	3.5
Panicum spp.	22	100	90.9	54.5	4	4	4	4
Parthenocissus quinquefolia	22	81.8	54.5	9.1	1	1	1	1.5
Persicaria (Polygonum) spp. (annual)	22	86.4	86.4	45.5	3	3	3	3
Persicaria (Polygonum) spp. (perennial)	22	100	95.5	54.5	3	3	3	3
Phalaris arundinacea	22	86.4	77.3	27.3	1	1	1	1
Phragmites spp.	22	100	95.5	68.2	1	1	1	1

Scientific Name	Total number of scores	% Scores remaining (Confidence ≥ 2)	% Scores remaining (Confidence ≥ 3)	% Scores remaining (Confidence = 4)	Median score (All responses)	Median score (Confidence ≥ 2)	Median score (Confidence ≥ 3)	Median score (Confidence = 4)
Pilea spp.	22	59.1	31.8	4.5	1	1	1	1
Pontederia cordata	22	100	77.3	18.2	3	3	3	3
Potamogeton spp.	22	95.5	86.4	40.9	3	3	3	3
Potentilla spp.	22	63.6	40.9	13.6	1	1	1	1
Quercus spp.	22	100	100	54.5	4	4	4	4
Ribes spp.	22	77.3	27.3	4.5	2	2	2	2
Riccia fluitans	22	45.5	36.4	4.5	1	2	2	2
Ricciocarpus natans	22	59.1	27.3	0	1.5	2	2	0
Rorippa spp. (annual)	22	77.3	13.6	4.5	2	2	2	2
Sagittaria spp.	22	100	100	59.1	4	4	4	4
Salix spp.	22	90.9	86.4	40.9	1	1	1	1
Schoenoplectus spp.	22	100	100	45.5	3	3	3	3
Scirpus spp.	22	100	90.9	31.8	3	3	3	3
Setaria spp.	22	100	86.4	31.8	3	3	3	4
Sparganium spp.	22	100	81.8	27.3	3	3	3	3.5
Spartina spp.	22	81.8	63.6	9.1	2	2	2	2
Spirodela polyrhiza	22	100	81.8	22.7	2	2	2	3
Stachys spp.	22	63.6	22.7	4.5	1	1	1	1
Strophostyles spp. (annual)	22	50	13.6	9.1	2	2	1	1.5
Strophostyles spp. (perennial)	22	59.1	22.7	4.5	2	2	1	1
Stuckenia pectinata	22	100	90.9	59.1	3.5	3.5	4	4

Scientific Name	Total number of scores	% Scores remaining (Confidence ≥ 2)	% Scores remaining (Confidence ≥ 3)	% Scores remaining (Confidence = 4)	Median score (All responses)	Median score (Confidence ≥ 2)	Median score (Confidence ≥ 3)	Median score (Confidence = 4)
Teucrium canadense	22	50	13.6	4.5	1	2	1	1
Thelypteris palustris	22	81.8	63.6	13.6	1	1	1	1
Triadenum spp.	22	59.1	36.4	9.1	1	1	1	1
Typha spp.	22	95.5	95.5	45.5	1	1	1	1
Utricularia spp.	22	90.9	63.6	4.5	2	2	2	1
Vallisneria americana	22	95.5	90.9	68.2	4	4	4	4
Wolffia spp.	22	90.9	72.7	22.7	2	2	2	2
Zizania spp.	22	100	100	77.3	4	4	4	4

However, this filtering exercise also provided additional evidence for the robustness of using the median as the final forage value coefficient. Despite substantial reductions in sample size for many taxa, the median forage value generally remained stable or changed minimally, even when a significant proportion of low-confidence scores was removed. This suggests that the median is not only appropriate for handling ordinal data but also resilient to potential outliers or uncertain expert evaluations, thereby reducing the risk of biased estimates caused by a few highly uncertain ratings.

Although the median forage value usually stayed the same even when many scores were removed, some plant taxa showed bigger changes when low-confidence scores were excluded. The plants with median coefficients more sensitive to exclusion of low-confidence scores included *Amphicarpaea bracteata*, *Apios americana*, *Boehmeria cylindrica*, *Butomus umbellatus*, *Calamagrostis* spp., *Decodon verticillatus*, *Eleocharis* spp., *Erechtites* spp., *Heteranthera dubia*, *Hydrocharis morsus-ranae*, *Lathyrus* spp. (perennial), *Nuphar* spp. (perennial), *Nymphaea* spp., *Riccia fluitans*, *Ricciocarpos natans*, *Setaria* spp., *Spirodela polyrhiza*, *Strophostyles* spp. (annual and perennial), *Teucrium canadense*, and *Utricularia* spp. These patterns likely reflect either limited available knowledge about these plant taxa or high variability in interpretation among experts. Overall, I do not recommend removing low-confidence ratings from the final analysis, particularly in small sample studies like mine, where preserving as much information as possible is critical to avoid bias and data loss.

Appendix N

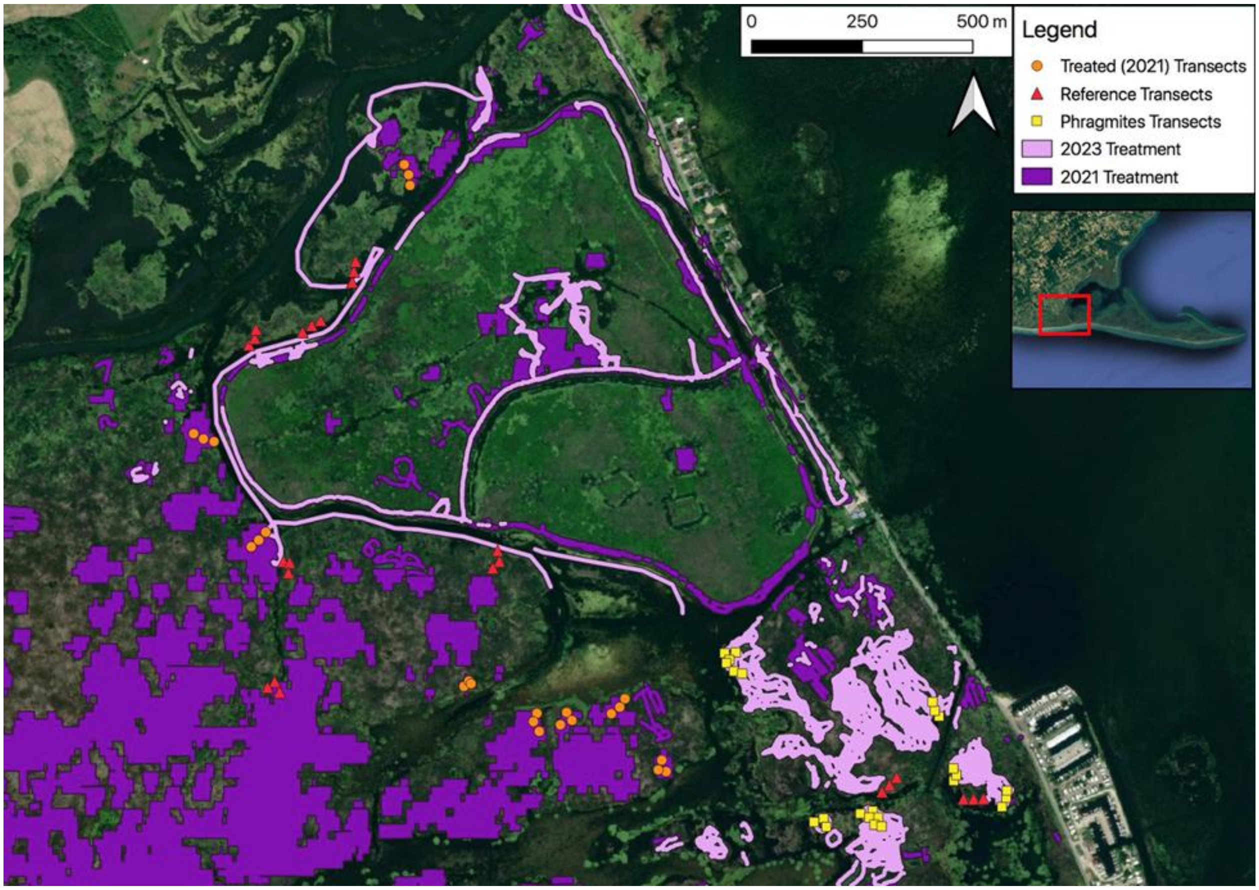


Figure N1-1. Locations of waterfowl foraging transects in Big Creek NWA, established in August 2022 in relation to herbicide treatment that took place in 2021 and October 2023. Points mark the start, middle and end of each 50 m transect. Symbology reflects the transect type (herbicide-treated, uninvaded reference, or *P. australis*-invaded control). Figure was created in qGIS3.8 (2023).

Figure N2-1. Locations of waterfowl foraging transects in Big Creek NWA, established in August 2022, in relation to herbicide treatment that took place in 2021. Points mark the start, middle and end of each 50 m transect. Symbology reflects the transect type (herbicide-treated, uninvaded reference). Figure was created in qGIS3.8 (2023).

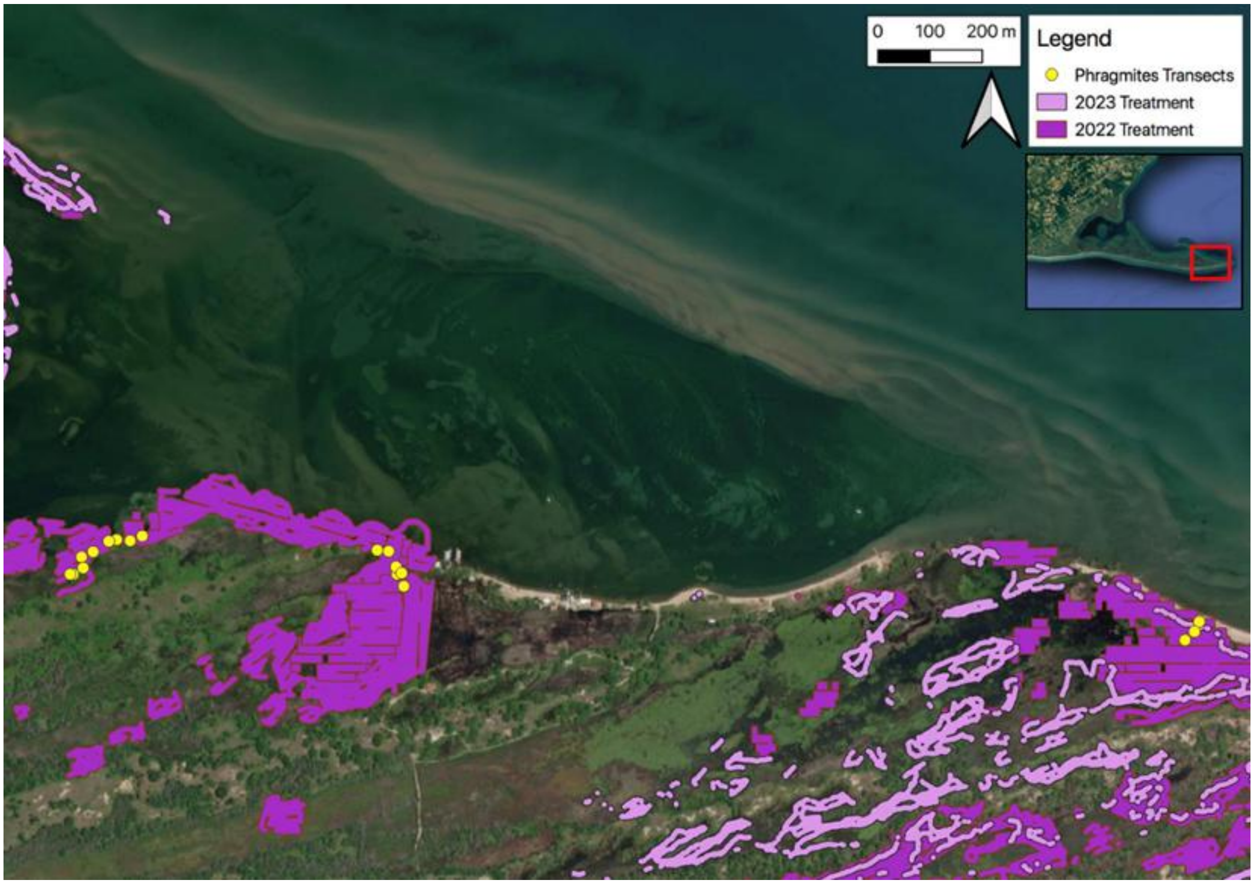


Figure N3-1. Locations of *P. australis*-invaded waterfowl foraging transects in the Long Point NWA, established in August 2022, in relation to herbicide treatment that took place in September 2022 and October 2023. Points mark the start, middle and end of each 50 m transect. Figure was created in qGIS3.8 (2023).

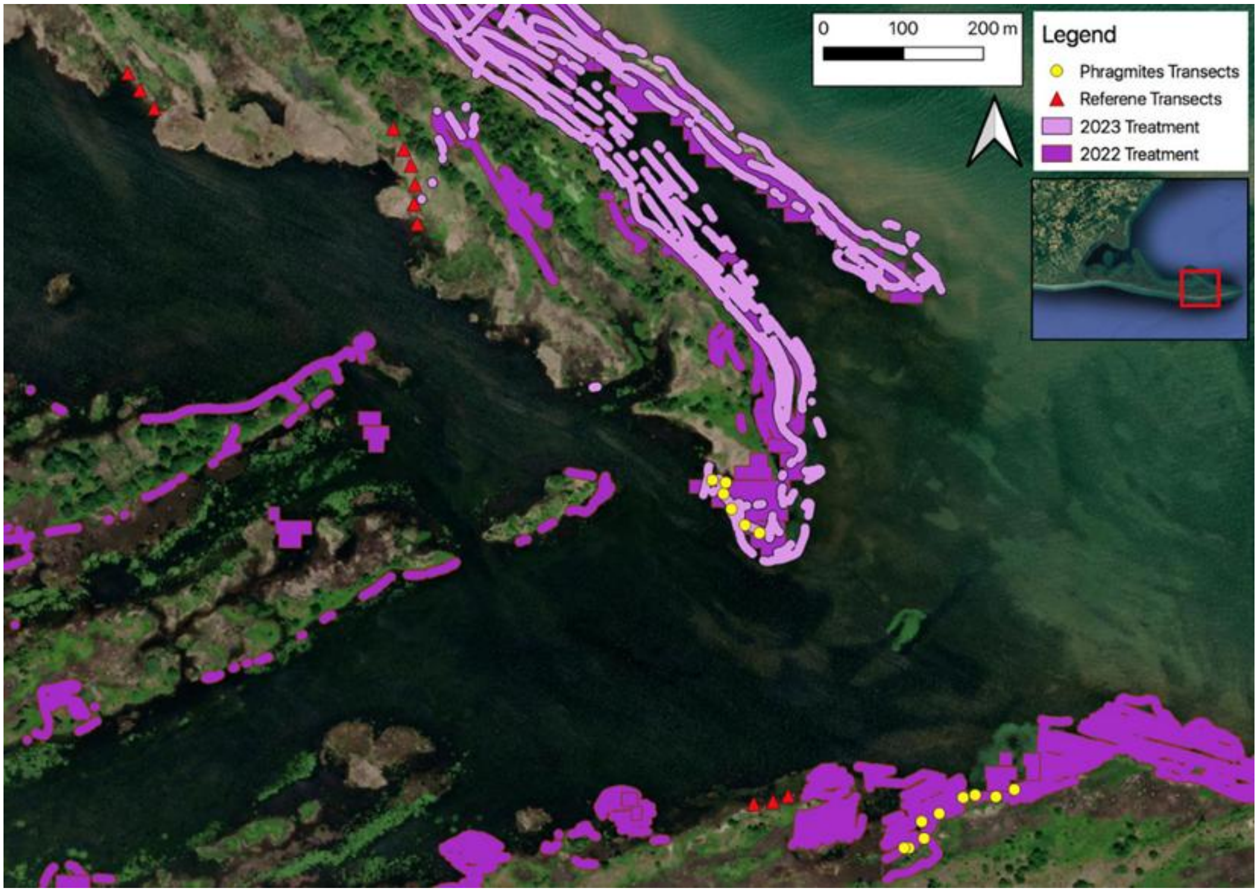


Figure N4-1. Locations of uninvaded reference and *P. australis*-invaded control waterfowl foraging transects in the Long Point NWA, established in August 2022, in relation to herbicide treatment that took place in September 2022 and October 2023. Points mark the start, middle and end of each 50 m transect. Figure was created in qGIS3.8 (2023).

Appendix O

Table O1-1. Mean water depth (cm) ± standard deviation across different vegetation types in the Big Creek and Long Point National Wildlife Areas in 2022 and 2023. "Control" refers to transects dominated by untreated *Phragmites australis*, "Treated 2021" and "Treated 2022" refer to areas sprayed one or two years prior, and "Reference" refers to native-dominated sites with no recent management interventions. In the case of Long Point, control transects were treated with herbicide after one year of observation to meet management objectives and changed their status to “Treated 2022”.

Location/ Vegetation type	Control	Treated 2022	Treated 2021	Reference
Big Creek NWA 2023	22.96 ± 11.02	-	28.05 ± 5.89	20.82 ± 6.41
Big Creek NWA 2022	17.09 ± 8.07	-	22.53 ± 6.21	18.87 ± 5.31
Long Point NWA 2023	-	9.00 ± 8.88	25.80 ± 8.73	15.21 ± 14.97
Long Point NWA 2022	5.09 ± 6.06	-	14.88 ± 6.33	6.98 ± 6.43

Table O2-1. Mean species richness (№ of taxa) ± standard deviation across different vegetation types in the Big Creek and Long Point National Wildlife Areas in 2022 and 2023. "Control" refers to transects dominated by untreated *Phragmites australis*, "Treated 2021" and "Treated 2022" refer to areas sprayed one or two years prior, and "Reference" refers to native-dominated sites with no recent management interventions. In the case of Long Point, control transects were treated with herbicide after one year of observation to meet management objectives and changed their status to “Treated 2022”.

Location/ Vegetation type	Control	Treated 2022	Treated 2021	Reference
Big Creek NWA 2023	5.37 ± 1.77	-	8.75 ± 3.24	10.62 ± 1.99
Big Creek NWA 2022	8.50 ± 2.98	-	10.00 ± 3.81	14.37 ± 2.20

Long Point NWA 2023	-	17.62 ± 7.05	18.25 ± 3.49	18.25 ± 6.62
Long Point NWA 2022	8.62 ± 4.56	-	8.37 ± 3.50	11.75 ± 2.05

Table O3-1. Mean canopy height (cm) ± standard deviation across different vegetation types in the Big Creek and Long Point National Wildlife Areas in 2023. "Control" refers to transects dominated by untreated *Phragmites australis*, "Treated 2021" and "Treated 2022" refer to areas sprayed one or two years prior, and "Reference" refers to native-dominated sites with no recent management interventions. In the case of Long Point, control transects were treated with herbicide after one year of observation to meet management objectives and changed their status to “Treated 2022”.

Location/ Vegetation type	Control	Treated 2022	Treated 2021	Reference
Big Creek NWA 2023	329.46 ± 46.86	-	75.73 ± 76.87	96.20 ± 95.87
Long Point NWA 2023	-	50.60 ± 39.03	27.68 ± 26.16	118.09 ± 38.20

Raw vegetation survey data from Long Point and Big Creek NWAs collected in 2022 can be found at:

https://doi.org/10.6084/m9.figshare.29497715.v1

Raw vegetation survey data from Long Point and Big Creek NWAs collected in 2023 can be found at:

https://doi.org/10.6084/m9.figshare.29497727.v1

A summary of key characteristics for each transect sampled in Long Point and Big Creek NWAs in 2022 and 2023, including plant species richness, average water depth, vFQI, WMWCs, canopy height, and sediment core-related variables (total seed mass, edible fraction, and inedible fraction), can be found at:

https://doi.org/10.6084/m9.figshare.29497739.v1

Raw sediment core data from Long Point and Big Creek NWAs collected in 2023 can be found at:

https://doi.org/10.6084/m9.figshare.29497745.v1

Appendix P

Protocol: Sediment Core Sample Processing for Vegetative Forage Quality Index

Created September 2022, by Estella Crosby

Reviewed September 2022, by Megan Jordan

Reviewed September 2022, by Rebecca Rooney

Purpose

One of the key motivations behind invasive *P. australis* suppression efforts undertaken by Environment and Climate Change Canada's Canadian Wildlife Service in the Long Point Walsingham Forest Priority Place was to enhance habitat for waterfowl and wildlife. Prior research showed that long-term invasion by *P. australis* reduces habitat quality for many wetland birds including waterfowl, but it is unclear whether simply removing it with herbicides actually restores habitat quality. A key aspect of habitat quality for waterfowl is forage quality and abundance, as *P. australis* displaces plant species that produce energy rich and nutritious seeds and tubers many waterfowl rely on.

This protocol outlines the steps to separate seeds, turions, rhizomes, tubers and stolons from sediment core samples taken at Big Creek NWA and Long Point NWA to obtain triplicate measures of their mass at each monitoring transect. The mass will be separated into edible (seeds, turions, and tubers or rhizomes from non-*Phragmites* plants) and inedible (fibrous roots, stolons and *Phragmites* rhizome) components. Litter and woody debris will not be included. The mass of the edible and inedible fraction will be expressed in two ways: 1) divided by the surface area of the core or cores collected from each sample and 2) divided by the sample volume to calculate the density of food available in *Phragmites australis*-invaded marsh, marsh that was treated with a glyphosate-based herbicide in 2021, and uninvaded reference marsh. These values will be contrasted with an ANOVA and post-hoc Tukey's Multiple Comparison test, assuming the data meet the test assumptions (residuals are normally distributed and homoscedastic).

Summary

The samples will be thawed and washed through 3 sieves of gradually smaller sizes to remove sediment and muck. First, plant litter and woody debris will be removed from each sieve fraction, washing larger pieces with the squirt bottle to ensure no seeds are lost. These will be discarded. Second, the *Phragmites australis* rhizomes and stolons as well as fibrous roots will be rinsed and removed to be dried and weighed as the inedible fraction. Last, the remaining seeds and tubers will be collected, dried and weighed as the edible fraction. All weights will be recorded to the nearest 0.1 mg. Dried and weighed samples will be retained, as specified in our CWS grant agreement.

Materials:

- Soil cores
- 3 sieves: Mesh size #50 (0.03 mm); Mesh size #10 (1.65 mm); Mesh size #4 (4.75 mm)
- Squirt bottle
- Plastic tubs
- Drying oven
- Forceps
- Analytical balance
- Weigh boats
- Pencil & sharpie
- Data sheets
- Paper sandwich bags to hold dried samples
- Camera and camera "rig" to standardize lighting, height and angle among photos
- Graduated cylinder to measure volume of "composite cores"
- Plastic spoon
- Light source like a head lamp
- Magnifying glass or hand lens

Procedure

1. Leave soil core samples in the fridge overnight to thaw completely.
2. For samples that comprise a single core, proceed to step 3. For samples that were a composite of multiple cores, you will need to record the total area captured in the sample (e.g., 3 cores would mean 3*pi*R2 for surface area) and, using the graduated cylinder, measure the volume of the total sample including any water, removing it from the sample bag as well as possible but using any rinse water (i.e. water should not be added or removed).
3. Prepare 2 weigh boats for each sample: 1) edible fraction, 2) inedible fraction. Label these and associated paper bags with the following info
    a. Your Name
    b. Sample collection date
    c. Sample ID including transect number and station ID (0, 25, or 50 m).
    d. Fraction type (Edible or Inedible)
    e. Project ID: Waterfowl Forage Mass
    f. Lab name: Waterloo Wetland Lab or Rooney Lab
4. Place the 4.75 mm mesh size sieve on top of the 1.65 mm mesh size sieve, and then place these sieves on top of the 0.03 mm mesh size sieve.
5. Place one soil core into the 4.75 mm mesh size sieve. Use a squirt bottle to remove the entire contents of the Ziploc bag via rinsing into the sieve stack. If you used the graduated cylinder, also rinse it out into the sieve stack.
6. Use a gentle stream of water to coax the soil core through the stack of sieves.
7. Separate the sieves and examine their contents.
    a. Remove any litter or woody debris, rising it off into the sieve stack to ensure no seeds cling. The litter and woody debris can be discarded.
    b. Remove any fibrous roots, *Phragmites* rhizomes and stolons, rinsing and then placing these into the inedible fraction weigh boat.

c. Remove any seeds, turions, tubers, and rhizomes of other plants and place them into the edible fraction weigh boat.

8. Once the sample is entirely picked over, examine it with a magnifying glass and headlamp to ensure no seeds are missed.
9. Place the labeled weigh boats with the edible and inedible fractions into the drying oven at 70 degrees Celsius until they reach a constant weight. You can test this by removing them, allowing them to cool for 5 min, and then checking their weight. If they have lost weight compared to the last time you checked, you should continue drying. Realistically, 4-6 hr should be ample drying time to reach a constant weight at 70 deg C for these small masses of material, but don't assume!
10. Once dried to a constant weight, remove the weigh boats from the oven and allow to cool for about 5 min.
11. Before using the balance, ensure that it is level and situated on a flat, stable surface free from vibrations. You may need to use the leg adjusters to re-level the balance if it has been moved – see the bubble in the window. Turn it on and tare to ensure that it zeros out properly and yields a stable value.
12. Weigh the total weigh boat, including the sample, and record this on the datasheet to the nearest 0.1 mg.
13. Position the weigh boat with the sample under the camera rig. Using your cell phone or equivalent camera, take a photo of the sample label, then photograph the sample with adequate lighting and magnification. Then take another photo of the sample label to indicate that all photos sandwiched between the two label photos belong to that sample. Document that the photo(s) was taken on the datasheet.

14. Carefully remove the sample from the weigh boat, transferring it into the labeled paper bag. Ensure that you match the fraction type (edible vs. inedible)! Once the weigh boat is clean of sample material, weigh it empty and record this mass to the nearest 0.1 mg on the datasheet. The mass of the empty boat is then subtracted from the mass of the total weight boat including sample, to derive the mass of the sample. This step can be done in Excel rather than by hand.
15. The edible and inedible fractions, in their respective labelled paper bags, can be rolled up tightly to prevent loss of material and then placed into a third paperbag to help keep them together. These should then be stored in an open topped Rubbermaid bin that is ladled with your name, the project name, term and year (Fall 2022), and the lab name (WWL). This should be stored on top of the shelves on the eastern wall of the lab B2-257.

References

Farley, E.B., Schummer, M.L., Leopold, D.J., Coluccy, J.M. and Tozer, D.C. (2022), Influence of water level management on vegetation and bird use of restored wetlands in the Montezuma Wetlands Complex. Wildlife Biology, 2022: e01016. https://doi.org/10.1002/wlb3.01016

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