Assessment of Duckling Abundance as a Biological Indicator of Wetland Health in the Prairie Pothole Region

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Abstract

Located in the central portion of North America, the Prairie Pothole Region (PPR) is one of the most biologically productive ecosystems in the world. Since European settlement, the region has undergone extensive human development, largely from agricultural practices, urban settlements, and in-part from climate change. Wetlands are often the last remaining natural ecosystems in many parts of the PPR, and they harbor critical habitat for numerous organisms, including nesting and stop-over habitat for the majority of North America's waterfowl. Due to widespread impacts from agricultural practices, the health and condition of wetlands is frequently degraded across the PPR which may affect their ability to support key species. I hypothesized that the presence and productivity of locally breeding waterfowl may be indicators of wetland condition.

I investigated relationships between duckling abundance and measurements of wetland quality to determine the potential use of ducklings as biological indicators of wetland health. In 58 wetlands located at 6 transect sites in central Saskatchewan over 2 years (2018-2019), I examined multiple factors which may determine wetland health including water quality (e.g. pH, conductivity, and Pesticide Toxicity Index), habitat characteristics (e.g. floating vegetation density, maximum water depth, percentage of surrounding grassland, and the extent of degradation of adjacent terrestrial wetland vegetation from agricultural practices), aquatic macroinvertebrate abundance and biomass, and duckling counts of dabbler species derived from bi-weekly surveys throughout the brood-rearing season. I tested relationships between dabbling duckling abundance and several wetland health measurements using generalized linear zero-inflated Poisson models. Model-averaged parameters and 95% confidence intervals (CI) indicated a significant negative effect of conductivity [-0.35 (CI: -0.83, -0.05)], a moderate adverse impact from pH [-0.26 (-0.75, 0.01)], and a slight negative, but nonsignificant effect from pesticides measured using an acute Pesticide Toxicity Index [-3.85 (-8.71, 0.54)]. Based on model-averaged confidence intervals, I found that floating vegetation density negatively impacted dabbling duckling abundance [-0.016 (-0.026, -0.006)], while maximum water depth of wetlands had a positive effect [0.703 (0.391, 1.015)]. Lastly, I found a positive association between aquatic macroinvertebrate abundance and dabbling duckling abundance (SE= 0.1144; P-Value= 0.0374). Based on my findings and the current understanding of the relationship between

wetland characteristics and biological productivity, I conclude that duckling abundance could potentially be used as a biological indicator of wetland health in the PPR. This insight may be useful for wetland conservation efforts in the region due to the high likelihood that human impacts from agrochemicals, drainage, and vegetation removal will continue to increase and the costs to monitor wetlands is high using traditional methods. Therefore, it is necessary to have accessible integrative tools for effectively monitoring wetland health. Based on results of my study, I suggest that surveys of duckling abundance could aid in this effort.

Acknowledgements

In the following section I wish to acknowledge individuals, groups and institution that contributed to the making of this thesis:

My study covered a large portion of the southern region of Saskatchewan and in my duties, I have traveled, worked, and camped in the historic land of the Cree, Saulteaux bands of the Ojibwa, the Assiniboine, Dene Suliné, and Nakota Sioux peoples. During my journey, fellow Indigenous Peoples have befriended me and from their insight of traditional practices they have inspired me in many ways, the most pertinent being a deepened affinity for the Prairies.

The entirety of my data was collected on privately owned land; therefore, if it wasn't for the generosity of those landowners who gave me permission to access wetlands on their land, this project would have remained theoretical. A list of names would be too extensive; however, I would like to thank every landowner for your selfless contributions to this project.

Since I was a child, I dreamed of living in Saskatchewan and studying wildlife of the region. If it weren't for the opportunity presented by my academic advisor, Professor Christy Morrissey, that dream and this thesis would have never come to be. Along with my advisor, my committee members, Dr. Jim Devries and Dr. Bob Clark have influenced and challenged my work; from which I've gained a better understanding of science, reflected in the following pages.

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My greatest supporter cannot speak nor read but I place this acknowledgement here in hopes that he will receive my gratitude. For my Labrador Retriever, Remy, invigorated me every day with his gentile demeanor, joyousness, determination, discipline, and loving disposition.

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"[...] it is a strange thing that most of the feeling we call religious, most of the mystical outcrying which is one of the most prized and used and desired reactions of our species, is really the understanding and the attempt to say that man is related to the whole thing, related inextricably to all reality, known and unknowable. This is a simple thing to say, but the profound feeling of it made a Jesus, a St. Augustine, a St. Francis, a Roger Bacon, a Charles Darwin, and an Einstein. Each of them in his own tempo and with his own voice discovered and reaffirmed with astonishment the knowledge that all things are one thing and that one thing is all things—plankton, a shimmering phosphorescence on the sea and the spinning planets and an expanding universe, all bound together by the elastic string of time. It is advisable to look from the tide pool to the stars and then back to the tide pool again."

-John Steinbeck, The Log from the Sea of Cortez

Dedications

The following thesis is dedicated to two individuals who, in their own ways, have shaped and molded my spirit and without them this thesis would have likely never been.

My thesis, in part, is dedicated in loving memory to my grandmother, Carolyn Anderson, who passed on before the completion of my work. Carolyn was one of the most loving people in my life. If wealth was measured in generosity, selflessness, love and compassion my grandmother was one of the richest people in existence. Her love enriched me in many ways and from those riches, this thesis was a product.

The other part of this thesis is dedicated to my dearest friend, Dr. Jo Campe. Several years ago, I met Jo at a coffee shop in Ely, Minnesota, where we discussed our mutual love for duck hunting and wilderness canoe trips. Unbeknownst to me he was the first person, that I am aware of, who accurately and honestly assessed me. What he found was someone who severely underestimated their own capabilities. From that time on, Jo has mentored me and through his love, patience, joy, and wisdom I have gained an accurate perception of myself. As you read through this thesis, behind each page, paragraph, sentence, and figure there was uncertainty, tribulation, and above all doubt. If it wasn't for Jo, I would have never had the confidence to persevere. I'm unsure where I would be without Dr. Jo Campe's intervention but, certainly my life would not be so happy, joyous, and free nor would I have felt worthy enough to pursue my dream to live in Saskatchewan and study wildlife.

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1. Introduction

1.1. Ecology of Wetlands in the Prairie Pothole Region

The North American Prairie Pothole Region (PPR) is one of the richest, most diverse and unique ecosystems in the world (Montgomery et al., 2021). It crosses the United States/Canada border and contributes land in 5 states (Iowa, Minnesota, the Dakotas and Montana) along with 3 Provinces (Manitoba, Saskatchewan, and Alberta), encompassing more than 770,000 km^2 of land (Doherty et al., 2018). The PPR is characterized by its mixture of grassland and wetlands that are commonly referred to as "prairie potholes". There is a large variation within the region of separate ecosystems that are largely defined by average annual precipitation (Smart et al., 2021). Extreme annual temperature fluctuations, drought-deluge cycles and frequent natural disturbances challenge the region's organisms to be highly adaptable (Neff & Rosenberry, 2018; Dixon et al., 2019; Mcintyre et al., 2019). Extensive alterations by agricultural practices have dramatically changed the historical condition of the PPR and recent studies have shown that this may be limiting or reducing populations of many different species (Buderman et al., 2020; Henry et al., 2020; Smart et al., 2021).

1.1.1. Historic Large-Scale Alterations of the PPR

Natural disturbances such as those created by, for example, wildfires and bison have shaped and changed the PPR since the last ice age and are an integral part of the ecosystem (Pindilli, 2021). Starting in 1830, the settlement of European migrants began widespread man-made change; converting grassland to crop production (Andersen et al., 1996). Agricultural practices have evolved in many ways especially with advancement in farming technologies; however, a general theme of landscape simplification has been common throughout the history of agricultural practices in North America. Landscape simplification is defined by Ricci et al. (2019) as landscape-scale intensification accounting for increases in the area of cropland, decreases in semi-natural habitats and reductions in the length of interfaces between crops and semi-natural habitat. Common modern agricultural practices which produce landscape simplification in the PPR include wetland drainage, shelterbelt and other vegetation removal, mechanical tillage, monocropping, and a heavy reliance on pesticide and fertilizer applications (Schilling & Dinsmore, 2018; Malaj et al., 2020; Ballard et al., 2021).

The scale of alterations by agricultural practices in the PPR is immense. It is estimated that upwards of 75 - 99% of grassland has been altered by agricultural practices and 50 – 99% of all wetlands have been drained (Schilling & Dinsmore, 2018; Mckenna et al., 2019). There is spatial variation in broadscale intensity of agricultural practices in the PPR. Generally the southern portion has the highest amount of landscape simplification; a recent study found that 95 - 99% of wetlands had been drained in Iowa and similar estimates reported from Minnesota (Johnson et al., 2008; Schilling et al., 2019). In comparison, the northern extent of the PPR has lower estimates of wetland drainage; for instance, southern Manitoba is estimated to have 76% of its wetland area altered by drainage (Badiou et al., 2018). This estimate coincides with the approximate median wetland loss of 70% across the Canada portion of the PPR (Pattison-Williams et al., 2018). Despite growing evidence of negative impacts from drainage and subsequent legislation discouraging this practice, the remaining wetlands are drained decennially at an estimated rate upwards of 4% (Minnes et al., 2020). Agricultural practices continue to intensify in the PPR; ultimately, converting a highly biologically diverse region to a simplified landscape of food production for human consumption.

1.1.2. General Ecology of Wetlands in the PPR

Wetland classification systems tend to vary by region however, the basis of the most wetland assessments derives from Stewart and Kantrud (1971) and (Cowardin, 1979). Both systems use vegetation to define zonation and wetland boundaries. Duration of saturation of soils changes biochemical properties and defines soil suitability to certain plant species; hence, the presence of hydrophilic vegetation can allow quick wetland assessments (Moor et al., 2017). Wetlands are divided into 5 different classes based on duration of soil saturation throughout the growing season (Montgomery et al., 2018). Class 1 wetlands are highly ephemeral, typically only retaining standing water for a few weeks after snowmelt. Subsequent classes are have increasingly longer durations of surface soil saturation while class 5 wetlands are defined as permanent bodies of water (Montgomery et al., 2018). Classes 1 – 3 are highly variable, maximum water depths and duration of ponded water can vary dramatically in a growing season dependent on seasonal conditions (Montgomery et al., 2018). To minimize spatial variability I focused on class 4 and 5 (semi-permanent and permanent) wetlands for this project (Mclean et al., 2020).

In general, wetlands in the PPR are defined as marsh and open water wetlands under the Stewart and Kantrud and Cowardin classification system (Stewart & Kantrud, 1971; Cowardin, 1979). The majority of the wetlands are ephemeral and have a maximum water depth of <1 m (Montgomery et al., 2018). Maximum water depth of wetlands can fluctuate dramatically within a growing season; largely, based on climate conditions like ambient temperatures and precipitation (Van Der Valk, 2005). During dry periods the reduction of water exposes mudflats and alkaline soil. The reduction in water level also cause submerged wetland to float on the water's surface (Moor et al., 2017). In nutrient rich wetlands floating vegetation can become matted and limit the amount of exposed open water (Vanausdall & Dinsmore, 2020). Wetland vegetation in the PPR is adapted to the cyclical water level fluctuations, hydrophobic vegetation will recede with reduction in water levels; conversely, reestablishment of hydrophilic vegetation occurs with prolonged periods of soil saturation (Bolding, 2018; Montgomery et al., 2018).

1.2. Threats to Wetlands of the PPR

Common agricultural practices including tilling hydric soils, removing hydrophilic vegetation, and chemical pollution have adversely impacted wetlands. These practices are changing historical dynamics and areas suitable for the biotic life that depends on wetlands. Along with loss of 50-99% of wetlands across the PPR by means of drainage there is also an increase of degradation of buffer zones between farm fields and wetlands (Gleason et al., 2011). Buffer zones are vegetated areas adjacent to wetlands that provide ecological services such as reduction of contaminants that enter water (Rickerl et al., 2000). As a result of loss of buffer zones, wetlands are increasingly more vulnerable to contamination from agricultural inputs. Moreover, since 1971 the use of agrochemicals increased by an estimated 500% across the Canadian PPR (Goldsborough & Crumpton, 1998). Currently, there is no systematic water quality monitoring system in place and therefore the extent of influence chemical contamination of wetlands across this agriculturally dominated region is largely unknown (Main et al., 2014; Malaj et al., 2020).

One group of chemicals that has gained attention in recent years are neonicotinoids, an insecticide class that is prized for its versatility in ways of application as a plant systemic and seed treatment, which is effective against a wide array of invertebrate pests and has greater acute toxicity to invertebrates over vertebrates (Elbert et al., 2008). The most commonly used neonicotinoids in Canada and the US are thiamethoxam, clothianidin, and imidacloprid. These highly effective insecticides are systemic and persist in the plant tissue (Main et al., 2017). After their introduction they rapidly grew in popularity across North America. However, alarming discoveries were made about their effects of non-target species (Whitehorn et al., 2012; Goulson, 2013; Hallmann et al., 2014) and for their ability to persist in soils (e.g., thiamethoxam DT_{50}=avg. 229 days; clothianidin DT_{50}=148–1,155 days) and high water solubility (e.g., thiamethoxam =4,100 mg/L; clothianidin =327 mg/L) (Main et al., 2014). Application of neonicotinoids is often in the form of seed coating or foliar spray (Main et al., 2017). Seed treatments are often applied to canola and a variety of grains which are also the dominant crops of the northern PPR (Rashford et al., 2011).

In 2007, a new class of insecticide was introduced to the market, phthalic and anthranilic acid diamides (i.e. flubeniamide, chlorantraniliprole, and cyantraniliprole) (Hasan et al., 2020). Diamides have the potential to replace neonicotinoids as they are also systemic seed treatments reported to have broad spectrum insecticidal activity (Ebbinghaus-Kintscher et al., 2006). Little is known about the effects of diamides on the environment but they have potential to harm aquatic and terrestrial ecosystems (Ma et al., 2022). The harmful effects of diamides are due to a growing popularity in the insecticide market, ranking third worldwide, and the insecticide is acutely toxic to non-target invertebrates and has detrimental sublethal impacts (Kadala et al., 2020; Ma et al., 2022). Other proposed replacements for neonicotinoids include other plant systemics used as seed treatments: flupyradifurone (butenolide insecticide), sulfoxaflor (sulfoximine insecticide), and flonicamid (pyridinecarboxamide) (Haas et al., 2021; Ullah et al., 2021; Watson et al., 2021). A study on the sublethal effects of flonicamid conducted by Ghelichpour and Mirghaed (2019) found that exposure resulted in an increase in stress and altered gill function in common carp (*Cyprinus carpio*). Siviter et al. (2018) reported that sulfoxaflor reduced reproductive success bumblebee (*Bombus terrestris*). A study by Bartlett et al. (2018) found flupyradifurone had lethal and sublethal effects on mayfly larvae (*Hexagenia* spp.).

The PPR is one of the world's most productive biomes, largely due to its vast area of wetland complexes. In a highly altered landscape, with large scale monocultures, wetlands are often the only remaining unaltered ecosystem (Tews et al., 2004), in-turn providing critical habitat for many PPR species and migratory birds. Given the importance of wetlands in the region, and the potential threats presented, there is an ongoing need to determine reliable and sensitive indicators of wetland condition.

1.3. Waterfowl Population Dynamics in the PPR

1.3.1. General Ecology of Waterfowl in the PPR

The PPR provides critical breeding habitat for 14 waterfowl species including mallard (*Anas platyrhynchos*), blue-winged teal (*Spatula discors*), green-winged teal (*Anas carolinensis*), American wigeon (*Mareca americana*), gadwall (*Mareca strepera*), northern shoveler (*Spatula clypeata*), northern pintail (*Anas acuta*, hereafter pintail), greater scaup (*Aythya marila*) and lesser scaup (*Aythya affinis*, hereafter greater and lesser scaup will be referred to as scaup), redhead (*Aythya americana*), canvasback (*Aythya valisineria*), bufflehead (*Bucephala albeola*), ruddy duck (*Oxyura jamaicensis*), and Canada goose (*Branta canadensis*) (Sorenson et al., 1998). The PPR also provides stop-over habitat for arctic-nesting waterfowl species such as greater white-fronted goose (*Anser albifrons*), snow goose (*Chen caerulescens*), Ross goose (*Chen rossii*), tundra swan (*Cygnus columbianus*) along with taiga nesting waterfowl like: ring-necked duck (*Aythya collaris*), common goldeneye (*Bucephala clangula*), trumpeter swan (*Cygnus buccinator*) (Fredrickson & Reid, 1988). It is estimated that 50 – 80% of the breeding production of ducks in North America occurs in the PPR, while comparatively, this region only makes up 10% of total breeding habitat; earning the reputation as North America's "duck factory" (Klett et al., 1988; Higgins et al., 2002; Niemuth & Solberg, 2003).

The wetland complexes of the PPR offer ideal resources for waterfowl to refuel during migrations and during the breeding season (Johnston & Mcintyre, 2019). Waterfowl have evolved with adaptations to exploit shallow water habitats such as: webbed feet, bill shape and lamellae, relatively high bone density and elongated neck (Pöysä, 1983; Pöysä et al., 1994; Isola et al., 2000). Ducks are split into two guilds based on feeding behavior, dabblers and divers (De Mendoza & Gómez, 2022). Dabblers include species like mallard, pintail, gadwall, etc.; which have evolved to dip the front of their body below the water's surface to forage on aquatic vegetation and macroinvertebrates; while divers, as the name suggests, dive below the water's surface, being able to reach to greater depths to forage (Rylander, 2021). Divers, which include species like common goldeneye, redhead, and bufflehead have specialized body features like high bone density and hips that are further back on the body to help the bird to be agile below the water's surface; however, divers are highly limited when traversing terrestrial terrain (Stephenson, 1993). A limited ability to forage terrestrially, may make diver ducks more dependent on wetlands; thus, theoretically more sensitive to agricultural impacts.

A generalized summary across all waterfowl species commonly found in the PPR is that an increase in protein consumption (i.e. aquatic invertebrates) begins during the spring migration and peaks during the breeding season; diet transitions to a largely herbivorous diet (seeds and various plant matter) later in the breeding season and throughout the non-breeding period (Stafford et al., 2014; Stafford et al., 2016). Dietary preferences during the breeding season vary by species ranging from more herbivorous species like gadwall (Ankney & Alisauskas, 1991) or canvasback (Bartonek & Hickey, 1969) to carnivorous species like scaup (Afton & Hier, 1991) or northern shoveler (Ankney & Afton, 1988). Relative to other avian species, waterfowl lay large and nutrient rich eggs that are produced at a high caloric cost often leaving hens with a protein and lipid deficit. Moreover, migration to the breeding grounds in the PPR often reduces the majority of the fat reserves before nesting and later egg laying may occur as a result (Yarrow, 2009; Stafford et al., 2016). Therefore, it is critical that breeding waterfowl have access to an ample abundance of macroinvertebrates on arrival in the PPR (Stafford et al., 2016). Energetic demands remain high throughout the breeding season to support growth of precocial young that begin feeding on aquatic macroinvertebrates typically within 24 hours of hatching (Sedinger, 1992). The distribution and abundance of aquatic macroinvertebrates can influence habitat use, behavior, and growth rates of broods and, thus, is commonly the focus of wetland management strategies in the PPR (Krull, 1970; Hanson & Butler, 1994; Sheehan et al., 2020).

1.3.2. Waterfowl Population Trends in the PPR

Several waterfowl species recovered from the brink of extinction in the early 1900s to meeting or exceeding population goals (Johnson et al., 2010; Roberts et al., 2018). Largely due to extensive management techniques including hunting regulations (harvest limit and methods, etc.) (Nichols et al., 2007), habitat conservation (Rashford et al., 2011) and further research; has in-effect, increased population levels (Johnson & Case, 2000). The survey known as the Waterfowl Breeding Population and Habitat Survey (WBPHS) is the largest collective waterfowl survey in North America and probably the largest wildlife survey in the world (Lewis et al., 2016). The WBPHS encompasses the PPR, expanding across the northern portion of North America covering more than 5.18 million km^2 (Boere et al., 2006). Conducted during each spring by the United States Fish and Wildlife Service (USFWS) and Canadian Wildlife Service (CWS), the survey has been orchestrated annually since 1955 (Service, 2012). The overarching objective of this survey is to estimate population size of waterfowl in North America; in-part, to guide migratory game bird hunting regulations of the corresponding year (Fish & Service, 2019).

A recent paper by Rosenberg et al. (2019) on avian population declines across North America used data collected during the WBPHS since 1970. This team estimated that North America has lost 3 billion birds since 1970 or 29% of total abundance. Researchers estimated that waterfowl are the only taxonomic avian order found in North America that had increased since 1970. While this finding is striking, the results may be misleading based on the temporal scale of the analysis. When the WBPHS started it was during a time waterfowl populations were at alarmingly low levels (Brasher et al., 2019). Since that time, conservation efforts like regulated hunting, migratory bird reserves, habitat improvements, etc. have been implemented and have drastically improved waterfowl numbers (Williams et al., 1999). Also, grouping all waterfowl species together is likely enhancing population trends because some waterfowl species, particularly granivorous species, like snow or Ross geese, are exceeding carrying capacity in some flyways (Dooley & Brook, 2018; Sears & Miller, 2018). This is largely attributed to the geese having ample supply to high caloric food resources (i.e. grain spillage during harvest) during migration (Sears & Miller, 2018). Duck population trends are less abundant in comparison to some goose species. Based on the latest published results of the

WBPHS titled '*Waterfowl Population Status, 2019*' on average, duck population levels in the PPR have increased marginally since 1955 (+10%; p-value <0.001).

On average, waterfowl populations in the PPR have improved and stabilized since 1955, especially in comparison to other avian species that breed in the PPR like obligate grassland birds (Li et al., 2020) or aerial insectivores (Tallamy & Shriver, 2021). However, it is well established that waterfowl population trends closely follow habitat quantity and quality (Lamb et al., 2018; Schrempp et al., 2019). For example, duck population trends are highly correlated with water availability (Sedinger et al., 2019). As discussed above, wetlands of the PPR have been drained or degraded since European establishment in the PPR (Schilling & Dinsmore, 2018); moreover, recent studies predict that global warming may increase wetland loss through evaporation (Chen et al., 2018; Zhang et al., 2021). Potentially, waterfowl populations may decrease with further wetland loss. Hence, the retention of intact wetlands is critical; furthermore, there is a need for knowledge on how the quality of the remaining wetlands may influence waterfowl along with other wildlife species that are dependent on wetlands of the PPR.

1.4. Impacts of Intensive Agricultural Practices on Waterfowl in the PPR

The PPR is estimated to be the most productive breeding area for waterfowl but also the most heavily impacted by agricultural practices in North America (Brasher et al., 2019). Modern agricultural practices often rely on intensive methods to produce high yields of crops, which manipulates, degrades and changes historic conditions of the PPR (Henry et al., 2020). The expansive change to the region since mechanization has drastically shifted historically productive grassland-wetland complexes; which in-turn, continue to impact breeding waterfowl (Williams & Sweetman, 2019). Agricultural practices such as wetland drainage, tillage, pesticide application, etc. adversely impact waterfowl by reducing the quality and quantity of available habitat. Ultimately, changes in wetland health can affect waterfowl directly or indirectly, during a nutrient-demanding stage of the waterfowl annual life cycle (Doherty et al., 2018; Aagaard et al., 2022). In the scope of this project, I define a direct impact on waterfowl as an external action that directly influences waterfowl survival and reproduction; conversely, an indirect impact is an external action that influences organisms that waterfowl rely upon (i.e. aquatic macroinvertebrate).

1.5. Ducklings as Biological Indicators of Wetland Health

A biological indicator or bioindicator is defined as a species that respond predictably to environmental change which is representative of other taxa present in the environment (Quigley et al., 2019). Waterfowl ducklings meet the criteria defined by Quigley et al. (2019) for both of these bioindicator types. The use of bioindicators aids in reducing the complexity of environmental change down to empirically derived units of information which can influence conservation efforts (Heink & Kowarik, 2010). In this context, I hypothesize that duckling abundance, can be a useful proxy of wetland health reflecting the often-cryptic influence of agricultural practices that may degrade wetland health.

Nest site selection of waterfowl is highly complex and influenced by broad factors like habitat availability or more localized features like competition, food availability, vegetation structure; furthermore, there is variation amongst species and individuals (Lokemoen et al., 1984). While factors like limited habitat availability can influence hens to select wetlands or areas that may not be as suitable for rearing young; hence, brood success, defined here as at least one young reaching to the full fledge stage, may be low (Buderman et al., 2020). The presence of a nesting pair may have a limited ability of detecting wetland health as the wetland may be perceived by the nesting pair as a suitable nesting area; however, limited resources may reduce the likelihood of rearing success (Ratti & Reese, 1988). Moreover, a nesting pair is highly mobile and the presence of a pair at a single wetland may be distant to the location of the nest (Brewster et al., 1976). The presence and continued observation over the rearing season of precocial young may be a strong indicator of wetland health because the wetland demonstrates the capacity of the habitat to support duckling production and sustain young through the rearing season (Dzus & Clark, 1997). Therefore, I investigated what wetland characteristics influence duckling abundance; with the goal to investigate if duckling abundance can be used as biological indicators of wetland health.

1.6. Study Objectives

The complex interactions in the PPR between the remaining natural ecosystems and agricultural practices present challenges for wildlife populations that are important to assess. A biological indicator may aid in effectively determining the health and condition of wetlands in the PPR to support biodiversity. Although research has been done on aquatic macroinvertebrate communities as biological indicators of various water body conditions, there are a few critical logistical constraints that limit their usefulness for wetland ecosystems. For example, sorting and identification protocols are commonly time consuming; especially, in eutrophic water bodies like wetlands. Precocial ducklings are highly dependent on a variety of aquatic macroinvertebrates throughout the rearing season and their diets vary by species and life stage. Therefore, it is likely that duckling abundance could be an index of wetland health, allowing easy interpretation of the numerous direct and indirect effects of agricultural practices. Merely determining a relationship between ducklings and aquatic macroinvertebrate abundance may be insufficient at indicating wetland health. Other potential factors which reflect wetland health such as the presence or concentrations of pesticides in water, vegetation composition and surrounding land use in combination with invertebrate communities provide a set of powerful tools to assess wetland condition. To my knowledge, an assessment of duckling abundance as a biological indicator of wetland health has not been completed. The overarching goals of my research were to provide an improved understanding of wetland-specific factors that could affect wetland use by brood-rearing female ducks and to evaluate the possible use of dabbler duckling abundance as a biological indictor of wetland health in the PPR. I hypothesized that duckling abundance is an effective biological indicator of wetland health such that counts of ducklings would be related to wetland characteristics (e.g. macroinvertebrates, water quality, etc.). More specifically, I predicted that wetlands surrounded by more agro-intense cropping (e.g. increased insecticide pollution, vegetation disturbance, etc.) will have fewer ducklings than wetlands from less impacted sites.

My four main study objectives were to:

I. Estimate dabbler duckling abundance on Saskatchewan wetlands that vary in agricultural intensity.

II. Determine which wetland habitat features are related to duckling abundance.
III. Evaluate the relationship between duckling abundance and aquatic macroinvertebrate prey abundance and biomass; and,
IV. Determine the relationship between duckling abundance and water quality metrics (physiochemical and insecticide pollution).

2. General Methodology

2.1. Study Sites

Selection of study sites for my surveys was determined from evaluation of three criteria: surrounding land use, wetland density and standing water availability. This approach allowed me to capture a large range of conditions that may influence waterfowl across the prairie portion of Saskatchewan. In the context of this project, my study sites were transects defined by a pre-existing road - commonly referred to as a “grid road”. Typically, in the prairie portion of Saskatchewan these roads rarely deviate from a straight East to West or North to South direction and are set every 1.6 km (1 mile) longitudinally and every 3.2 km latitudinally. The length of the transects in this study varied by location and averaged 16 km. To narrow site selection across southern Saskatchewan I selected from transects used in the WBPHS conducted annually by the United States Fish and Wildlife Service (USFWS) and Canadian Wildlife Service (CWS). A detailed review and description of methodology of the WBPHS can be found in (Smith, 1995). To minimize spatial variability I focused on class 4 (semi-permanent) and 5 (permanent) wetlands (Mclean et al., 2020). There were 6 transect sites with roughly 10 wetlands per transect (n = 58 wetlands total), located within 250 km from Saskatoon (n= 3 in 2018, n= 3 in 2019): 2018 [Ibstone (52°34'52.9"N 108°23'57.9"W), Tichfield (51°14'32.5"N 107°12'29.5"W), Peterson (52°08'37.4"N 105°17'21.8"W)]; and, 2019 [Buchanan (51°42'27.1"N 102°55'07.9"W), Grayson (50°43'30.0"N 102°40'50.4"W), and Zealandia (51°38'59.9"N 107°21'13.2"W).] All sites were selected based on similar density of wetlands and water levels, and presence of waterfowl but with variation in agricultural intensity that was hypothesized to affect water and habitat quality. Wetlands were separated by 500 m to 2 km, within view of the road, and chosen to allow an observer to view the majority of the open water area from the roadside. For example, an ideal wetland would have minimal trees or shrubs and be large enough to hold water throughout the survey periods but be appropriately sized so the observer could view the entire wetland from a stationary position. At each wetland from 6 transects, I conducted repeated duckling surveys, water sampling, macroinvertebrate collections, and a wetland habitat assessment (Figure 1).

2.2. Duckling Surveys

Duckling surveys were conducted over four different observational periods to count waterfowl ducklings including 16 different waterfowl species, the majority of which are commonly found in the region including: mallard (*Anas platyrhynchos*), northern pintail (*Anas acuta*), northern shoveler (*Spatula clypeata*), gadwall (*Mareca strepera*), blue-winged teal (*Anas discors*), green-winged teal (*Anas carolinensis*), American wigeon (*Mareca americana*), redhead (*Aythya americana*), canvasback (Aythya valisineria), scaup (*Aythya*), bufflehead (*Bucephala albeola*), ruddy duck (*Oxyura jamaicensis)*, and Canada goose (*Branta canadensis*). Also included was species that are less commonly found in the region: scoter (*Melanitta*), goldeneye (*Bucephala*), and cinnamon teal (*Anas cyanoptera*). Other waterbird young were included in the survey such as coots (*Fulica*), various grebes (*Podicipedidae*) and mergansers (*Mergus*). The rounds were separated by 14 day intervals beginning in June and lasting until late August to capture the full production phenology of multiple species (Rumble & Flake, 1982; Johns, 2019). Round 1-3 start time was 05:00 and Round 4 started at 05:30. Round 1 and 3 were surveyed from east to west on transect, and Round 2 and 4 was west to east. Each round had two survey periods. The second period began immediately after the first. It took approximately 2 hours to survey all wetlands on the transect. Survey weather conditions needed to be favorable with winds under 25 km/h, no precipitation, and visibility greater than 100 m. The data recorded during the survey included: start time, ambient temperature, wind speed, wind direction, precipitation, water level, cloud cover, species, counts and age class. Surveys are conducted in a passive approach, using a 4-wheel drive vehicle as a blind. At each wetland, I set up a spotting scope or binoculars and record the survey window of 6 minutes with a stopwatch. The survey team consisted of an observer, who is experienced in waterfowl identification paired with an experienced transcriber.

All ducklings were counted during the 6-minute interval. Data included species, number of ducklings within the brood, and age class. A second observation period began immediately after the first concluded and followed the same protocol. This was done to correct for observational error which commonly occurs during brood surveys do to a hen's cryptic behavior (Carrlson et al., 2018). A second observation is commonly used during waterfowl surveys to account for missed observations (Harms, 2021; Roy et al., 2021). The highest of the 2 repeated counts done on the same day was used.

2.3. Aquatic Macroinvertebrate Sampling

Sampling procedures were conducted once at each of the study wetlands (n=58 on 6 transects) following the Canadian Aquatic Biomonitoring Network Wetland Macroinvertebrates Protocol (CABIN; http://www.ec.gc.ca/rcba-cabin/). The CABIN protocol was developed by Environment and Climate Change Canada to assess the condition of aquatic ecosystems across Canada (Reynoldson et al., 2003). Determination of wetland condition is based on aquatic invertebrate population abundance and diversity. Sampling was timed to correspond with the peak of the growing season (mid-June through July). Requirements for sampling included: two crew members, a sampling net with 400-µm mesh size, stopwatch, waders, sample jars and preservative (70% ethanol). Sampling for aquatic macroinvertebrates involved moving the net through and around the wetland submerged or semi-submerged vegetation for 2 minutes to collect a diverse range of taxa. One member of the crew used the sampling net, while the other member keeps track of sampling time. Collected samples were stored in 1 L containers and stored in 70% ethanol. Samples were kept in a refrigerator, until the time of sorting. During sorting samples had the vegetation removed, and invertebrates were sorted and identified to taxonomic Order using guidelines found in the CABIN macroinvertebrate protocol (Reynoldson et al., 2003) and later counted. The samples collected in 2018 were counted to entirety while 2019 samples were subsampled using a Marchant Box and total abundance was then estimated by multiplier.

Macroinvertebrate samples were prepared before being placed in the Marchant Box by being placed in a 5-gallon bucket filled with water and gently agitated to separate macroinvertebrates from debris and vegetation which were removed from the sample. Samples were then poured through a 400-μm mesh sieve to remove fine sediment from the sample. Rinsed samples were transferred into a 100-cell Marchant box along with enough water to fill all Marchant box cells. The Marchant box was agitated until the macroinvertebrate sample was evenly distributed across all cells. A random number generator was then used to select multiple cells from the 10 x 10 grid of 100 Marchant box cells. All invertebrates from the selected cell were sorted into vials containing 70% ethanol by order and tallied. Cells were randomly selected, and subsamples sorted until two criteria were reached: 1) a minimum of 5 cells were sampled and 2) a minimum of 300 insect macroinvertebrates had been tallied. A sample estimate was derived from dividing the total number of macroinvertebrates in the subsample by the total number of used cells then multiplied by 100. To examine the level of estimate accuracy, 5 of the samples were randomly selected after subsampling and the entirety of the sample was sorted and counted. Dividing the Marchant Box subsampling estimate to the total count indicated an average of 87% accuracy for the estimate. When all samples were sorted and counted, macroinvertebrates were dried for 48 hours in a drying oven at 60°C and weighed to record biomass.

2.4. Wetland Assessment

The protocol to assess the study wetlands was a slightly modified version of Rapid Wetland Assessment Criteria that include both quantitative and qualitative parameters (Main et al., 2015). At each wetland, measurements of 13 predictor variables were collected (see Table 1 for a full list of variables measured). Water depth was measured in approximately the deepest area of the wetland, typically in the central portion of the pond. I estimated percentages of broad characteristics such as surrounding cropland or wooded area based on field observation. The percentage of floating vegetation was estimated by recording visible vegetation that was floating on the water, which typically intertwined with adjacent plants and formed mats or vegetation that has floating leaves that obstruct the surface of the water. Percentage of bare substrate was estimated by the amount of exposed sediment lack of submerged aquatic vegetation. Basin fill was a field-based observation estimating the level of water relative to the approximate maximum capacity of the wetland basin. Determining factors that influenced this estimate include surrounding land topography, time of year, wetland vegetation zones and seasonal climate conditions. The distance of degraded wetland vegetation was determined by selecting four cardinal points around the wetland measuring the distance from the outermost point where hydrophilic vegetation was present but, altered more than 25% of the natural condition by agricultural practices (e.g. cropped plants, tillage, herbicide spray or haying) to the boundary where hydrophilic vegetation is still intact. For example, the outermost wetland boundary may have been determined by the presence of horsetail (*Equisetum)*in a canola dominated area, from that boundary a measurement was taken towards the open water area to the edge of cattails (*Typha*) which were mostly unaltered. In between these two boundaries the measurement of vegetation degradation was derived. This method was repeated for the remaining transects around the wetland and then the 4 transects around the wetland were averaged to produce a single variable.

2.5. Water Quality Sampling and Analysis

Levels of insecticide pollution in water were assessed through analysis of neonicotinoid and diamide insecticides (Jeanguenat, 2013; Main et al., 2017). Water sampling was conducted once a year after seed germination (late June or early July). Water was collected with one chemically-cleaned (acetone: hexane washed) 1 L amber glass bottle used to collect a subsurface sample (10-15 cm)(Main et al., 2017). Collected samples are stored and chilled in the field aided by coolers filled with ice. Samples were stored at 4°C until ready for analysis. The samples were analyzed for neonicotinoids and replacements at the Environment and Climate Change Canada National Hydrology lab in Saskatoon, Saskatchewan. Water sample analyses for other pesticides, herbicides, fungicides, and nutrients was considered however, due to logistical constraints were not included in my study. Lastly, I collected basic water quality data [dissolved oxygen (mg/L), conductivity (µS/cm), pH and temperature] with the use of a handheld water quality meter (YSI ProPlus multimeter).

Water samples were extracted for insecticide analysis by passing 500ml through Oasis HLB cartridges which were conditioned with methanol (10 mL) and water (10 mL). Once sample loading was complete, the cartridges were then washed with de-ionized water (5 mL) and dried under vacuum for 5 minutes. The mixture was eluted with methanol (10 mL), then evaporated to dryness and the extract residues reconstituted in 500 mL of water followed by addition of internal standards. Analysis of samples was carried out by Liquid Chromatography with tandem mass spectrometry (LC-MS-MS). A Waters 2695 Alliance HPLC system consisting of a solvent degassing unit, pump and autosampler was used with a Waters Xterra MS-C8 (3.5 mm dia. particle size) column (2.1-6100-mm) at 30uC. A uniform elution of the analytes was accomplished with an 80/20 mix of solvent A (99.9% water and 0.1% formic acid) and solvent B (89.9% acetonitrile, 10% water and 0.1% formic acid). The injection volume was 20 mL with a run time of 10 minutes. Quantification of insecticides were done using an internal standard method then presence was confirmed using the Micromass Quattro Premier triple quadrupole mass spectrometer equipped with an electrospray ionization interface set to positive ion mode. To optimize ionization and MS-MS by infusing a 0.5 mg/L solution of each insecticide into an ion source in a 50:50 (v/v) acetonitrile water solution with a syringe pump. A four-level calibration curve (5 to 50 mg) was analyzed before and after each batch of 10 samples which also contained a laboratory blank and a fortified sample. Limits of quantification (LOQ) in water were as follows: imidacloprid, 0.0010 µg/L; thiamethoxam, 0.0013 µg/L; clothianidin, 0.0011 µg/L; acetamiprid, 0.0003 µg/L; chlorantraniliprole, 0.0003 µg/L; cyantraniliprole, 0.0008 µg/L; flonicamid, 0.0008 µg/L; flubeniamide, 0.0008 µg/L; flupyradifurone, 0.0006 µg/L; sulfoxaflor, 0.0010 µg/L. A correction for % recovery was made on all insecticide concentrations and all laboratory blanks were non-detectable.

2.6. Data Analyses

I considered using the hierarchical abundance models (Royle, 2004) within the Program R package *unmarked* (Fiske & Chandler, 2011). However, the study protocol was not adequate to meet the assumptions of the models (population closure) and I lacked sufficient sample sizes to reliably estimate detection probabilities. While there are options to relax some assumptions, it is likely they would require more data than available. To address the likelihood of counting the same brood twice I took the maximum count from same day observational visits per species for each survey period. Maximum individual species counts were then summed to generate total counts of ducklings and other young water birds for each survey. I then separated duckling species into guild (dabbler or diver) and summed the average counts across all survey periods. To better focus the analysis, other water bird species were excluded from the analysis. Canada goose goslings were also excluded largely due to goslings being primarily grazers (Brook et al., 2015). Diving duck ducklings were also removed from the analysis due to an overabundance of zero counts resulting in quasi-complete separation of test models.

I quantified the contribution of insecticides to the overall wetland mixture toxicity based on the pesticide toxicity index (PTI) (Nowell et al., 2014). This method has proven to be a robust method to evaluate toxicity of pesticide mixtures, especially when evaluating aquatic invertebrate communities. In general, this index combines measurements obtained from two classical approaches, toxic unit (TU) and concentration addition (CA). Species sensitivity distributions (SSD) were calculated following Malaj et al. (in prep) and comprised literature values of acute toxicity tests conducted on aquatic benthic invertebrate species in order to derive an acute HC5 (a hazardous concentration for 5% of species) for each of the tested insecticides. The acute HC5 value was then used to weight the insecticide concentrations found in the water samples by dividing the concentration detected in the water sample by the HC5 of the corresponding insecticide. The derived value relative to the HC5 value of all detected insecticides per wetland were summed to give a single value of acute PTI. Chronic PTI was considered for analysis; however, multiple water samples with varied sampling periods within a growing season, are necessary to derive chronic PTI, which was not logistically possible. Hence, only acute PTI was included in my analyses.

To minimize the number of explanatory variables, I used Principal Component Analysis (PCA) to select habitat parameters which explain a larger portion of variance in comparison to other variables (Brown, 2009). For the PCA, 10 habitat parameters were used (percentage of cropland, percentage of wooded area, percentage of roadway, percentage of grassland, percentage of algae, percentage of bordering wetland, percentage of bare bottom, percentage of floating aquatic vegetation, average distance of disturbed wetland vegetation, maximum water depth). PCA was performed in R using the package *vegan* (Oksanen et al., 2013).

Spearman r-correlation indices of grouped variables (water quality, habitat, etc) were all less than 0.60 confirming non-collinearity of the variables. I fit models using Program R package *glmmTMB* (Brooks et al., 2017) with zero-inflated Poisson (ZIP) distribution, as count data are commonly Poisson distributed (Joe & Zhu, 2005). The duckling data contained an overabundance of zeros as no ducklings were present at 48% of wetlands. I initially explored use of models with negative binomial and Poisson distributions, but the ZIP models yielded a better model fit. In each model, I included a random effect of transect to account for spatial and temporal variation not accounted for by covariates. I also included a log link offset term of “open water area”. I then used similar explanatory variables to build each model, dividing analyses into three separate categories: effects of water quality, influence of habitat characteristics and effects of food availability and composition. To improve assumptions and normality, I log10 transformed 2 variables: aquatic macroinvertebrate abundance and conductivity. I used Akaike’s information criterion (AIC_c) for small sample sizes to select the most parsimonious model via the model.sel function from the ‘mumin’ package (Barton & Barton, 2015). For models with similar AIC_c (<2) I used model.avg function from the ‘mumin’ package (Barton & Barton, 2015) and *AICcmodavg* (Mazerolle, 2020) to average across models to calculate parameter estimates and 95% confidence intervals (CIs). Parameters were considered significant if the 95% CIs did not overlap zero. I considered relaxing model constraints however, this is not a common practice of studies with a similar design as mine (Denes et al., 2015; Kemink et al., 2019).

3. Results

3.1. Duckling Community Composition

I surveyed 58 wetlands for ducklings bi-weekly from June – August (n = 26 wetlands in 2018, n= 32 in 2019). During the first observation period, which started close to sunrise, I typically observed twice as many ducklings (1024 individuals) compared to the 2^{nd} period (467 individuals). Observed species included mallard, gadwall, green-winged teal, blue-winged teal, northern shoveler, northern pintail, redhead, canvasback, scaup, ring-necked duck, bufflehead, and ruddy duck. Dabbling ducks were more commonly observed than diving ducks, with blue-winged teal having the most observations of any species. Diversity also varied across transects with a maximum of 10 duck species at Buchanan and a minimum of 3 at Peterson. The peak number of duckling observations occurred during the later 2 survey periods (late July- early August). Diving duck ducklings were excluded from the statistical analysis based on too few observations resulting in quasi-complete separation in test models. (Table 2; Figure 2).

3.2. Effects of Water Quality on Dabbling Duck Duckling Abundance

Wetland water quality was sampled once for each of the 58 wetlands in June of each year (2018-2019) for a total of 58 water samples (n= 26 for 2018; n= 32 for 2019). Insecticides were detected in 46.2% of the wetlands in 2018 (max. concentration 2.1953 μg/L) and in 56.3% of wetlands in 2019 (max. concentration 0.0242 μg/L). The number of insecticide detections was highly variable across transects. Among transects, the highest number of detections ranged from 16 with the corresponding maximum concentration of 2.1953 μg/L to 1 with corresponding minimum detected concentration (0.0004 μg/L). The average conductivity of wetlands of each transect were highly variable, with Grayson having the highest average conductivity (3062 μS/cm) and Ibstone with the lowest average (1257 μS/cm). In comparison there was little variance of average pH between transects, being highest at Peterson (9.3 pH) and lowest at Tichfield (8.45 pH) (Table 3).

Results based on AIC model selection suggest that the most parsimonious model included both pH and conductivity as predictors of duckling abundance. Based on the GLM ZIP water quality model I found that both high conductivity and high pH negatively influenced duckling abundance (Table 4-5; Figure 3). I found a negative effect of acute PTI on ducklings, however, the small change in AIC_c and model-averaged confidence intervals overlapped zero indicating that the effect was non-significant. Overall, model-averaged predictions indicated a significant negative effect from conductivity and a moderate impact from pH (Table 4-5; Figure 3).

3.3. Effects of Habitat on Dabbling Duck Duckling Abundance

Wetland habitat assessments were characterized by 10 variables that included wetland features and surrounding landscape characteristics (Table 1). My PCA indicated that aquatic floating vegetation density, maximum water depth, percentage of surrounding grassland, and the degradation of wetland vegetation from agricultural practices had the highest explanatory power and low collinearity (Figure 4).

On average there was variation of habitat variables between transect sites. Maximum percent grassland was found at the Zealandia transect (21.9 %) while the minimum was at

Grayson (11.4%); Maximum floating aquatic vegetation was found at Peterson (49.5%) while the minimum was at Zealandia (8.3%); Maximum average water depth was at Buchanan (0.82 m) and lowerst at Tichfield (0.21 m); and the maximum wetland vegetation degradation based on the tillage boundary was at Ibstone (40 m) and minimum was at Zealandia (19.5 m). Wetlands within transects had higher variability in characteristics such as water depth and metrics of agro-intensity (Table 1).

Based on the GLM ZIP habitat model, water depth and density of aquatic vegetation significantly influenced dabbler duckling abundance (Table 6). Duckling abundance was negatively impacted by floating aquatic vegetation encroachment of open water and was positively associated with maximum water depth (Table 6-7; Figure 5).

3.4. Effects of Macroinvertebrate Prey Availability on Dabbling Duck Duckling Abundance

In total, I found 11 different Orders of aquatic macroinvertebrates. The abundance and community structures of aquatic macroinvertebrates were highly variable across transects. The maximum occurred at the Buchanan transect (10 Orders) and the minimum at Peterson (3 Orders). The mean total number of aquatic macroinvertebrates was found at the transect Grayson (464.6) while the minimum was recorded at Peterson (63.2); mean aquatic macroinvertebrate biomass was found at Tichfield (4.84 g) and the minimum was at Buchanan (0.26 g) (Table 8).

There was a positive Spearman correlation between aquatic invertebrate biomass and abundance ($r=0.626$); however, abundance provided a better model fit so I elected to use aquatic invertebrate abundance and excluded biomass from the analysis. The global model containing aquatic macroinvertebrate abundance was more parsimonious than a null model based on AIC_c (SE = 0.1144; P-Value = 0.0374). I inferred that higher aquatic invertebrate abundance had a positive influence on duckling abundance (Table 9; Figure 6).

3.5. Comparative Analysis of Multiple Explanatory Models

To assess the most important variables from separate models above (top models for habitat, water quality, and food availability), I selected the variables from each top model that significantly impacted duckling abundance and then combined these variables into a single GLM

ZIP model. These variables include conductivity, floating aquatic vegetation density, maximum water depth and aquatic invertebrate abundance. I then used AIC_c and model weight to determine the most parsimonious model containing the explanatory variables of conductivity and floating aquatic vegetation density. All of these variables were retained and were significantly related to duckling abundance (Tables 10 and 11).

4. Discussion

4.1. Primary Findings

Growing concerns over the effects of intensive agricultural impacts, specifically on wetlands of the PPR, has led to a need to determine wetland health. My study presents new insights regarding how duckling abundance is influenced by agricultural impacts. My results indicated that variables from each component of wetland health (habitat, water quality, and food availability) influenced dabbling duck duckling abundance. These results support the idea that duckling abundance could be used as a biological indicator of wetland health.

At my study sites, variability in wetland condition was likely influenced by surrounding agricultural practices. Both study years (2018-2019) had a large reduction in water availability throughout the growing season, with 2019 being the drier of the two years. Maximum water depth can be interpreted as water permanency. Shallow wetlands tend to encourage evapotranspiration, due to an increase in surface area and higher water temperatures compared to wetlands with deeper basins (De La Fuente & Meruane, 2017). My finding that water permanency was a predictor of duckling abundance is in line with several previous studies (Pietz et al., 2003; Krapu et al., 2006; Bloom et al., 2012). Previous research indicates that water permanency is a better predictor of duckling abundance during drought conditions, when surface water is scarce (Guntenspergen et al., 2006). Because of this, the results of my study indicating the importance of wetland depths should be interpreted in the context of long-term hydrologic cycles. My study occurred near the beginning of a multi-year drought and therefore it was predictable that wetland depth would be important, whereas in wetter periods the result may have been marginal or not detected. It is likely that weather events will increase in variability based on predictions of increased ambient temperatures and variable precipitation events (Mcintyre et al., 2019; Zhang et al., 2020); compounded by further drainage of existing wetlands (Daniel et al., 2022). Furthermore, it is critical to understand key wetland conditions which are beneficial to support a diverse array of biological life.

My results suggest that standing water in wetlands with lower conductivity and neutral pH levels supported the highest number of dabbling duck ducklings. This finding is supported by previous studies that found higher conductivity wetlands increases duckling mortality (Schacter et al., 2021). Compared to conductivity levels reported by Schacter et al. (2021) in coastal wetlands in California, wetlands sampled in Saskatchewan tended to have lower conductivity and ducklings exhibited a higher sensitivity to conductivity ranges. Ingestion of water with a salinity levels as low as 2 ppt (3600 μS/cm) can impair duckling growth and influence behavior, with mortality occurring above 9 ppt (14800 μS/cm) (Schacter et al., 2021). It is known that water parameters such as conductivity and pH can effect ducklings indirectly by impacting aquatic macroinvertebrate communities (Lovvorn & Crozier, 2022). Similarly, insecticide pollution in wetlands from agricultural practices reportedly impacts aquatic macroinvertebrate abundance (Sumudumali & Jayawardana, 2021).

I hypothesized that insecticide pollution would be a strong predictor of duckling abundance; however, my results only weakly supported this idea. It may be that the range or types of pesticide concentrations I measured during a single sampling event were not high enough to predict a reduction in the abundance of insects and in turn affect duckling abundance. Insecticide pollution in wetlands has been shown to be prevalent in the PPR. Contamination by insecticides, notably neonicotinoids, occurs frequently in wetlands across the PPR, including in this study. Main et al. (2014) detected insecticides in 62% of wetlands in spring and summer in Saskatchewan and my detection frequency was somewhat lower (44.8%). An unpublished study by Malaj et al. (2020) assessing the distribution and concentration on neonicotinoids across the PPR found similar results to mine, with neonicotinoids detected in 29% of wetlands while mean concentrations varied (0.0108 μg/L); similar to my findings (0.0565 μg/L). Potentially, insecticide pollution did adversely impact aquatic macroinvertebrate communities in my study wetlands based on the growing evidence that neonicotinoids are acutely toxic to non-target invertebrates and exhibit detrimental non-lethal impacts to vertebrates and invertebrates (Eng et al., 2019; Schepker et al., 2020). Further support of this comes from a study conducted in central

Saskatchewan by Cavallaro et al. (2019) which concluded that neonicotinoid pollution in wetlands significantly reduced aquatic macroinvertebrates and altered community composition. Aquatic macroinvertebrate abundance was clearly predictive of duckling abundance in my study wetlands. The abundance of aquatic macroinvertebrates is critical for the growth and survival of ducklings, more so than biomass or abiotic factors (Bataille & Baldassarre, 1993). In a comprehensive study of variables which determine dabbler duckling abundance, Seymour and Jackson (1996) examined 20 abiotic (pH, nutrient loading, water depth, etc.) and biotic (aquatic macroinvertebrate abundance, macrophyte community structure, etc.) variables of 32 lakes in Nova Scotia. Findings suggested that quality duckling habitat was defined by eutrophic areas which support abundant macrophyte and invertebrate communities; moreover, aquatic macroinvertebrate abundance was the most important factor which influenced duckling abundance.

The abundance of aquatic macroinvertebrates is well established as a highly significant factor which influences nesting waterfowl and duckling abundance (Seymour & Jackson, 1996; Gurney et al., 2017; Kahara et al., 2022); equally, aquatic macroinvertebrates demonstrate sensitivities to environmental factors which influence abundance thus macroinvertebrate communities are often used to determine water quality (Castillo-Figueroa et al., 2018; Collins et al., 2019; Dallas, 2021). My result that aquatic macroinvertebrate abundance was a predictor of duckling abundance is a strong support of duckling abundance as a biological indicator of wetland health. A recently thesis by Wade (2021) examined the role of terrestrial vegetation between the cropland and wetland, which acts as a contamination sink or commonly referred to as a buffer. The study area was located in central Saskatchewan and the wetland characteristics closely resemble wetlands of my study. Within the study, aquatic macroinvertebrate communities within the wetland were used as a biological indicator for the quality of the wetland- further supported by abiotic variables like PTI and other agricultural contaminants. Wade's (2021) findings suggest that pesticide pollution (PTI) was a significant predictor of aquatic macroinvertebrate abundance; furthermore, aquatic macroinvertebrate communities were a better predictor of agro-intensity than surrounding land use (e.g., landscape simplification). Contrary to expectation, the vegetative structures of the wetlands did not influence aquatic macroinvertebrate communities.

My finding that duckling abundance is related to aquatic macroinvertebrate abundance is consistent with other studies, and collectively suggests that insecticide pollution could influence duckling abundance indirectly by impacting aquatic macroinvertebrate communities. This is further supported by my findings that water parameters (pH and conductivity) significantly predicted duckling abundance. Furthermore, dabbling duck ducklings may not be as sensitive to aquatic macroinvertebrate communities compared to diving duck ducklings, especially among older dabbling duck ducklings. Hence, a need to further examine the sensitivity of guilds and species to determine the broader value of using duckling abundance as a biological indictor of wetland health.

Wetlands tend to be highly variable while commonly differing by region and between individual ponds. These factors make wetlands difficult to define and assess; compounded by the alterations from the agro-industry and rapid evolution of the mechanical and chemical industries this makes studies of wetlands difficult. The overarching aim of my study is to simplify wetland assessments by assessing whether duckling abundance could be used as a biological indicator of wetland health. My comparison of significant variables which predict duckling abundance is meant to extrapolate which factors are the most influential in determining duckling abundance. My findings suggested that multiple factors were affecting duckling abundance in my study sites including conductivity, pH, water permanency, available open water, and aquatic macroinvertebrate abundance. In contrast, broadscale landscape characteristics are weak predictors of dabbling duck duckling abundance. I propose that my comparative analysis demonstrates dabbling duck ducklings may be a sensitive biological indicator of wetland health.

I found dabbling duck duckling abundance differed among study sites – differences were reflective of traditional influential factors (water parameters, habitat, and food availability). Wetlands that exhibited lower conductivity, balanced pH, higher abundance of aquatic invertebrates and ample available open water had higher dabbling duck duckling abundances; meaning, duckling abundance is a potentially useful biological indicator of wetland health. Still, PTI was not a strong significant predictor of duckling abundance which may be due to the lack of resolution in the method or that other factors in the sampling method that confounded this result. These findings emphasize the importance of preserving, restoring, and protecting wetlands and the macroinvertebrate communities of the PPR as a conservation effort for waterfowl and wildlife in general. Future work should attempt to further disentangle the specific drivers of quantity and quality of wetlands and the relationship to duckling abundance.

4.2. Implications of Findings

The PPR is one of the world's most agro-intensive regions due to its flat terrain, nutrient rich soils and low abundance of woody plants, in addition it is estimated that 75 - 99% of grassland has been converted to crop production (Mckenna et al., 2019). Current agricultural practices commonly use large scale monoculture plantings, convert field margins to crop and drain or degrade wetlands which dramatically decrease available habitats for many grassland and wetland species (Bartzen et al., 2010). Intensive agricultural practices also heavily rely on harmful pesticides (insecticides, fungicides and herbicides) to control invertebrate pests, fungal diseases, and undesirable vegetation. Recent studies indicate wide scale agricultural pollution found in many wetlands across the PPR that could have a significant negative impact on the aquatic and terrestrial food webs, namely aquatic macroinvertebrates (Main et al., 2014; Morrissey et al., 2015). While broad characteristics of each wetland are similar, there are unique differences between wetlands (Holland et al., 1990). These variations make some wetlands more capable of residing a high level of biodiversity while others may limit the amount of diversity and biological productivity (Keddy & Fraser, 2003).

Conservation of vast and dynamic ecosystems, like the PPR, requires monitoring approaches that favour low cost, low effort and high efficiency. In intensively cropped areas where the extent of the impact from the alterations to the landscape from agricultural practices is largely unknown, there is a need to identify an effective predictor of wetland health over short and long time scales. Traditionally, researchers have conducted chemical assays and directly measured physical parameters of aquatic environments (e.g., water temperature, conductivity, nutrients, pollutants, etc.). Commonly, many chemical and physical measurements only characterize conditions at the time of sampling, likely making measures difficult to extrapolate the extent of impact. In addition, water analysis tends to be time consuming and monetarily costly. Alternatively, bioindicators represents the biological impact of direct and indirect stressors in the environment and therefore a superior gauge of overall condition of a habitat when supported by analytical evidence.

Historically, aquatic macroinvertebrate communities have proven to be sensitive and useful bioindicators of physiochemical water parameters, contaminants, and overall habitat condition (Lenat, 1988). While there are lengthy and detailed protocols developed to utilize aquatic macroinvertebrates, particularly for monitoring oligotrophic systems (e.g., streams), a logistical challenge has been the identification process (Bonada et al., 2006). Typically, a high level of taxonomic expertise is required; especially, when identify to taxonomic levels below family. Furthermore, reference material is of variable quality or absent for the local region (Jones, 2008). As a result, the identification process is lengthy and can have a high rate of misidentification (Sweeney et al., 2011). Also, it can be difficult to identify some specimens beyond a higher taxonomic level (i.e. order or family) due to deteriorated condition, morphologically immature, or cryptic. Commonly, identification is limited to higher taxonomic levels, such as families, which can be effective at regional or catchment scales, it is likely less sensitive to minute disturbances (Hewlett, 2000). Logistical constraints are further compounded when assessing eutrophic systems, such as wetlands, which due to higher nutrient loads have very high productivity and requires subsampling (as was done in year 2 of my study). Hence, there is a need to determine species of higher trophic level which exhibit an accurate response to wetland disturbance which can be rapidly assessed and cost effective.

Merely retaining wetlands in the PPR may not be enough, the quality of the wetlands may have a substantial impact on conservation efforts. However, factors which determine the quality of wetlands are currently unclear. While widely used bioindicators such as aquatic macroinvertebrates are relatively effective, they exhibit logistical limitations. Moreover, the PPR is a vast region that is also highly dynamic demonstrating the need for a rapid and easily conducted assessment which can be implemented across a large spatial. Here I found that dabbler ducklings exhibited sensitivities to aquatic habitat conditions while abundance was predicted by traditional measure of wetland health. With evidence that ducklings are effective indicators of wetland health, the use of brood surveys could be more widely used with relatively fewer logistical and cost constraints when compared to traditional measures. Furthermore, the gauge of health or rather the ability for the wetland to sustain a high level of biological activity could be better represented by ducklings which are an integrated measure. However, the use of higher taxonomic organisms, like waterfowl, can be misleading as there are numerous ecological and environmental factors which influence species specific and age specific abundance. Accordingly, I acknowledge that the effects of wetland condition on duckling abundance in this study may not be fully generalizable to other regions, time frames or conditions, but represent an important step to employ duckling abundance in wetland monitoring programs in the PPR.

4.3. Limitations and Future Directions

To my knowledge, this is the first assessment of ducklings as a biological indicator of wetland health in the Prairie Pothole Region. Commonly, research on waterfowl broods is conducted with the aim to assess the impact of hypothesized or known variables which influence duckling abundance or survival. Whereas my study was conducted by examining known health conditions of wetlands and if duckling abundance respond to that condition. Due to a lack of true reference sites, I elected a data collection process of a large scope of sampling variables across a gradient of agricultural intensity. This limited extent of sampling which later limited the power of my statistical analysis. In general, wildlife surveys are prone to biases due to the complex interactions (e.g. environmental, behavioral, genetic, etc.) which influence observations. Waterfowl brood surveys tend to exhibit a higher level of observation biases, largely due to the cryptic behavior of hens which varies by species and individuals. Furthermore, it is unclear how environmental factors may influence nesting success. I attempted to address many of these uncertainties by casting a wide set of protocols such as double observations, multiple surveys, collection of a variety of known influential environment variables, etc. However, logistical constraints limited the study scope and my data analysis was relatively coarse, and as a result, interpretation of my results may be considered preliminary. After careful consideration, here, I outline suggestions for future research that aims to assess the effectiveness of ducklings a biological indicator of wetland health in the PPR.

First and arguably, the largest pitfall of this study is a lack in diversity of waterfowl I sampled and the lumping of species into a single abundance metric. It is highly likely that there are varying sensitivities amongst waterfowl guilds, species, and age class. This is a critical knowledge gap of ducklings as a biological indicator, determining which species that are sensitive to wetland condition could provide powerful information to the rapid assessment of wetland health. Furthermore, determination of species which exhibit acute sensitivity to wetland health could add clarity to why some species of waterfowl are thriving while other populations are decreasing. I attempted to test this in my study; however, due to a limit in observations I was unable to separate dabbler and diving ducks; much less, any individual species differences. Thus, my first recommendation for future work is to assess potential guild or species-specific sensitivity to wetland health. I advise, a study could decrease the frequency of surveys to increase the number of study wetlands, this would likely marginally increase observational error but, disproportionately increase the statistical power of the data. My findings suggest that later in the rearing season (e.g., late July-early August) is an optimal time for brood observation (peak counts). There is likely an annual variance of optimal observational periods likely driven by water availability and ambient temperatures.

Second, I assumed that I measured variables relevant to duckling abundance, but other site characteristics (e.g. local predators or weather events) could strongly affect duckling abundance. The effects of nest predation remain unclear despite numerous studies on the topic, largely due to conflicting results between studies, for example the importance of nest density (Ringelman et al., 2012; Gunnarsson et al., 2013), the effect of vegetation (Thompson et al., 2012; Ringelman, 2014), or the overall impact in relation to other variables (Pieron & Rohwer, 2010; Ringelman et al., 2017). The conflicting results of studies on the impact of predation on duckling abundance is likely due to correlated factors like agricultural intensification. Similarly, the effects of weather events (precipitation and ambient air temperature) appears to have an effect on duckling survival rates; however, to what degree is largely unknown (Pietz et al., 2003). A general consensus across several studies is that ambient temperature marginally impact duckling abundance (Pietz et al., 2003). Furthermore, habitat quality is likely a stronger predictor of fitness, which influences ducklings' aptitude to endure variable weather events. Conversely, Stafford and Pearse (2007) found that precipitation events significantly impacted duckling survival of mallards. Likely due to ducklings limited ability to thermoregulate body temperature, relying on the hen to shield against adverse weather conditions which can dramatically impact the duckling's ability to forage and if exposed the duckling may quickly become hypothermic. There are only a few papers on which examine the adverse impacts of precipitation in the PPR while results from all indicate it remains and important predictor of duckling abundance. Agricultural intensification may magnify the effects of adverse weather events by reducing available food resources and lack of vegetation to shelter ducklings. I recommend that future studies address this potential issue by a radio telemetry mark-recapture study which examines the role of weather events and habitat selection of brood-rearing hens.

Third, my study assumes that ducklings were only indirectly impacted by insecticides by reducing available food resources (e.g. aquatic macroinvertebrates). It is probable that indirect impacts are possibly a more significant threat, potentially direct effects impact duckling abundance. There are several recently published studies which have found non-lethal concentrations of insecticides in wild populations of vertebrates and invertebrates (Hagen et al.(2020); Pereira et al., 2020). These suggest that insecticides, namely neonicotinoids, are bioaccumulating in invertebrate and seed eating consumers of the PPR. The impacts of this on waterfowl and avian species in general are largely unknown. Based on a study by Eng et al. (2019) researchers found that song birds that ingested neonicotinoids exhibited behaviors which impacted foraging and decreased body weight. Furthermore, a recently published paper by (Elgin et al., 2020) found ubiquitous detections of neonicotinoids in tree swallows (*Tachycineta bicolor)* that forage on emerged invertebrates of wetlands in the PPR. Based on these recent studies, I find it likely that ducklings may be both directly and indirectly impacted by agricultural insecticides. There is a need to examine the potential adverse impacts of insecticide exposure on precocial waterfowl young and the various sublethal effects it may have.

Fourth, I equally weighted the dietary preference of ducklings across all aquatic macroinvertebrates by summing the abundance of all taxa. This assumes that taxonomic orders of invertebrates respond similarly to changes in water quality. The metric of total aquatic macroinvertebrate abundance can be a generalized indicator of water quality; however, it is likely less accurate when compared to macroinvertebrate community compositions. There is extensive research that has addressed the influence of biotic and abiotic factors on aquatic macroinvertebrate communities; moreover, evidence alludes to macroinvertebrate community composition as a reliable bioindicator. In-turn, many water quality agencies across North America have developed and implemented aquatic macroinvertebrate surveys used as bioindicators of various water bodies, including wetlands. While there is an in-depth understanding of how aquatic macroinvertebrate communities change to various water conditions, there is a limited understanding of the dietary preferences of waterfowl. There are several studies which highlight the importance of aquatic macroinvertebrates as a food resource for waterfowl, especially, precocial young (Krull, 1970; Mcnicol & Wayland, 1992). There are only a few studies which assess the importance of diversity of aquatic macroinvertebrates; furthermore, to the best of my knowledge the research on the dietary preferences of ducks in the PPR is highly limited. This is an overlooked knowledge gap that could provide a critical insight of dietary preferences of waterfowl and establish a connection with the knowledge on aquatic macroinvertebrate communities. Hence, with a knowledge of waterfowl adult and duckling dietary preferences could further illuminate the accuracy of using waterfowl as biological indicators of wetland health.

Fifth, I attempted to have variation in the size and location of wetlands in my study; however, due to environmental constraints (e.g., drought conditions) I focused on only class 4 and 5 wetlands which are the more permanent classes. Furthermore, I limited my study to wetlands that could be viewed in entirety from one stationary position on the roadside, to reduce observational error and disturbance during the duckling surveys. These constraints likely limit how generalizable my findings are to other wetland types found in the PPR (e.g., saline wetlands, large wetlands, lower classed wetlands, etc.). Moreover, although my study spanned 6 transects in a large area within central Saskatchewan, this is only a small portion of the PPR. The conditions which were present in my study may not be applicable in other areas of the PPR. Future studies should include more diversity in wetland type and examine potential variation in location within the PPR by expanding the study area.

Sixth, I assumed that ducklings were isolated to a single wetland, when it is likely females attending ducklings moved to more favorable conditions (e.g. wetlands with relatively higher water permanency). Dzus and Clark (1997) determined that within the first two weeks after hatching, mallard broods were found at up to 5 different wetlands, with an average distance of 211 m for successfully reared young, while broods which exhibited total duckling mortality averaged 311 m. These findings suggest along with supporting studies that external stressors (e.g., agricultural intensity, water availability, etc.) may influence overland travel of waterfowl broods and that merely observing young at a wetland may not indicate the brood’s dependency on that wetland. Rather, several surrounding wetlands may be supporting rearing efforts and sampling of wetland clusters may have been more appropriate to detect effects.

Finally, my findings suggest that ducklings may be a biological indicator of wetland health; however, the repeatability of my findings may be limited based on evidence that duckling surveys tend to have a higher degree of observational error in comparison to other wildlife surveys and even adult waterfowl surveys (Giudice, 2001; Pagano & Arnold, 2009). This is largely due to the cryptic behavior waterfowl hens (Lyons et al., 2020). Pagano and Arnold (2009) found that 67.5% of present broods were missed during a ground survey in wetlands where duckling estimates were derived form closed-population mark-recapture techniques. Furthermore, Pagano and Arnold (2009) examined detection probability of ground-based waterfowl surveys and found that the experience level of observers highly influenced detection rate. They found that an experienced waterfowl observer had a detection probability of 0.911 (range = 0.866-0.944) while a novice observer detection probability was 0.790 (range = 0.537-0.890). A relatively new waterfowl survey method may be able to address these limitations; results from some studies suggest higher detection rates compared to more traditional waterfowl survey methods from the use of an inferred and visible light cameras attached to an unmanned aerial vehicle (UAV), commonly referred to as a "drone" (Pöysä et al., 2018; Bushaw et al., 2021).

To summarize, in waterfowl and wetland research, appropriate survey method and data collection protocols are a common debate. This is largely due to knowledge gaps or uncertainties that derive from influential factors which are highly correlated. As a result, a single study is likely not effective in determining the effects of complex interactions. Accordingly, I acknowledge that the effects of wetland characteristics on duckling abundance observed in this study may not be fully generalizable to other spatial temporal scales, conditions, or locations, but represent an important first step for identifying higher trophic organisms such as waterfowl as candidate integrative biological indicators of wetland health in the PPR. My results suggest that not only habitat quantity is important but, rather quality may be a significant influential factor of waterfowl populations. With further supporting research, it appears ducklings may be a powerful tool to measure wetland health; detecting multiple cumulative factors that are likely otherwise imperceptible. Since these factors collectively impact ducklings and likely many other organisms which rely on wetlands of the PPR, estimates of waterfowl brood productivity shows good promise. It is probable that human impact on the PPR will increase including landscape simplification, agro-chemical usage, and global warming. Hence, it is critical to not only preserve wetlands but, to establish and implement standardized assessments of wetland health which can be used to better focus and enhance conservation efforts. I find it likely waterfowl populations could aid in this and further research should aim to assess the accuracy of duckling abundance or other more species-specific measures as a biological indicator of wetland health in the PPR.

Tables

Table 1.Summary of habitat variables collected at each study wetland (n=58) in the 6 survey transect sites. The mean and standard deviation of habitat variables including cropland percentage, grassland percentage, roadway disturbance, percentage of adjacent wetland, percentage of wooded area, percentage of algae cover, percentage of aquatic vegetation cover, percentage of bare wetland bottom, maximum water depth, wetland basin fill, distance of wetland vegetation degradation. The area of sampling is the prairie region of Saskatchewan during August of 2018 and 2019.

Transect	**Ibstone**	**Peterson**	**Tichfield**	**Buchanan**	**Grayson**	**Zealandia**
Year	2018	2018	2018	2019	2019	2019
Average wetland class	4.7	4.86	4.89	5	5	5
Dominant surrounding crop type	Canola	Canola	Cereal	Cereal	Cereal	Cereal
% Cropland	82.5 ± 22.39	90.71 ± 12.05	60.56 ± 15.7	64.58 ± 38.64	45.91 ± 26.91	58.89 ± 34.35
% Grassland	0 ± 0	0 ± 0	12.78 ± 20.17	15.42 ± 36.15	11.36 ± 25.89	21.88 ± 41.05
% Roadway	15 ± 22.85	7.14 ± 12.2	26.67 ± 17.32	14.17 ± 13.46	28.64 ± 21.11	17.22 ± 11.21
% Adjacent wetland	2.5 ± 7.91	2.14 ± 5.67	0 ± 0	0 ± 0	14.09 ± 20.1	4.44 ± 7.26
% Wooded	0 ± 0	0 ± 0	0 ± 0	5.83 ± 13.79	0 ± 0	0 ± 0
% Algae cover	8 ± 13.98	5.71 ± 15.12	3.33 ± 10	25.42 ± 28.24	11.36 ± 17.33	15.56 ± 22.42
% Aquatic vegetation cover	36 ± 35.34	33.57 ± 34.97	16.67 ± 26.46	24.58 ± 29.03	17.27 ± 23.28	8.33 ± 23.18
% Bare sediment	36 ± 39.43	15.71 ± 15.92	45 ± 43.59	9.58 ± 14.84	29.09 ± 30.89	22.78 ± 34.2
Max water depth (m)	0.44 ± 0.38	0.4 ± 0.39	0.22 ± 0.24	0.82 ± 0.53	0.69 ± 0.32	0.37 ± 0.37
Basin fill score (0-5)	3 ± 1.33	2.43 ± 1.81	1.89 ± 1.54	3.92 ± 0.29	3.36 ± 0.5	2.78 ± 1.56
Distance of degraded wetland vegetation (m)	40 ± 28.77	42.73 ± 8.51	9.1 ± 18.28	23.27 ± 21.49	28.76 ± 28.31	39.93 ± 62.11

Table 2. A summary of dabbler duckling brood counts collected in the June - August 2018 and 2019 in the prairie region of Saskatchewan. Values represent a mean (±SD) of counts from four survey periods at each transect site (n=6). Included are mean and max counts of ducklings by individual species and all species combined (mallard, gadwall, green-winged teal, blue-winged teal, northern shoveler, northern pintail, redhead, canvasback, scaup, ring-necked duck, bufflehead, and ruddy duck).

	Transect					
	Ibstone	Peterson	Tichfield	Buchanan	Grayson	Zealandia
Year	2018	2018	2018	2019	2019	2019
n Wetlands surveyed	10	7	9	12	11	9
n Species detected	6	3	6	10	5	5
Mean count of ducklings (all species)	0.24±0.85	0.13±0.57	0.32±1.13	0.21±0.62	0.13±0.64	0.12±0.56
Max number of ducklings	5.75	3.25	6.75	4	6.5	4
Mallard	0.23±0.53	0±0	1.28±1.92	0.65±1.16	0.8±1.87	0.81±1.52
Gadwall	0.45±0.93	0±0	0.42±1.18	0.06±0.21	0.16±0.5	0.25±0.5
American wigeon	0±0	0±0	0.08±0.24	0±0	0.09±0.29	0±0
Green-winged teal	0±0	0±0	0±0	0.19±0.62	0±0	0±0
Blue-winged teal	0.93±1.31	0.75±1.23	1.89±2.69	0.9±1.02	0.55±0.91	0.28±0.79
Northern shoveler	0.45±0.7	0.14±0.35	0.33±0.78	0±0	0.25±0.59	0.19±0.55
Northern pintail	0±0	0±0	0.44±0.84	0±0	0±0	0.17±0.47
Redhead	0±0	0±0	0±0	0.13±0.41	0±0	0±0
Canvasback	0±0	0±0	0±0	0.15±0.36	0±0	0±0
Scaup	0.7±1.55	0.86±1.36	0±0	0.33±0.65	0±0	0±0
Ring-necked duck	0±0	0±0	0±0	0.13±0.41	0±0	0±0
Bufflehead	0±0	0±0	0±0	0.29±0.65	0±0	0±0
Ruddy duck	0.58±1.73	0±0	0±0	0.17±0.55	0±0	0±0

Table 3. Summary of water quality parameters collected at each wetland (n=58) across the 6 transect sites. Water parameters include mean ± SD values by transect for conductivity and pH, the number of insecticides detected, mean concentration of insecticides, and Acute Pesticide Toxicity Index (PTI) calculated for each wetland in Saskatchewan during June or July of 2018 and 2019.

Transect	**Ibstone**	**Peterson**	**Tichfield**	**Buchanan**	**Grayson**	**Zealandia**
Year	2018	2018	2018	2019	2019	2019
n Wetland surveyed	10	7	9	12	11	9
Conductivity (µS/cm)	1257.23± 1068.69	1811.14± 359.02	2683.11± 1761.89	1391.58± 687.31	3062± 892.05	2767± 2109.13
pH	9±0.85	9.3±0.76	8.45±0.59	8.54±0.32	8.73±0.35	9.18±0.34
Total number of Insecticides Detected	16	6	2	10	1	6
Insecticide Total Conc. (ng/L)	328.3±232.1	3.6 ± 2.2	0.7 ± 0.5	2.9 ± 1.4	0 ± 0	3.6 ± 2.6
PTI (ng/L)	213.1±152.8	3.6 ± 2.2	0.1 ± 0.1	0.6 ± 0.3	0 ± 0	1 ± 0.6

Table 4. Akaike's information criterion corrected for small sample sizes (AIC_c) and model weight (W_i) for each candidate models to explain variation of duckling abundance in relation to pH, conductivity, and Pesticide Toxicity Index (PTI). Water parameter and duckling data were collected during June or July 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan. Also shown are number of model parameters (k) and log-likelihood value ($\log(L_i)$).

	Independent Variable	**k**	**Log(L_i)**	**AIC_c**	**ΔAIC_c**	**W_i**
Dabbler Abundance	log (Conductivity) + pH	5	-119.46	250.07	0.00	0.6737
	acute PTI+ pH+ log(Conductivity)	6	-119.18	252.00	1.93	0.2565
	pH	4	-122.92	254.60	4.53	0.0699
	NULL	3	-133.64	273.73	23.66	0.0000

Table 5. Model averaged parameter estimates and 95% confidence intervals (CI) explaining variation of dabbler duckling abundance. Parameters included pH, conductivity and acute Pesticide Toxicity Index (PTI). Water quality and duckling count data were collectedin June or July of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan. Parameters with significant effects are indicated in bold font.

	Parameter estimates (CI)
(Intercept)	6.21 (1.48, 11.12)
Acute Pesticide Toxicity Index	-3.85 (-8.71, 0.54)
log(Conductivity)	**-0.35 (-0.83, -0.05)**
pH	-0.26 (-0.75, 0.01)

Table 6. Akaike's information criterion corrected for small sample sizes (AIC_c) and model weight (W_i) for each candidate models to explain variation of duckling abundance in relation to maximum water depth, aquatic vegetation density, grassland percentage, and distance of wetland vegetation degradation. Wetland characteristic variables and duckling data were collected during August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan. Also shown are number of model parameters (k) and log-likelihood value ($\log(L_i)$).

	Independent Variable	**k**	**log(L_i)**	**AICc**	**ΔAIC**	***W_i***
Dabbler Abundance	Max. Water Depth + Aquatic Veg. Density	5	-126.28	263.71	0.00	0.42
	Max. Water Depth + Aquatic Veg. Density + Grassland	6	-125.10	263.84	0.13	0.39
	Max. Water Depth + Aquatic Veg. Density + Grassland+ Wetland Veg. Degradation	7	-124.75	265.74	2.03	0.15
	Max. Water Depth	4	-129.83	268.41	4.70	0.04
	NULL	3	-133.64	273.73	10.02	0.00

Table 7. Model averaged parameter estimates and 95% confidence intervals (CIs) explaining variation of dabbler duckling abundance. Three variables were important in the top model: floating aquatic vegetation density, maximum water depth and percentage of grassland. Water parameters and duckling count data were collected during the August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan. Parameters with significant effects are indicated in bold font.

	Parameter Estimates (CI)
(Intercept)	0.635 (-0.001, 1.272)
Floating Aquatic Veg. Density	**-0.016 (-0.026, -0.006)**
Max. Water Depth	**0.703 (0.391, 1.015)**
Grassland	-0.001 (-0.012, 0.004)

Table 8. Summary, including mean ± SD, of aquatic macroinvertebrate samples collected at each study wetland (n=58) across the 6 transect sites in Saskatchewan. Macroinvertebrate samples were collected once during the August 2018 and 2019.

Transect	**Ibstone**	**Peterson**	**Tichfield**	**Buchanan**	**Grayson**	**Zealandia**
Year	2018	2018	2018	2019	2019	2019
n wetlands surveyed	10	9	9	12	11	9
n Aq. Invert. Orders	6	3	6	10	6	6
Max Total Aq. Invert.	11540	1330	20580	5640	9820	8920
Max Biomass of Aq. Invert. (g)	4.488	2.554	30.871	0.941	1.261	2.748
Mean Total Aq. Invert.	308.04 ± 1331.89	63.22 ± 163.79	428.56 ± 2119.94	392.06 ± 1067.73	464.6 ± 1333.13	377.89 ± 1113.2
Biomass of Aq. Invert.	1.15 ± 1.23	0.94 ± 0.86	4.84 ± 9.25	0.26 ± 0.25	0.34 ± 0.34	0.54 ± 0.8
Diptera	2467.5 ± 3705.04	139.6 ± 163.77	1645.11 ± 719.69	1145.67 ± 1230.72	845.36 ± 1123.58	557.22 ± 567.39
Gastropoda	0 ± 0	0 ± 0	0 ± 0	2407.42 ± 2199.51	3608.45 ± 2593.15	1856.44 ± 2686.32
Hemiptera	525.9 ± 537.33	304.3 ± 359.52	362.33 ± 342.6	552.25 ± 658.49	283.45 ± 223.5	670.89 ± 512.73
Amphipoda	76.4 ± 227.87	0.3 ± 0.64	2614.78 ± 6413.76	35.83 ± 116.45	38.18 ± 89.63	848.89 ± 1558.44
Ephemeroptera	0.1 ± 0.3	0 ± 0	5 ± 14.14	19.75 ± 38.93	1.82 ± 5.75	16.78 ± 19.97
Odonata	63 ± 81.12	158.4 ± 188.17	1.89 ± 4.12	56.17 ± 80.78	60.18 ± 112.91	15.56 ± 20.92
Tricoptera	4.2 ± 6.76	7.2 ± 13.43	41.89 ± 64.45	3.58 ± 5.92	41.73 ± 84.55	4.44 ± 12.57
Coleoptera	40.3 ± 58.91	7.8 ± 7.45	34.67 ± 43.98	24.17 ± 31.47	61.36 ± 51.78	18.89 ± 14.74
Plecoptera	201 ± 248.26	73.1 ± 67.92	7.89 ± 10.82	34.33 ± 31.43	118 ± 95.06	104.67 ± 119.3
Hydracarina	9.5 ± 12.38	4.7 ± 7.66	0.56 ± 1.57	32.42 ± 28.32	52 ± 66.57	63 ± 116.08
Megaloptera	0.5 ± 1.5	0 ± 0	0 ± 0	1.08 ± 3.59	0 ± 0	0 ± 0
Zooplankton	N/A	N/A	N/A	68.83 ± 45.21	46 ± 92.02	446.22 ± 656.91

Table 9. Akaike's information criterion corrected for small sample sizes and model weights for the top candidate model and null model explaining variation in dabbler duckling abundance. The only parameter retained was aquatic macroinvertebrate abundance (log). Aquatic macroinvertebrates and duckling data were collected during August 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan.

Independent Variable	**k**	**log(L_i)**	**AICc**	**ΔAIC**	W_i
Log (Aq. Invert. Abundance)	4	-131.398	271.5502	0	0.748
NULL	3	-133.643	273.7295	2.179283	0.252

Table 10. Akaike's information criterion corrected for small sample sizes (AICc) and model weight (Wi) for each candidate models to explain variation of duckling abundance in relation to aquatic macroinvertebrate abundance, conductivity, maximum water depth and aquatic vegetation density. Wetland and duckling data were collected during June - August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan. Also shown are number of model parameters (k) and log-likelihood value (log(Li)).

	k	**logLik**	**AICc**	**delta**	**weight**
log(Conductivity) + Aquatic Veg. Density	5	-121.76	254.68	0	0.71
log(Conductivity)+Aq. Invert. Abundance + Aquatic Veg. Density	6	-121.72	257.08	2.40	0.21
log(Conductivity)+Aq. Invert. Abundance + Max. Water Depth +Aquatic Veg. Density	7	-121.70	259.63	4.96	0.06
log(Conductivity)	4	-126.94	262.63	7.95	0.01
NULL	3	-133.64	273.73	19.05	<0.0001

Table 11. Parameter estimates from a zero-inflated Poisson model of dabbling duck duckling abundance in relation to conductivity, and aquatic vegetation density. Wetland and duckling data were collected during the June - August 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan.

	Estimate	**Std. Error**	**z value**	**Pr(>|z|)**
(Intercept)	-2.40	1.70	-1.41	0.16
log(Conductivity)	-0.83	0.21	-3.88	**<0.0001**
Aquatic Veg. Density	-0.02	0.01	-2.86	**<0.0001**

Figures

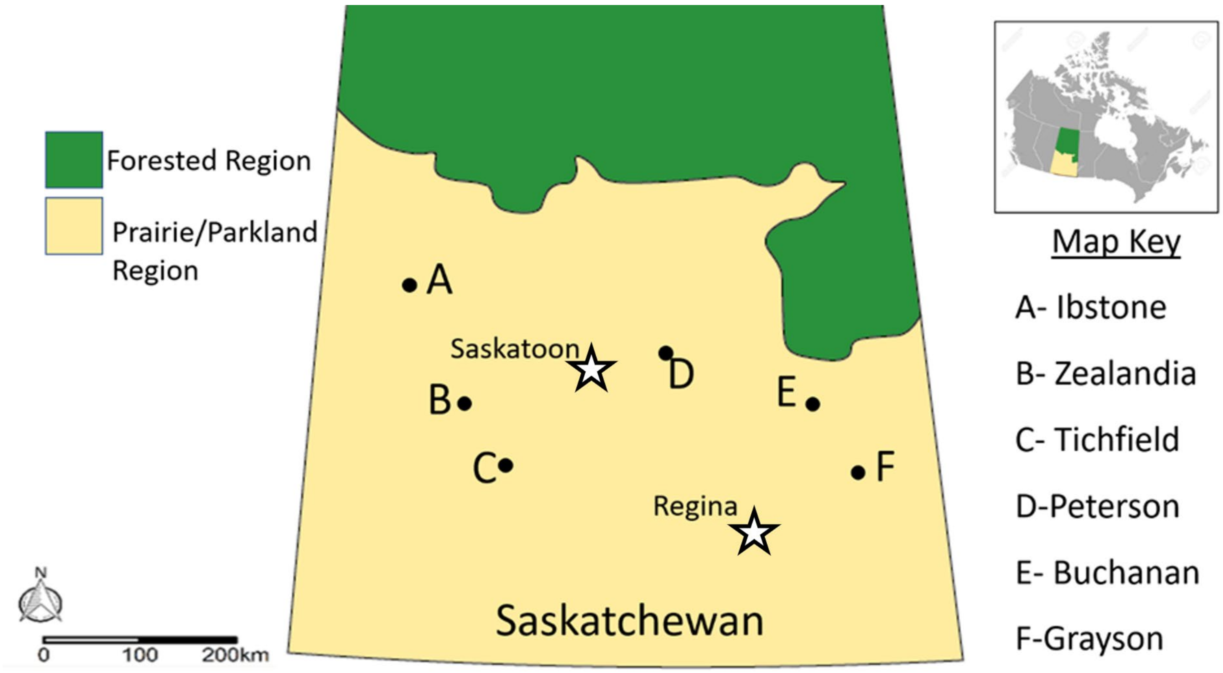

Figure 1. Map showing the 6 sampling sites in southern Saskatchewan where collection of duckling brood surveys, wetland water quality, aquatic macroinvertebrates, and wetland and landscape habitat features occurred in 2018 and 2019.

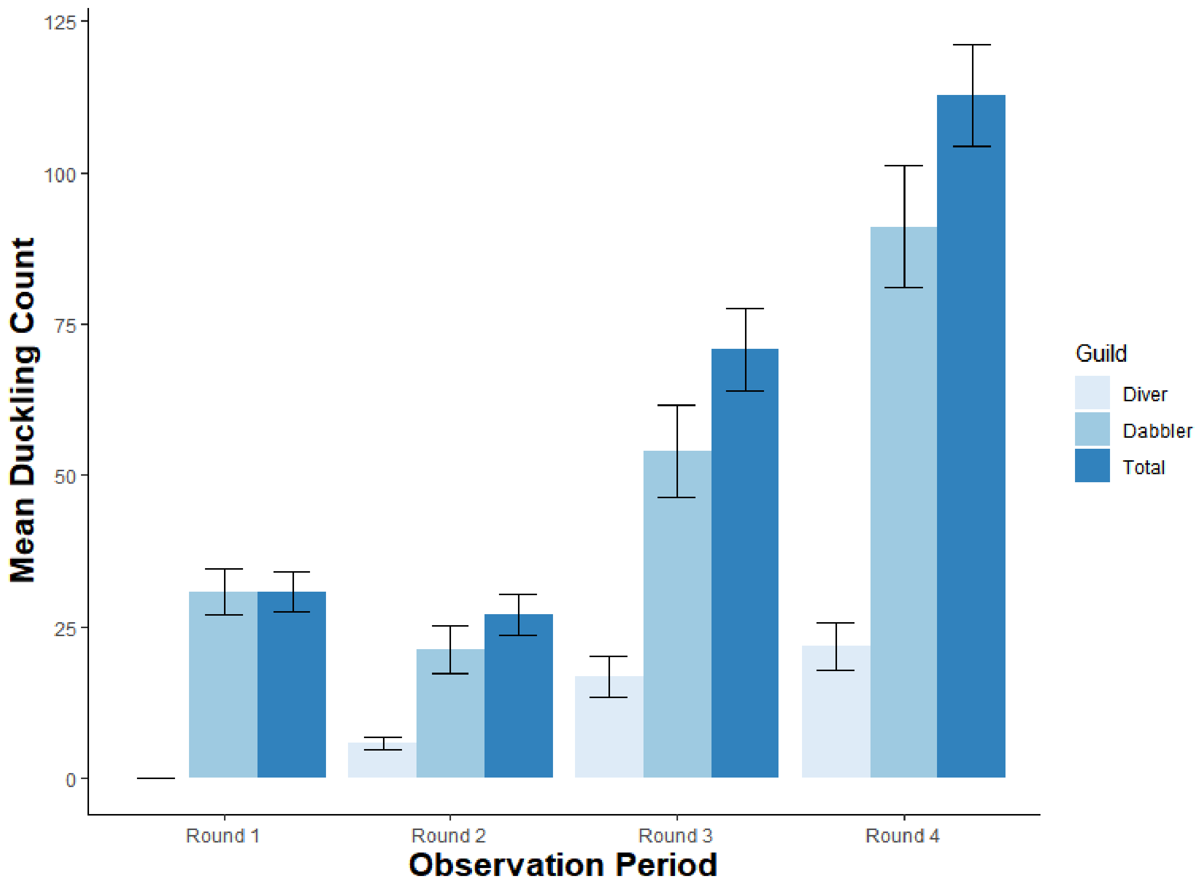

Figure 2. Mean dabbling duckling brood counts across four observational periods during surveys of wetlands (n=58) in the June - August 2018 and 2019 in Saskatchewan. Ducklings have been separated by guild (diver & dabbler) and total abundance.

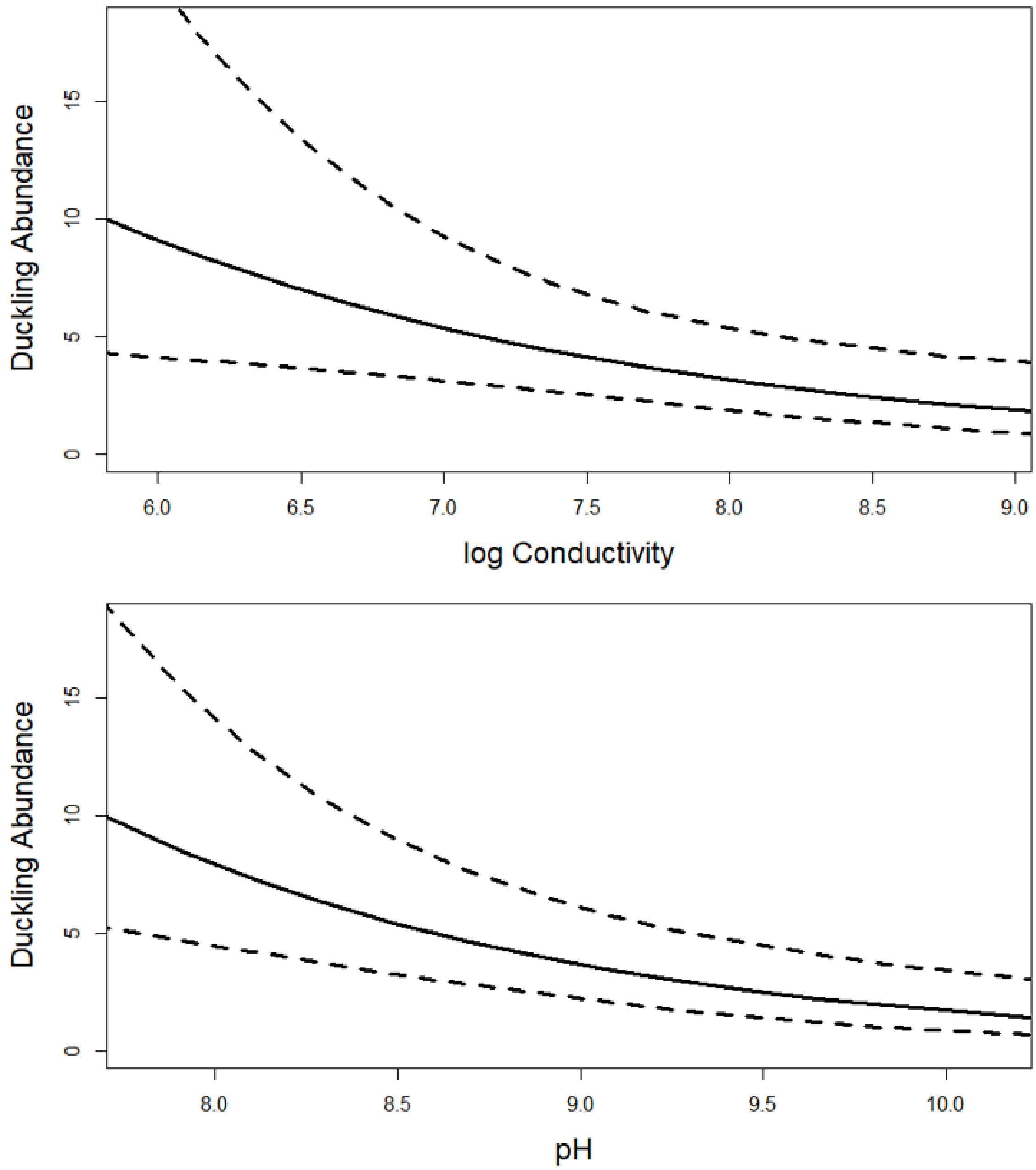

Figure 3. Predictions of zero-inflated Poisson model for dabbling duck duckling abundance in relation to pH and conductivity. The model-predicted duckling abundance (solid line) is shown with the 95% confidence intervals (dashed lines). Water chemistry and duckling data were collected during the June - August 2018 and 2019 at wetlands (n=58) found in the prairie region of Saskatchewan.

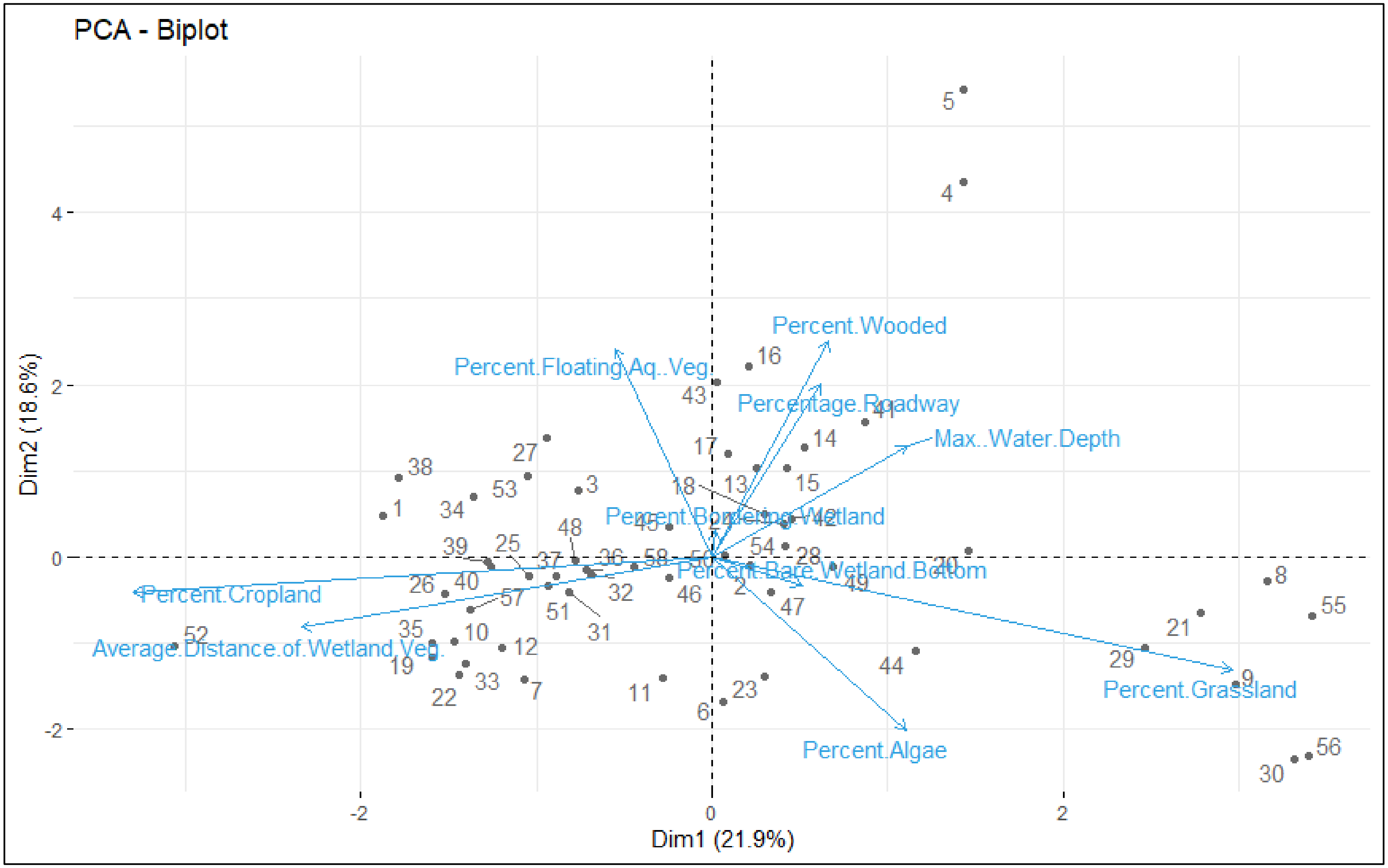

Figure 4. PCA biplot of wetland and surrounding habitat variables (floating veg. encroachment, density of wooded plants, roadway displacement, max. water depth, percentage of bare bottom, cropland percentage, distance of wetland vegetation disturbance, percentage of algae, percentage of grassland) highlighted in blue. Habitat data were collected during June - August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan.

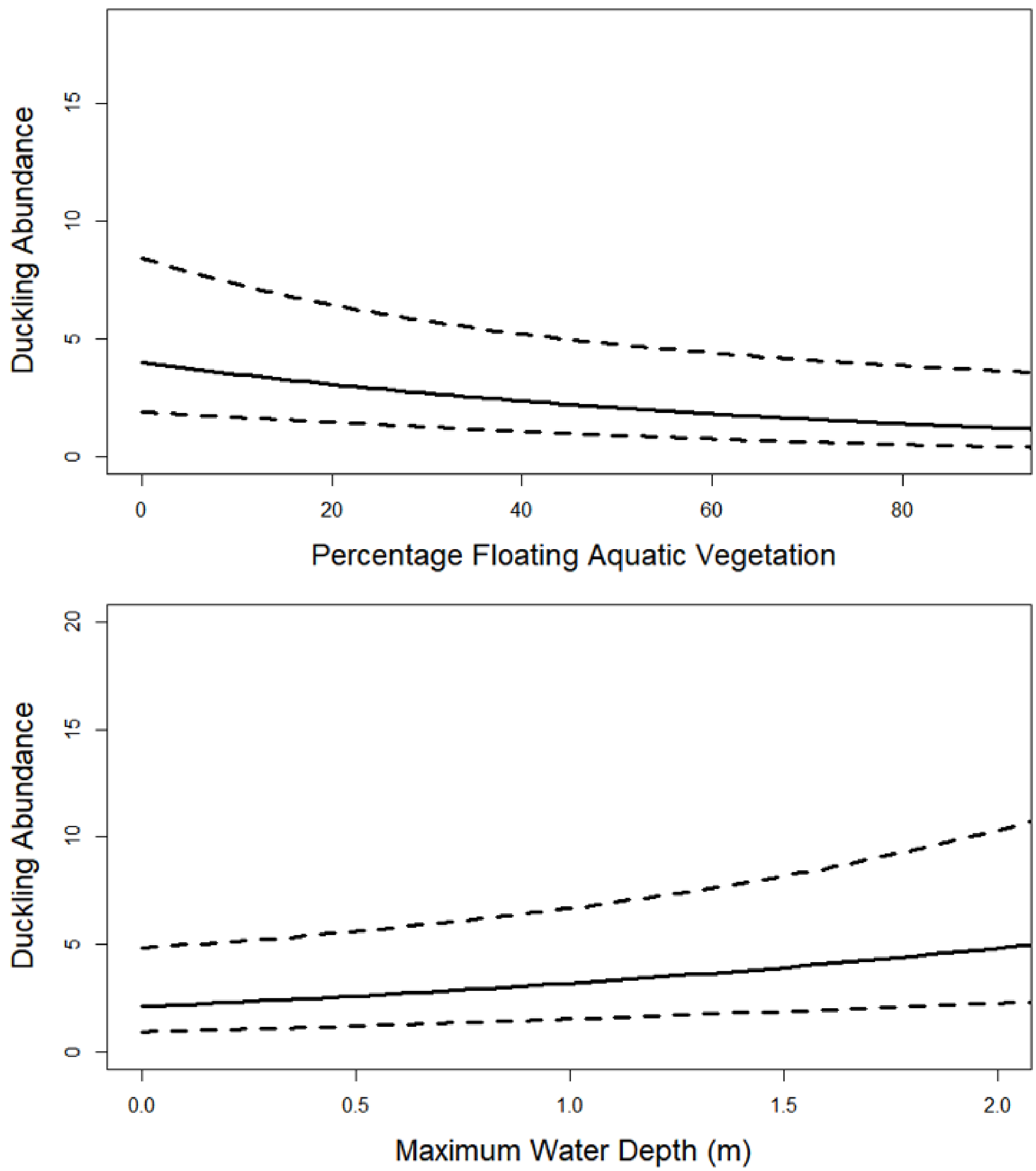

Figure 5. Predictions of zero-inflated Poisson model on dabbler duckling abundance in relation to wetland habitat variables (maximum water depth, aquatic vegetation density, and surrounding grassland). The predicted model is symbolized as a solid line while the 95% confidence intervals are dashed lines. Habitat variables and duckling data were collected during June - August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan.

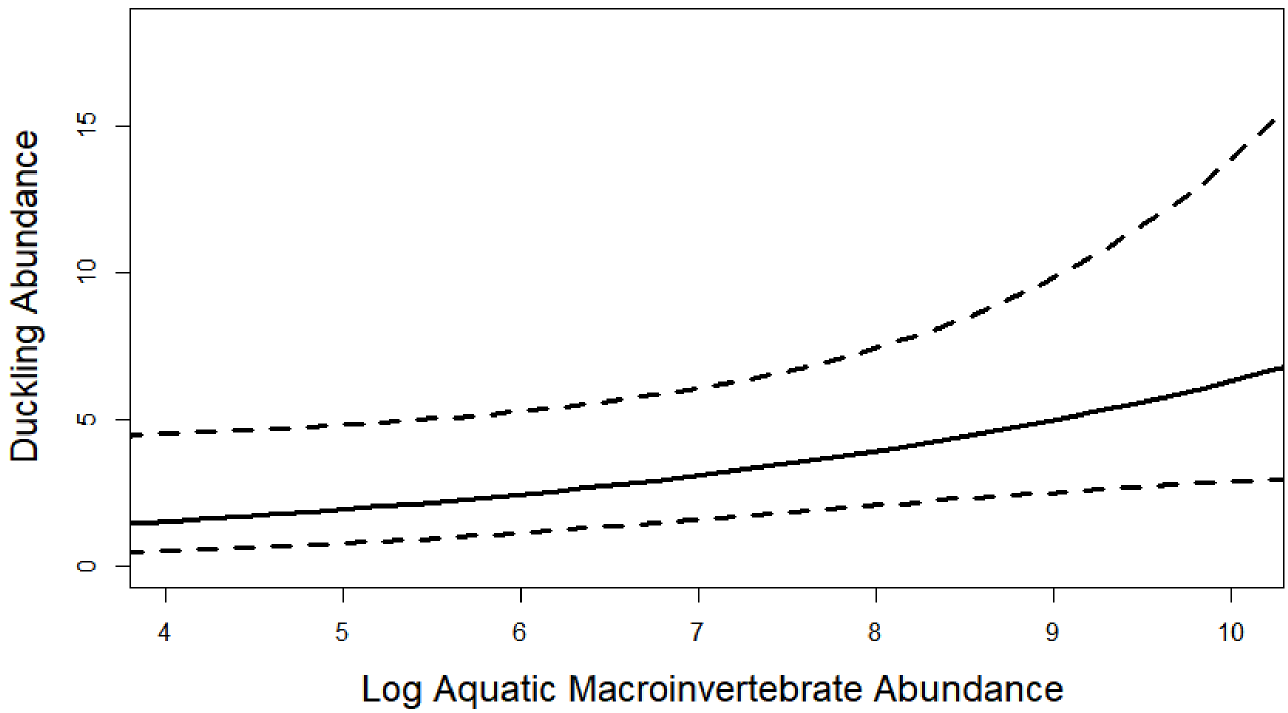

Figure 6. Predictions of zero-inflated Poisson model on dabbler duckling abundance in relation to aquatic macroinvertebrate abundance. The predicted model is symbolized as a solid line while the 95% confidence intervals are dashed lines. aquatic macroinvertebrates and duckling data was collected during June - August of 2018 and 2019 at wetlands (n=58) in the prairie region of Saskatchewan.

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