

EVALUATING THE RELATIONSHIP BETWEEN LOCAL FOOD AVAILABILITY AND
WETLAND LANDSCAPE STRUCTURE IN DETERMINING DABBLING DUCK HABITAT
USE DURING SPRING MIGRATION

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Masters of Science

by
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The undersigned, appointed by the dean of the Graduate School, have examined the thesis entitled

EVALUATING THE RELATIONSHIP BETWEEN LOCAL FOOD AVAILABILITY AND
WETLAND LANDSCAPE STRUCTURE IN DETERMINING DABBLING DUCK HABITAT
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THESIS ABSTRACT

Wetlands in the Nebraska's Rainwater Basin (RWB) have decreased by 90% over the past two centuries and are subject to on-going degradation of quality from urban and agricultural land-use practices. Losses in wetland habitat quantity and quality are important because the RWB serves as a critical spring staging area to ~7 million dabbling ducks, including approximately 50% of North America's mid-continent mallard (*Anas platyrhynchos*) population, and 30% of North America's total Northern pintail (*A. acuta*) population. During spring, waterfowl depend on wetland habitat for aquatic invertebrates and plant materials to accumulate the energy and protein needed to complete migration and initiate egg production. If demands for quality food resources are not met, waterfowl may arrive at breeding grounds in poorer body condition, and consequently be less likely to achieve reproductive success. This cross-seasonal effect is believed to be driven by excessive habitat loss at mid-latitudes, introduction of invasive plant species, and depletion of food resources by fall migrants. Given the importance of food resource acquisition at mid-latitude stopover sites and subsequent effects on recruitment, the goal of this study was to improve understanding of food resource availability in wetlands and the relationship to habitat use by spring-migrating waterfowl.

I conducted weekly waterfowl surveys and quantified local habitat characteristics including seed density (kg/ha), invertebrate density (kg/ha), energy derived from food resources (kcal/ha), water depth, wetland area, vegetative cover, and several water quality parameters at 32 wetlands in spring 2014 and 35 wetlands in spring 2015. Additionally, I quantified wetland habitat surrounding each study site by assessing wetland area and number of wetlands (>1ha) within 2.5km and 5km of a study site. Study sites were located on public lands managed by the Nebraska Game and Parks Commission and the U. S. Fish and Wildlife Service, private

conservation easement lands enrolled in the Wetlands Reserve Program (WRP), and on private lands managed for agriculture (cropped and non-cropped).

A set of species distribution models were developed to explain spring dabbling duck density and species richness in the RWB. I hypothesized that a combination of local (food density, energy, water depth, wetland area, and vegetative cover) and landscape variables would explain the greatest amount of variability in dabbling duck density. In 2014 (a dry year), energy, seed density, water depth, wetland area, and wetland density in the surrounding landscape were positively associated with dabbling duck density; however, invertebrate density and vegetative cover had no influence on dabbling duck density. In 2015 (wet year), seed density and energy were positively associated with dabbling duck density; however, water depth, wetland area, vegetative cover, invertebrate density, and wetland area in the surrounding landscape had no influence on dabbling duck density. Wetland area and water depth were the only useful explanatory variables for explaining species richness in 2014, whereas in 2015 dabbling duck species richness was best explained by wetland area and vegetative cover.

I used non-parametric analyses to compare seed density, and true metabolizable energy (TME) at three wetland types; public, WRP, and cropped wetlands. Seed density did not vary among wetland types in 2014 or 2015. Median seed density estimates during both years at public, WRP, and cropped wetlands were 593kg/ha ($\bar{x} = 621\text{kg/ha}$), 561kg/ha ($\bar{x} = 566\text{kg/ha}$), and 419kg/ha ($\bar{x} = 608\text{kg/ha}$) respectively. Seed density was consistent between years for public and WRP wetlands, but varied between years for cropped units ($p < 0.05$). Variation in seed density between years at cropped wetlands was likely influenced by the presence/absence of agricultural waste grains. Cumulative TME varied among wetland type in 2014 and 2015, with greater TME at cropped wetlands (median = 2431kcal/kg) than public (median = 1740kcal/kg)

and WRP wetlands (median = 1781kcal/kg), however TME did not differ between WRP and public wetlands. TME was consistent among wetland types between 2014 and 2015. Seed density estimates from this study were statistically greater than estimates currently used for management planning in the RWB, however, TME estimates were statistically less than estimates currently assumed for WRP and public wetlands in the region. My estimates for mean aquatic invertebrate density were approximately 40-fold less than estimates for mean seed density. Benthic communities accounted for 68% of the total invertebrate density, however invertebrate diversity was greater in nektonic communities.

Neonicotinoid synthetic insecticides are believed to have a deleterious effect on aquatic invertebrate communities in agricultural areas, although their occurrence in RWB wetlands were previously unknown. I detected trace levels of neonicotinoids in 92% of water samples collected in wetlands sampled in the RWB during the spring of 2015. I predicted a relatively high detection rate given the intensity of row crop production in the region, though concentrations were lower than expected. Concentrations at 26 wetlands sampled fell below toxicity benchmarks proposed by the Canadian Environmental Quality Guidelines, and only 11% of wetlands sampled had concentrations exceeding the most conservative benchmark proposed by the Environmental Protection Agency. Neonicotinoids concentrations were minimal at wetlands with vegetative buffers strips $\geq 50\text{m}$ between a wetland and a cropped field, relative to wetlands with vegetative buffers strips $< 50\text{m}$. Although neonicotinoid levels were below lethal concentrations for all aquatic invertebrates identified in this study, I observed a negative association between neonicotinoid concentrations and aquatic invertebrate density (g/m^2).

**CHAPTER I: EVALUATING THE RELATIONSHIP BETWEEN LOCAL FOOD
AVAILABILITY AND WETLAND LANDSCAPE STRUCTURE IN DETERMINING
DABBING DUCK HABITAT USE DURING SPRING MIGRATION**

With fewer than 50% of historic wetlands remaining in mid-latitude states, determining how dabbling ducks distribute themselves in relation to food availability and wetland habitats during spring migration is essential to strengthen the ecological approach to conservation planning (Dahl 1990; Straub et al. 2012). During spring, wetlands at mid-latitude migration sites provide essential resting and foraging opportunities for dabbling ducks as they migrate from southern wintering areas to higher latitudinal breeding grounds (LaGrange and Dinsmore 1989; LaGrange 2005; Arzel et al. 2006; Bishop and Vrtiska 2008). Spring migrants rely on plant materials and aquatic invertebrates as food sources at stopover wetlands to accumulate energy and protein reserves needed to complete migration and initiate egg production (LaGrange and Dinsmore 1989; Tidwell et al. 2013). Moreover, numerous studies have hypothesized wetland food availability as a limited resource during spring, as early migrants await the thaw of wetland habitats and likely encounter diminished seed banks depleted by preceding autumn migrants (Stafford et al. 2006; Greer et al. 2009; Straub et al. 2012; Stafford et al. 2014). In some instances, wetland food resources may fall below a threshold at which time the energetic cost of foraging for food items is greater than the energetic benefits acquired from those foods (Reinecke et al. 1989; Hagy and Kaminski 2012b). When foraging conditions become unfavorable, ducks must decide to remain at a site or undergo the energetic expense of pursuing habitats with better foraging conditions (Pearse et al. 2012; Williams et al. 2014; Hagy and Kaminski 2015).

Inundated wetland density and distribution across the landscape likely influence dabbling duck distribution and foraging strategies at migration stopover sites (Webb et al. 2010, Tidwell et al. 2013). In addition to seasonal variation in seed availability (Stafford et al. 2011; Straub et al. 2012), invertebrate communities at temperate latitudes can be unpredictable and sporadically distributed (Murkin and Kadlec 1986a; Stafford et al. 2016), and thus dabbling ducks likely supplement their diets and modify their foraging strategies by using multiple wetlands within a landscape (Taft and Haig 2006; Webb et al. 2010). Although ducks are highly mobile and capable of making decisions beyond the local scale, it may be advantageous to identify and select wetlands in areas with high wetland density, where the energetic costs of traveling among foraging sites can be minimized based on proximity of nearby wetlands (Brown and Dinsmore 1986; Fredrickson and Heitmeyer 1988; Pearse et al. 2012; Beatty et al. 2014). However, wetland distribution across a landscape is rarely spatially or temporally consistent, and the initial decision to select an isolated wetland or wetland within a complex may be driven by chance of discovery (Taft and Haig 2006). However, once a bird has committed to a wetland, local conditions and food availability likely influence the duration of occupancy.

Seasonal diet shifts observed in dabbling ducks are thought to be driven by nutritional requirements associated with annual life history events, such as molt, migration, and egg production (Ballard et al. 2004; Baldassarre and Bolen 2006; Anteau and Afton 2008; Williams et al. 2014). Optimal foraging theory predicts that during migration, dabbling ducks should select food resources that provide the greatest energetic benefit to sustain the energetic demand of prolonged flight and thermoregulation (LaGrange and Dinsmore 1989; Ballard et al. 2004; Arzel et al. 2006; Anteau and Afton 2008; Williams et al. 2014). Although adults of many dabbling duck species are classified as foraging generalists with regard to wetland food items,

the energetic quality or true metabolizable energy (TME; gross energy of feed consumed minus the gross energy of excreta) can vary considerably among food items (Reinecke et al. 1989; Strader and Stinson 2005). For example, invertebrates are an important source of calcium and protein for dabbling ducks, but estimated to contain 5-8 times fewer calories than moist-soil seeds commonly consumed by dabbling ducks (Krapu and Swanson 1975; Sherfy 1999; Baldassarre and Bolen 2006). Moreover, TME among moist-soil seeds can range from < 1.00 kcal/g in *Scirpus spp* and *Sparganium spp* to >2.50 kcal/g in *Echinochloa spp*, and >3.0 kcal/g in agricultural waste grains (Reinecke et al. 1989; Sherfy 1999; Dugger et al. 2007; Straub 2008). Although food density estimates have been used to explain variation in dabbling duck habitat use (Murkin and Kadlec 1986a; Greer et al. 2006; Hagy et al. 2014), additional resolution may be gained by analyzing the energetic quality of available food resources, particularly during migration (Beatty et al. 2015).

Spring migration is an annual life history requirement for most dabbling ducks; however, factors influencing habitat use and selection at local and landscape levels within stopover latitudes are poorly understood (Arzel et al. 2006; Newton 2006; Webb et al. 2010). Previous studies have examined the importance of local habitat conditions and landscape structure, (Webb et al. 2010; Pearse et al. 2012; Beatty et al. 2014); however, no studies have accounted for local food resource availability, which may mediate the role of landscape structure (Taft and Haig 2006). Thus, I examined the influence of wetland food availability, as well as additional local habitat conditions and wetland distribution on wetland habitat use by spring migrating dabbling ducks. During two springs with differing precipitation and subsequent wetland inundation and distribution, my objectives were to: (1) determine if dabbling duck density and species richness were associated with available seeds and/or aquatic invertebrate density, (2) assess variation in

wetland habitat use relative to available energy derived from wetland plant seeds, and (3) evaluate the relationship between wetland use and available wetland distribution within the surrounding landscape.

STUDY AREA

Nebraska's Rainwater Basin spans 21 counties in south-central Nebraska, located south of the Platte River (Figure: 1.1). A majority of wetlands within the region are classified as playas, described as small depressional sink wetlands lined with a relatively impermeable clay layer and located at the low spot of a closed-basin watershed (Bolen et al. 1989; Smith 2003). Most playas in the RWB are seasonally or ephemerally inundated, with historic hydrologic processes driven by surface runoff following intense precipitation and accumulated snowmelt (Bolen et al. 1989; Smith 2003; Cariveau et al. 2011). Historically, the RWB included >11,000 playas, which would have provided approximately 80,000 ha of wetland habitat (Bishop and Vrtiska 2008). However, drainage ditches, concentration pits, sedimentation, and agricultural expansion resulted in long-term declines of up to 90% of wetland area within the region (Raines et al. 1990; Bishop and Vrtiska 2008; LaGrange 2005). Over 70% of the RWB upland landscape has been converted from mixed prairie grassland to cropped maize, soybean, and wheat (United States Geological Survey 2014), which corresponds to the approximate 70% of RWB wetlands considered degraded by agricultural runoff (Smith 2003). Precipitation preceding spring of 2014 was minimal, and consequently available (i.e., inundated) wetland habitat was relatively limited (National Oceanic and Atmospheric Administration 2017; Table 1.1). Precipitation preceding spring of 2015 was moderate, which corresponded to more frequently observed wetland inundation frequencies.

METHODS

Study Site Selection

Study sites were located on public lands managed by the Nebraska Game and Parks Commission and the United States Fish and Wildlife Service, private conservation easement lands enrolled in the Wetlands Reserve Program (WRP), and on inundated private lands managed for agriculture production (Figure 1.1). In the RWB, public and WRP wetlands were managed for early successional plant species using vegetative disturbance techniques including grazing, mowing, disking, and prescribed burns. For this portion of my study, inundated private lands managed for agricultural production include historical wetlands embedded within cropped fields that may be farmed during dry years, but capable of producing native moist-soil plant species during wet years when not farmed. I stratified potential study sites by county and identified the six counties containing the greatest number of inundated wetlands (with inundated area \geq 1ha), then randomly selected individual wetlands within these counties (Stafford et al. 2006; Tapp and Webb 2015). In 2014, I apportioned 11 public, three WRP, and three inundated private lands managed for agriculture from Phelps, Clay, Fillmore, York, Seward, and Custer counties (Custer County is located 40km to the North of the RWB in the Central Table Playa wetland complex). In 2015, I selected 12 public, 10 WRP, and two inundated private lands managed for agriculture from Phelps, Clay, Fillmore, York, Seward, and Hamilton counties. Availability of WRP and agriculture wetlands was limited by lack of precipitation and landowner cooperation during both years. I obtain access to an additional 22 dry private wetlands in 2014 and 2015 in the event of precipitation sufficient to inundate those wetlands. I obtained information from land managers on management strategies that occurred during the previous three years, including grazing, mowing/disking, burning, ground water pumping, and idle.

Waterfowl Surveys

Prior to the arrival of spring migrants, I established the boundary of each study site by walking the wetted perimeter with a handheld GPS, and surveyed for observation/vantage points. The number of observation points used at each wetland was based primarily on inundated area: 1 point in wetlands ≤ 5 ha; 2 points in wetlands 5 -25 ha, 3 points in wetlands 25 -100 ha; and 4 points in wetlands > 100 ha (Brown and Dinsmore 1986; Webb et al. 2010). Additional vantage points were added when vegetative complexity within a wetland prevented an observer from viewing $\geq 70\%$ of wetland surface water.

To quantify dabbling duck habitat use, I conducted weekly avian surveys from 23 February through 26 April 2014 and from 22 February through 18 April 2015. I divided daylight hours into 4 time intervals: sunrise-8:00, 8:00-12:00, 12:00-16:00, and 16:00-sunset and randomly sampled wetlands during different time intervals each week (Webb et al. 2010). I visited each vantage point for 10 minutes that allowed me to identify all visible species present. Exact counts were made for each species with < 100 individuals observed from a vantage point; however, when individuals representing a single species were > 100 , visual estimates were made (Webb et al. 2010).

Local Habitat Characteristics

Local wetland variables evaluated in this study included those thought to influence dabbling duck habitat use at spring migration stopover sites (Taft and Haig 2006; Webb et al. 2010; Williams et al. 2014). Here I describe collection methods for three local food and three local habitat conditions at each study site (Table 1.2).

Seed Collection and Processing

I quantified moist-soil seed and waste grain density in study sites during two sampling events each spring. Initial seed samples were collected prior to migrant arrival (mid-February through the first week of March) and during a second event after the majority of dabbling ducks had left the region (last two weeks of April). During each sampling event, I collected 10-20 soil cores along a single transect oriented through the widest portion of each study site (Figure 1.2.; Greer et al. 2007; Behney et al. 2014). Soil core samples were 10cm in diameter (78.54cm²) and included the top 5cm of substrate (Olmstead et al. 2013, Evans-Peters et al. 2012). All samples were collected randomly along the established transect in locations where water depths < 30cm (preferred range of foraging depths for dabbling ducks; Fredrickson 1991) and where vegetative cover was < 50%. Soil core samples were stored at ~ -10°C and transported to the University of Missouri for processing.

Soil cores were thawed at room temperature and deflocculated in a hydrogen peroxide and baking soda solution (Hagy and Kaminski 2012b). I rinsed soil samples through a series of two graduated sieves (#10[250 µm], and #50 [500 µm]) to remove small clay particles, and isolated remaining materials into coarse and fine samples (Greer et al. 2007). Samples were dried to constant dry-mass at 60° C for 48 hours in a convection oven and weighed to the nearest 0.1mg. Seeds were removed from the entire coarse sample, whereas fine samples were subsampled (1/4 volumes). I identified all seeds to genus, and weighed them to the nearest 0.1mg (Naylor 2002) and converted to kg/ha (Kross et al. 2008). I determined available energy derived from seeds (kcal/ha) by multiplying mean density of an individual seed taxa by the corresponding published value of TME (Appendix A).

Aquatic Insect Collections and Processing

I assessed benthic and nektonic aquatic invertebrate communities at study sites in alternating weeks during both years. At each wetland I established 3-5 (dependent on wetland area) randomly located 3m x 3m sample plots at water depths < 30cm (preferred range of foraging depths for dabbling ducks; Fredrickson 1991) and where vegetative cover was < 50% (Figure 1.3.). Within each plot, I collected two nektonic samples using a 500 μm rectangular sweep net (Murkin et al. 1994; Tapp and Webb 2015). The net was lowered vertically into the water column, pressed firmly against the substrate, and bounced through the water column for a distance of 1.1 m (0.5m^2 ; Klemm et al. 1990; Davis and Bidwell 2008). Within each sample plot, I also collected two benthic samples using a 10-cm diameter x 5-cm deep benthic core sampler (Swanson 1983; Tapp and Webb 2015) in an undisturbed area adjacent to the corresponding nektonic sampling location. Benthic and nektonic samples were preserved in 70% ethanol to prevent deterioration (Murkin and Kadlec 1986b) and transported to the University of Missouri for processing.

To increase efficiency in invertebrate sorting and identification, I stained samples with rose bengal for 24 hours before processing (Sherfy et al. 2000; Tapp and Webb 2015). I rinsed invertebrate samples through a series of two graduated sieves (#50 [500 μm] and #10[250 μm]) to remove small clay particles and partitioned remaining materials into coarse and fine samples (Tapp and Webb 2015). I used a Folsom wheel sample splitter (Aquatic Research Instruments) to subsample (1/4 volumes) debris retained by the 250 μm sieve and 500 μm (Meyer et al. 2011; Whiting et al. 2011; Tapp and Webb 2015). Aquatic invertebrates were removed from remaining debris, identified to the lowest practical and reliably identifiable taxonomic level, measured to the nearest millimeter, and catalogued (Kaminski and Prince 1981; Straub 2012). Density

estimates for individual taxa were obtained using published dry length-mass regressions (Duffy and Labar 1994; Benke et al. 1999; Appendix B). When a length-mass regression was not available for specific taxa, I used estimates from similar species observed in comparable habitats (Benke and Huryn 2006). I was unable to identify published length-mass regression models for taxon similar to phylum Nematoda and class Hirudinea. Further, I was unable to obtain published length-mass regression models for taxa within the order Gastropoda that did not include shell mass. Using a subset of specimens collected from my study, I developed dry length-mass regressions for Nematoda and Hirudinea, and length-mass regressions that excluded the shells of families Planorbidae, Physidae, and Bithyniidae following methods described by Benke et al. (1999; Appendix C). Biomass estimates were pooled and averaged for all taxa collected at a wetland for each sampling event and converted to biomass density estimates (kg/ha).

Additional Local Habitat Variables

Inundated wetland area was measured every other week by walking the wetted edge of each study site with a handheld GPS unit. When available, aerial and satellite imagery were used to estimate wetland area for weeks when ground estimates were not measured. On alternating weeks when area was not physically measured and aerial imagery was not available, an average area was calculated based on area measured during the preceding and following week. I measured percent vegetative cover and water depth during alternating weeks at 12-16 random locations within a wetland. I visually estimated percent vegetative cover and open water within a 1m x 1m quadrat (Wirwa 2009) and measured water depth (cm) at the center of each quadrat. I averaged all habitat measurements collected during a single sampling event. On alternating

weeks when I did not measure water depth and percent vegetative cover, I calculated an average for both metrics based on measurements recorded during the preceding and following week.

Landscape Structure Variables

I classified four explanatory variables describing landscape context: 1) number of inundated wetlands ≥ 1 ha within 2.5km of a study site, 2) inundated wetland area (ha) within 2.5km of a study site, 3) number of inundated wetlands ≥ 1 ha within 5km of a study site, and 4) inundated wetland area (ha) within 5km of a study site. For this portion of my thesis, inundated wetland area and density includes historical wetlands modified for crop production, however capable of ponding water during years of high precipitation. I selected these ranges because they were similar to mean female mallard foraging flight distances reported by Link et al. (2011). I downloaded Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) data sets (from www.earthexplorere.usgs.gov) using the USGS Bulk Download Application. I downloaded 2014 imagery from three scenes using the Worldwide Reference System (WRS; path/row) during dates ranging from February 21 – April 26. I downloaded 2015 imagery from two scenes using WRS during dates ranging from February 24 – April 13 (Appendix D.1). Scenes were visually inspected and considered unusable when atmosphere disturbance occurred (Hansen and Loveland 2012). Supplemental satellite imagery was obtained from Google Earth recorded on 17 April 2014 and multispectral orthophotography collected by aircraft during the weeks of March 9, 2014, and March 8, 2015 (Rainwater Basin Joint Venture's annual habitat survey). I used supplemental imagery to estimate wetland area on the landscape during weeks when Landsat imagery was not available. For weeks when no imagery was available, I projected wetland area based on the mean rate of change during the preceding two weeks.

I processed Landsat 8 imagery in ArcMap 10.3 (ESRI 2015) using ModelBuilder to develop, edit, and manage model workflow (Appendix D.2). I combined bands 1-7 into a single RGB layer; Band 1 (Blue_0.45–0.52 μm) was used to identify water, and Bands 5 (SWIR_1.55-1.75 μm) and 6 (SWIR_1.55-1.75 μm) offered the best contrast to blue (Li et al. 2013; Yang et al. 2015). Bands 1, 5, and 6 were stacked to differentiate inundated wetlands from upland cropped fields, grasslands, and urban areas. Band 8 (15-meter resolution panchromatic raster) was fused with the stacked composite RGB raster to improve resolution using the Pan Sharpening Function (Narma et al. 2010).

I performed a supervised classification in ArcMap that allowed me to create specific land classes based on the location of known land cover types (Ozesmi and Bauer 2002; MacAlister and Mahaxay 2009; Liu et al. 2016). I apportioned four specific land cover classifications: water, cropped fields, upland grasslands, and urban areas. I used wetlands that were inundated during my field seasons as training data for water (polygons developed by walking wetted edge of study sites), and high-resolution multispectral orthophotography aerial imagery to visually develop training data for cropped fields, upland grasslands, and urban centers. I used a Maximum Likelihood Classification to assign each raster pixel with a unique value that corresponded with pixels observed in my training data (Ozesmi and Bauer 2002; MacAlister and Mahaxay 2009). Pixels classified as water were extracted using the Raster Calculator function and converted to shapefiles.

I designated 2.5km and 5km buffers around each study site and clipped all inundated polygons within the buffers. The clipped shapefile included wetland polygons, as well as impoundments, streams, and false surface water polygons (e.g. shadows casted by cloud cover). To distinguish wetland polygons from other aquatic non-wetland polygons, I superimposed the

constructed shapefile with a layer provided by the National Wetlands Inventory (NWI; USFWS 2011). I visually inspected contradictions between the two layers, and eliminated discrepancies based on personal judgement and knowledge of the area.

I conducted a paired t-test to validate Landsat methods developed in this study by comparing known wetland area (ground truthed by walking wetted edge of each wetland with handheld GPS unit) and projected wetland area obtained from processed Landsat data. Area estimates were only included if Landsat imagery was recorded within 7 days of a wetland being ground truthed (Appendix D.3). Additionally, wetland sites concurrently pumped with ground water were not included in the analysis. There were no statistical differences in known wetland area and projected wetland area ($t_{76,0.05} = 0.45$).

Statistical Analysis

I analyzed habitat use for mallards, Northern pintails, and all other dabbling ducks (hereafter other dabbling ducks) independently given differences in spring migration chronology and foraging strategies in the RWB (Pearse et al. 2011; Tidwell et al. 2013). I developed a set of *a priori* candidate models for explaining mallard, Northern pintail, and other dabbling duck density based on ecologically reasonable scenarios supported by published literature and personal experience (Taft and Haig 2006; Webb et al. 2010; Williams et al. 2014; Table 1.3). I also developed a similar set of *a priori* candidate models for explaining species richness of all dabbling ducks (Table 1.4). Observations were removed if hunting activity was observed at a wetland during the time of a survey, or if vegetative or ice cover was >90%, thus precluding use by dabbling ducks. I fit linear mixed effects models in program R (lme4: Bates and Maechler 2016) to assess *a priori* candidate models. Duck density for all species assessed had zero

inflated distributions. I normalized distributions using a Yeo-Johnson Power Transformation (Yeo and Johnson 2000). To prevent heteroscedasticity in fitted residuals, I scaled and centered independent variables on zero (Gelman 2008). Site was included in all *a priori* models as a mixed effect since observations were taken multiple times (weekly surveys) from the same wetlands (Hagy et al. 2014). I considered week in all models as a fixed effect, given the strong relationship it had with dabbling duck density due to migration chronology during both years. I evaluated multicollinearity among covariates within models using Pearson's r-correlation index and discarded models with unacceptable levels ($|r| > 0.70$; Dormann et al. 2013) until only independent variables remained (Kross et al. 2008).

I analyzed *a priori* models from each year separately because of differences in available wetland habitat and consequently, dabbling duck density. Fit linear mixed effect models were used in R version 3.3.1 (lme4; Bates and Maechler 2016) to test *a priori* models for dabbling duck density and species richness. I used Akaike Information Criterion (AICc, AICcmodavg; Mazerolle 2016) to assess the relative utility of candidate models, and considered models within two AICc values of the best fit model as competitive (Burnham and Anderson 2002; Richards 2005). For each model selected, I performed a Shapiro-Wilk test on the distribution of the fitted residuals for normality and discarded models when the resulting p-value was < 0.05 (Shapiro and Francia 1972). Given the strong correlation between seed density and energy, as well as wetland densities within 2km and 5km of a study site ($R^2 > 0.70$), I did not average parameter estimates in competing models (Cade 2015).

RESULTS

Waterfowl Surveys

During spring 2014, I conducted 147 waterfowl surveys at 17 wetlands from 23 February through 26 April, although two study sites became dry during the final three weeks of the field season and thus were not surveyed for bird use. I observed eight species of dabbling ducks (Anatinae; n = 278,590), seven species of diving ducks (Aythinae; n = 789), and five goose species (Anserinae; n = 11,685; Appendix E). Mallards (*Anas platyrhynchos*) and Northern pintails (*A. acuta*) accounted for >50% of the total number of waterfowl observed, at 37% and 32% respectively. Species classified as other dabbling ducks, diving ducks, and geese accounted for approximately 27%, < 1%, and 4% of the remaining waterfowl observed, respectively. During spring 2015, I conducted 184 waterfowl surveys at 24 wetlands from 22 February through 18 April. Two study sites were dry during the final four weeks of my field season. I observed eight species of dabbling ducks (n = 41,774), six species of diving ducks (n = 807), and four goose species (n = 18,920; Appendix F). Mallards and Northern pintails accounted for <50% of the total number of birds observed at 25% and 19% respectively. Other dabbling ducks, diving ducks, and geese accounted for approximately 25%, 1%, and 30% of the remaining waterfowl observed, respectively.

Local Habitat Characteristics

I collected 1,260 seed core samples at wetlands where waterfowl surveys were conducted in 2014 and 2015. Mean seed density in 2014 was 443 (± 185)kg/ha, and ranged from 21 – 1,459kg/ha. The weighted average for TME across all sites in 2014 was 1,793 (± 195)kcal/kg, and mean energy availability was 747,586kcal/ha. Mean seed density in 2015 was 614

(± 124)kg/ha, and ranged from 34 – 1462kg/ha. The weighted average for TME across all sites in 2015 was 1,679 (± 125)kcal/kg, and mean energy was 1,046,096kcal/ha. I collected 1,524 aquatic invertebrate samples in 2014 and 2015. I identified 45 invertebrate families from nektonic and benthic communities (Appendices H and I). Among the most frequently observed taxonomic orders during both years were *Diptera*, *Gastropoda*, *Odonata*, *Copepoda*, and *Anostraca*. In 2014, mean invertebrate density was 10.26 (± 4.4)kg/ha and ranged from 0.27 – 33.20kg/ha, whereas, in 2015, mean invertebrate density was 16.16 (± 4.65)kg/ha and ranged from 1.34 – 44.34kg (Appendix G).

Water depth varied by week ($F_{8,144,0.05} = 5.06$), however wetland area and vegetative cover did not differ among weeks in 2014. Across all weeks, mean wetland area was 17.18 (± 4.51) ha and ranged from 0.30 – 131.80ha, mean water depth was 13.34 (± 1.22) cm and ranged from 0.30 – 31.05cm, and mean vegetative cover was 27 (± 28)% and ranged from 0 – 65%. In 2015, wetland area ($F_{7,184,0.05} = 2.79$), and water depth ($F_{7,184,0.05} = 3.22$) varied among weeks; however, percent vegetative cover did not differ among weeks. Across all weeks, mean wetland area was 10.20 (± 0.98)ha and ranged from 0.40 – 35.32ha, mean water depth was 14.68 (± 1.09)cm and ranged from 4.88 – 29.10cm, and mean vegetative cover was 26 (± 8)% and ranged from 6 – 84% (Appendix G).

Landscape Characteristics

Mean inundated area within 2.5km of a study site was 4.19 (± 1.39)ha and ranged from 0.00 – 45.75ha during spring 2014. The mean number of wetlands within 2.5km of a study site was 0.78 (± 0.19) and ranged from 0 – 4. Mean inundated area within 5km of a study site was 19.93 (± 6.14) ha and ranged from 0.00 – 155.33ha during spring 2014. The mean number of wetlands within 5km of a study site was 1.70 (± 0.32) and ranged from 0 – 5. Inundated area

within 2.5km and 5km varied across weeks in 2014 ($F_{(8, 128, 0.05)} = 4.80$ & $F_{(8, 128, 0.05)} = 7.76$ respectively). In 2015, mean inundated area within 2.5km of a study site was 21.26 (± 5.30)ha, and ranged from 0.00 – 167.00ha. The mean number of wetland units within 2.5km of a study site was 1.44 (± 0.22), and ranged from 0 – 5. Mean inundated area within 5km of a study site was 46.49 (± 9.86)ha, and ranged from 0.00 – 278.99ha during spring 2015. The mean number of wetland units within 5km of a study site was 2.99 (± 0.42), and ranged from 0.00 – 9.00. Inundated area within 2.5km and 5km varied across all weeks in 2015 ($F_{(7, 161, 0.05)} = 11.83$ & $F_{(7, 161, 0.05)} = 18.91$ respectively; Appendix G).

Mallard Models

Only observations from weeks 3-7 were considered in 2014 mallard models since they accounted for >98% of the total mallards observed. Wetlands < 1ha in wetted area were also omitted since only 17% of those wetlands had mallard densities > 0. I identified three competing models, which accounted for 72% of the total AICc weight (Table 1.5). Explanatory variables in the most likely model included wetland area (+), water depth (+), energy (+), and wetland density within 2.5km (+; Figure 1.4). Seed density (+) and wetland density within 5km (+) also appeared in subsequent competing models. Water depth, energy, seed density, and wetland density within 2.5km and 5km were significant across all models in which they occurred ($p > 0.05$; Table 1.6).

Only observations from weeks 3-6 were considered in 2015 mallard models since they accounted for >95% of the total mallards observed. I identified four competing models, which accounted for 68% of the total AICc weight (Table 1.5). Energy (+) was the only explanatory variable included in the most likely model (Figure 1.5). Seed density (+), and wetland density

within 2.5km (-) and 5km (+) were observed in subsequent competing models. Energy and seed density were significant in all competing models in which it occurred ($p > 0.05$; Table 1.6).

Northern Pintail Models

Only observations from weeks 3-6 were considered in 2014 Northern pintail models since those weeks accounted for $> 95\%$ of the total number of Northern pintails observed. I identified three competing models (Table 1.7) explaining Northern pintail density, which accounted for 68% of the total AICc weight. Explanatory variables included in the most likely model were wetland area (+), water depth (+), seed density (+), and wetland density within 5km (+; Figure 1.6). Subsequent competing models also included energy (+) and wetland density within 2.5km (+). All explanatory variables were significant in competing models in which they occurred ($p < 0.05$; Table 1.8).

Only observations from weeks 3-5 were considered in 2015 Northern pintail models since those weeks accounted for $> 95\%$ of the total number of Northern pintails observed. Wetlands < 1 ha in wetted area were also omitted since abundance in those units were zero. I identified four competing models (Table 1.7) that accounted for 60% of the total AICc weight. Energy (+) was the only explanatory variable included in the most likely model (Figure 1.7). Seed density (+), wetland density within 2.5km (-), and wetland density within 5km (+) occurred in subsequent competing models. Energy was the only explanatory variable significant in all models in which it occurred. ($p > 0.05$; Table 1.8).

Other dabbling ducks Models

Only observations from weeks 3-9 were included in 2014 other dabbling ducks models since those weeks accounted for $>95\%$ of the total number of other dabbling ducks ducks

observed. I identified three competing models, which accounted for 6% of the total AICc weight (Table 1.9). Explanatory variables in the most likely model included wetland area (+), water depth (+), energy (+), and wetland density within 2.5km (+; Figure 1.8). Subsequent competing models also included seed density (+), and wetland density within 5km (+). Water depth, energy, seed density, wetland density within 2.5km and 5km were significant in all competing models in which they occurred (Table 1.10).

Only observations from weeks 4-8 were considered in 2015 other dabbling ducks models since those weeks accounted for > 95% of the total number of other dabbling ducks observed. I identified six competing models (Table 1.9) that accounted for 65% of the total AICc weight. Seed density was the only explanatory variable observed in the most likely model (+; Figure 1.9). Subsequent competing models also included energy (+), wetland density within 2.5km (-) and wetland density within 5km (-). No explanatory variables were useful for explaining other dabbling duck density (Table 1.10).

Species Richness Models

Observations from all weeks were included when modeling species richness in 2014. I identified eight competing models, which accounted for 56% of the total AICc weight (Table 1.11). Wetland area (+), percent vegetative cover (+) and water depth (+) were observed in the most likely model (Figure 1.10). Subsequent competing models also included energy (+), seed density (+), wetland density within 2.5km (+), and wetland density within 5km (+). Wetland area and water depth were significant in all competing models in which they occurred (Table 1.12).

Observations from all weeks were included when modeling species richness in 2015. I identified six competing models, which accounted for 72% of the total AICc weight (Table 1.11). Three models had fitted residuals that were not normally distributed and thus were removed from the competing model set. Explanatory variables in the most likely model included wetland area (+), water depth (-), and vegetative cover (-; Figure 1.11). Invertebrate density (+) was the only additional explanatory variable observed in subsequent competing models. Wetland area and percent vegetative cover were significant in all competing models in which they occurred (Table 1.12).

DISCUSSION

Dabbling Duck Density

The strong relationships I observed between dabbling duck densities and measures of local food availability in 2014 and 2015 support previous research indicating the importance of food resource acquisition for dabbling ducks during spring migration (Arzel et al. 2006; Anteau and Afton 2008; Williams et al. 2014; Beatty et al. 2014). Although seed density was a strong explanatory variable for habitat use, I was able to increase resolution of models by including a measurement of plant seed derived energy (i.e. energy; kcal/ha) for mallards and other dabbling ducks in 2014, and mallards and Northern pintails in 2015. Because dabbling ducks are tactile foragers, they are able to differentiate among natural seed species and may select seeds based on energetic and nutritional value (Lovvorn and Gillingham 1996; Gurd 2005; Klaassen et al. 2007; Hagy and Kaminski 2015). Although I did not collect esophagus contents of dabbling ducks, and therefore unable to verify whether ducks were selecting seeds with the greatest energetic benefit, confirmation by previous spring diet studies in the RWB support these findings. Tidwell et al. (2013) reported percentages of multiple native seed taxon with relatively high TME values

observed in esophagi of mallards and blue-winged teal (*Anas discors*) to be disproportionately greater than what was available in wetlands where birds were collected.

Minimal precipitation during both years limited the number of inundated cropped fields in the RWB, and consequently, I was unable to integrate inundated cropped fields into my study. Waste grains reportedly have TME values four times greater than common native wetland seeds (Reineke et al. 1989; Sherfy 1999; Checkett et al. 2002); however, they fail to provide dabbling ducks with all nutrients needed to complete life history requirements and maintain optimal body condition (Baldassare et al. 1983; Loesch and Kaminski 1989; Miller et al. 2000). A better understanding of habitat selection and tradeoffs between wetlands comprised of native vegetation and row crops would assist wetland managers when determining which plant communities to promote.

Aquatic invertebrate density did not appear in competing models for dabbling duck density, and all model estimates from my candidate model set indicated no relationship between dabbling duck density and aquatic invertebrate density. Relative to moist-soil seeds, the energetic value of aquatic invertebrates is minimal (Sherfy 1999; Dugger et al. 2007). Further, mean available dry invertebrate density for 2014 and 2015 was 14kg/ha or < 3% of total available food aggregate, whereas estimated available seed density during the same period was 528kg/ha or >97% of total available food aggregate. It is unlikely that invertebrates provide a meaningful energetic benefit to dabbling ducks; however, spring diet studies have estimated aggregate invertebrate density in esophagi to range from 28–35% in mallards (Hitchcock 2008; Tidwell et al. 2013), 36–58% for blue-winged teal (Hitchcock 2008; Tidwell et al. 2013), and 5% for Northern pintails (Pearse et al. 2011).

My study indicates invertebrate density has no influence on dabbling duck density, although previous diet studies demonstrated dabbling ducks selected for invertebrates disproportionate to their availability (Hitchcock 2008; Tidwell et al. 2013). One explanation for these conflicting results is that I estimated invertebrate density at the wetland level as opposed to a finer patch level within a wetland (Johnson 1980; Klaassen et al. 2007; Hagy and Kaminski 2015). That is, my estimates assume invertebrate density is homogeneous throughout a wetland, but in reality, densities were likely patchily distributed (Murkin et al. 1994; Taft and Haig 2006). Alternatively, dabbling ducks select a wetland and then select a patch within the wetland to forage where invertebrate abundance is relatively dense (Johnson 1980; Klaassen et al. 2007; Hagy and Kaminski 2015). A temporal bias in my study design that did not account for a seasonal diet shift, which likely occurs in the RWB given its near proximity to breeding grounds may affect results (~250km; Taylor 1978; Hitchcock 2008). Hitchcock (2008) reported that selection for invertebrates increased towards the end of spring migration in the Upper Mississippi Great Lakes Region, positioned geographically on the southern edge of North America's breeding grounds. Although the exact timing of when dabbling ducks shift diets from high-caloric to high-protein food resources is unclear and is likely influenced by a number of factors (e.g., food availability, use of exogenous nutrients in clutch formation, annual weather). Regardless, the diet shift is a requirement for optimal reproduction to occur (Swanson et al. 1985; Euliss and Harris 1987; Ankney and Alisauskas 1991).

Wetland area and water depth were useful for explaining Northern pintail and other dabbling ducks densities in 2014, but were not useful when explaining distribution of any taxa in 2015. Given the dry conditions associated with 2014, mean water depths at several sites fell far below optimal foraging depths (13cm - 25cm; Colwell and Taft 2000), which may have impeded

dabbling duck maneuverability and limited foraging efficiency. I would anticipate an opposing result during years of greater precipitation and deeper water depths, which would prevent dabbling ducks from reaching benthic food resources (Fredrickson 1991). Previous studies have also reported vegetative structure as a useful explanatory variable of wetland use given its effect on seed and invertebrate production, maneuverability, and ability to provide seclusion (Smith et al. 2004; Davis and Bidwell 2008; Webb et al. 2010; Tapp and Webb 2016). Although percent vegetative cover was not present in any competing models for explaining density, it was useful for explaining 2014 mallard and other dabbling ducks densities in multiple *likely* candidate models.

The most likely models for explaining mallard, Northern pintail, and other dabbling ducks densities in 2014 included a strong positive relationship with wetland density in the surrounding landscape. Minimal precipitation in spring 2014 resulted in limited inundated wetland habitat, and consequently, both seed and invertebrate availability were limited at the landscape scale. These unfavorable conditions made it not only important for dabbling ducks to identify sites with the greatest energetic benefits, but equally important to use habitat within a complex where the energetic costs of traveling among foraging sites were minimized (Brown and Dinsmore 1986; Fredrickson and Heitmeyer 1988; Pearse et al. 2012; Beatty et al. 2014). However, large-scale distribution of wetland habitat might also clarify why I did not observe significant relationships between dabbling ducks and wetland density during 2015 (Taft and Haig 2006). Mean wetland area within 2.5km of a study site was five times greater in 2015 than 2014 (Appendix G). Accordingly, mean dabbling duck density in 2014 was 76 ducks/ha and only 23 ducks/ha in 2015. It is likely that dabbling ducks were selecting habitat within a wetland

complex; however, given the increase in adjacent wetland habitat, ducks would have spread-out and decreased densities (per unit area).

Dabbling Duck Richness

Preferred habitat conditions for the eight species of dabbling ducks commonly observed in the RWB have been widely studied during breeding and wintering seasons; however, rarely at mid-latitudes during spring migration (Arzel et al. 2006; Newton 2006; Webb et al. 2010). My results indicated that species richness is likely greater at larger wetlands with heterogeneous habitat conditions, which is consistent with previous results in the RWB (Webb et al. 2010). Anecdotally, I noticed that larger wetlands provided a greater amount of inundated area between the depths of 1 – 30cm, which would accommodate the precise tipping depths (foraging strategy used to reach benthic food resources) of all dabbling duck species observed in the RWB (Euliss and Harris 1987; Guillemain et al. 2002). Subtle morphological differences observed in dabbling ducks are a hallmark example of how congeneric species coexist (Schoener 1971; Guillemain et al. 2002). For dabbling ducks, ecomorphology (differences in body length) allow birds to segregate along a foraging depth gradient (Lack 1971; Euliss and Harris 1987; Guillemain et al. 2002). Although I did not assess species richness for other water birds, I predict partitioning wetland habitat by a combination of depth and area would be more obvious when considering habitat requirements of diving ducks, geese, and shorebirds common to the region (Brown and Dinsmore 1986; Fairbairn and Dinsmore 2001; Webb et al. 2010).

To my knowledge, this is the only study that simultaneously recognized the importance of food availability and landscape structure when assessing habitat use by dabbling ducks during spring migration. My AICc model weights implied that models including multiple spatial scales explained the most variation in dabbling duck density during dry years when wetland habitat was

limited; however, the importance of landscape structure may diminish as wetland habitat availability increases at a regional scale. Similar results observed for wintering dunlin (*Caldiris alpine*) in the Willamette Valley of Oregon by Taft and Haig (2006) led the authors to conclude that wetland density is likely a mediating influence between foraging birds and food availability. This concept may also apply at a smaller scale for selection of specific food items. Tidwell et al. (2013) reported dabbling ducks collected in wetland complexes selected some seed species disproportionately to availability; however, dabbling ducks collected in isolated wetlands selected seeds in proportion to availability. Thus, my results indicate that optimal spring habitat conditions at both the local and landscape level will not only allow dabbling ducks to access food resources that satisfy daily nutritional and energetic requirements, but also attain and improve nutrient reserves that influence future success on the breeding grounds (Taylor 1978; Devries et al. 2008).

MANAGEMENT IMPLICATIONS

With limited wetland habitat remaining at historic spring staging areas (Dahl 1990), determining how dabbling ducks distribute themselves in relation to local habitat conditions and landscape structure is critical (Webb et al. 2010; Pearse et al. 2012; Beatty et al. 2014). Results of this study suggest spring migrants are concentrating at wetlands that provide not only the greatest seed densities, but also habitats that provide the greatest energetic benefit. Managers will be most effective in accommodating the energetic demand of spring migrants by promoting early-successional plant communities that offer greater yields of seed density and often associated with favorable TME values (Fredrickson and Taylor 1982; Kross et al. 2008; Williams et al. 2014). However, there is a need for future studies to focus on estimating TME values for all dabbling ducks and food resource items commonly consumed (Williams et al.

2014). Previous studies focused primarily on TME estimates for mallards and Northern pintails, which only account for ~50% of the dabbling ducks using the RWB (Sherfy 1999; Sherfy et al. 2001; Checkett et al. 2002; Kaminski et al. 2003; Ballard et al. 2004). Improved TME estimates for all plant species will allow wetland managers to determine which plant species to promote, given the objective of providing the best foraging conditions possible.

Wetland restoration is an essential tool for conservation planning in regions where wetland loss is excessive (North American Waterfowl Management Plan 2012). My results indicate that future restoration planning should identify areas nearby isolated wetlands, or within established wetland complexes. Doing so will not only maximize dabbling duck use at the restored wetland, but also increase use of nearby wetlands. Conservation strategies accounting for wetland density within a region are likely to be particularly important in semi-arid regions where wetland availability may be sporadic and inundation frequency difficult to predict (Fairbairn and Dinsmore 2001; Petrie et al. 2016).

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Table 1.1. Monthly precipitation accumulation from November through April as an index to assess spring inundation frequency at playa wetlands in the Rainwater Basin. Rainfall and snowfall data summarize long-term average precipitation (year 2006-2015), as well as years during which the study was conducted (2014 and 2015) and a year of above average precipitation (2010) for comparison.

Metric	Month	Long-Term Average	2010	2014	2015
Rainfall (cm)	November_Prior Year	3.00	0.43	2.46	0.74
	December_Prior Year	1.60	4.47	0.28	1.93
	January_Year of	1.30	1.37	0.81	1.19
	February_Year of	1.70	1.68	0.84	1.78
	March_Year of	4.60	6.43	0.48	0.41
	April_Year of	6.40	4.93	7.39	8.81
	Total		18.59	19.30	12.27
Snowfall (cm)	November_Prior Year	7.62	0.00	0.51	5.59
	December_Prior Year	12.70	67.31	5.59	14.99
	January_Year of	17.78	14.22	11.43	2.29
	February_Year of	15.24	18.80	16.76	28.19
	March_Year of	2.54	1.27	3.05	1.52
	April_Year of	< 2.54	0.00	4.32	2.03
	Total		55.88	101.60	41.66

*Source National Oceanic and Atmospheric Administration 2017: Grand Island Nebraska

Table 1.2. Variable designations, descriptions, and methods used to collect local and landscape habitat data used to explain dabbling duck density and species richness in Rainwater Basin wetlands during spring 2014 and 2015.

Variable	Description	Methods
Area	Wetted wetland area	Wetland area was estimated bi-monthly by walking the wetted edge with a handheld GPS unit.
Veg	% emergent vegetation	Percentage of wetland area where emergent vegetation occurred opposed to open water.
Depth	Mean water depth	Water depth (cm) was measured bi-monthly at each wetland.
Invert	Invertebrate density	Aquatic invertebrate density (kg/ha) available as food resources. Six nektonic sweep and six benthic core samples were collected at each wetland bi-monthly.
Seed	Seed density	Seed density (kg/ha) available as food resources. Ten - Fifteen cores were collected prior to spring migration and an additional 10-15 cores were collected when it was believed that the majority of waterfowl had migrated out of the RWB.
Energy	Kilocalories	Energy (kilocalories/ha) is the estimated seed density (kg/ha) of an individual plant taxa multiplied by an estimated true metabolizable energy (gross energy of a plant seed minus gross energy of the excreta) value for that plant taxa.
2kmDen	2.5km Density	Inundated wetland and inundated cropped field density within a 2.5km radius of a study site. Inundated conditions were assessed every ~16 days using satellite and aerial imagery.
5kmDen	5km Density	Inundated wetland and inundated cropped field within a 5km radius of a study site. Inundated conditions were assessed every ~16 days using satellite and aerial imagery.
Week	Week of survey	Week designated as a numeric number between 1-9 in 2014 (23 February – 26 April) and 1-8 in 2015 (22 February – 18 April 2015).
Site	Study site	Individual wetland included in study during spring 2014 (n=18) and 2015 (n=24).

Table 1.3. *A priori* candidate models used to explain variation in dabbling duck density at wetlands in Nebraska's Rainwater Basin during spring 2014 and 2015.

Level	Model
Local_Non Food Resources	Week + Area + Depth + Veg + (1 Site)
Local_Non Food + Food Resources	Week + Area + Depth + Veg + Invert + (1 Site) Week + Area + Depth + Veg + Seed + (1 Site)† Week + Area + Depth + Veg + Energy + (1 Site)†
Local_Non Food + Landscape	Week + Area + Depth + Veg + 2kmDen + (1 Site) Week + Area + Depth + Veg + 2kmDen + (1 Site)
Local_Food Resources + Landscape	Week + Invert + 2kmDen + (1 Site) Week + Invert + 5kmDen + (1 Site) Week + Seed + 2kmDen + (1 Site) Week + Seed + 5kmDen + (1 Site) Week + Energy + 2kmDen + (1 Site) Week + Energy + 5kmDen + (1 Site)
Local_Food Resources	Week + Invert + (1 Site) Week + Seed + (1 Site) Week + Energy + (1 Site)
Landscape	Week + 2kmDen + (1 Site) Week + 5kmDen + (1 Site)
Local_Non Food + Food Resources + Landscape	Week + Area + Depth + Veg + Invert + 2kmDen + (1 Site) Week + Area + Depth + Veg + Invert + 5kmDen + (1 Site) Week + Area + Depth + Veg + Seed + 2kmDen + (1 Site)† Week + Area + Depth + Veg + Seed + 5kmDen + (1 Site)† Week + Area + Depth + Veg + Energy + 2kmDen + (1 Site)† Week + Area + Depth + Veg + Energy + 5kmDen + (1 Site)†

† Vegetation was not included in 2014 model because of multicollinearity with seed and energy variables

Table 1.4. *A priori* candidate models used to explain dabbling ducks species richness at wetlands in Nebraska's Rainwater basin, during spring 2014 and 2015.

Level	Model
Local_Non Food	Week + Area + (1 Site)
	Week + Area + Depth + Veg + (1 Site)
Local_Non Food + Food Resources	Week + Area + Depth + Veg + Invert + (1 Site)
	Week + Area + Depth + Veg + Seed + (1 Site)†
	Week + Area + Depth + Veg + Energy + (1 Site)†
Local_Non Food + Landscape	Week + Area + Depth + Veg + 2kmDen + (1 Site)
	Week + Area + Depth + Veg + 2kmDen + (1 Site)
Local_Resources + Landscape	Week + Area + Invert + 2kmDen + (1 Site)
	Week + Area + Invert + 5kmDen + (1 Site)
	Week + Area + Seed + 2kmDen + (1 Site)
	Week + Area + Seed + 5kmDen + (1 Site)
	Week + Area + Energy + 2kmDen + (1 Site)
	Week + Area + Energy + 5kmDen + (1 Site)
Local_Food Resources	Week + Area + Invert + (1 Site)
	Week + Area + Seed + (1 Site)
	Week + Area + Energy + (1 Site)
Landscape	Week + Area + 2kmDen + (1 Site)
	Week + Area + 5kmDen + (1 Site)
Local_Non Food + Food Resources + Landscape	Week + Area + Depth + Veg + Invert + 2kmDen + (1 Site)
	Week + Area + Depth + Veg + Invert + 5kmDen + (1 Site)
	Week + Area + Depth + Veg + Seed + 2kmDen + (1 Site)†
	Week + Area + Depth + Veg + Seed + 5kmDen + (1 Site)†
	Week + Area + Depth + Veg + Energy + 2kmDen + (1 Site)†
	Week + Area + Depth + Veg + Energy + 5kmDen + (1 Site)†

† Vegetation was not included in 2014 model resulting from multicollinearity with seed and energy

Table 1.5. Competing models explaining mallard density from linear mixed effect regression models. Candidate models were ranked using Akaike’s Information Criterion (AICc) and models within two AICc ($\Delta AICc$) were considered competing models. The AICc weight of a model relative to all candidate models (n=23) is denoted by ω and ML is the relative likelihood of the model given the data.

Year	Model ¹	K	AIC _c	ΔAIC_c	ω	ML
2014	Week + Area + Depth + Energy + 2kmDen + (1 Site)	8	251.09	0.00	0.28	1.00
	Week + Area + Depth + Seed + 2kmDen + (1 Site)	8	251.14	0.06	0.27	0.97
	Week + Area + Depth + Energy + 5kmDen + (1 Site)	8	252.03	0.95	0.17	0.62
2015	Week + Energy + (1 Site)	5	345.19	0.00	0.22	1.00
	Week + Seed + (1 Site)	5	345.39	0.20	0.20	0.90
	Week + Energy + 2kmDen + (1 Site)	6	346.17	0.97	0.14	0.61
	Week + Energy + 5kmDen + (1 Site)	6	346.49	1.29	0.12	0.52

¹Potential model variables (n =10) included wetland area (Area), mean water depth (Depth), kilograms/ha of seed (seed), kilocalories/ha derived from seed density (Energy), kilogram/ha of aquatic invertebrates (Invert), number of wetlands greater than 1ha in area within a 2.5km radius of a study site (2kmDen), and number of wetlands greater than 1ha in area within a 5km radius of a study site (5kmDen). The period assessed for an individual observation (week) was included in all models as a fixed effect, and the wetland where reoccurring observations were collected (Site) was included in all models as a random effect.

Table 1.6. Parameter estimates for competing candidate models ($\Delta AIC_c = 2$) from results of linear mixed effect models of mallard density during spring 2014 and 2015.

Year	Parameter	Model 1		Model 2		Model 3		Model 4	
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
2014	Week	-0.078	0.13	-.079	0.13	-0.12	0.13	-	-
	Area	0.20	0.12	0.20	0.12	0.20	0.13	-	-
	Water Depth	***0.65	0.14	***0.64	0.14	***0.54	0.15	-	-
	Energy	***0.38	0.13	-	-	***0.39	0.13	-	-
	Seed	-	-	***0.38	0.13	-	-	-	-
	2kmDen	***0.33	0.12	***0.35	0.12	-	-	-	-
	5kmDen	-	-	-	-	**0.32	0.13	-	-
	Intercept	***1.98	0.12	***1.98	0.12	***1.98	0.12	-	-
2015	Week	***-0.43	0.09	***-0.43	0.09	***-0.43	0.09	***-0.41	0.09
	Energy	***0.51	0.20	-	-	**0.50	0.20	***0.56	0.20
	Seed	-	-	**0.49	0.19	-	-	-	-
	2kmDen	-	-	-	-	-0.02	0.18	-	-
	5kmDen	-	-	-	-	-	-	0.21	0.19
	Intercept	***2.16	0.19	***2.17	0.19	***2.16	0.19	***2.17	0.18

*p<0.1; **p<0.05; ***p<0.01

Table 1.7. Competing models explaining Northern pintail density from linear mixed effect regression models. Candidate models were ranked using Akaike’s Information Criterion (AICc) and models within two AICc ($\Delta AICc$) were considered competing models. The AICc weight of a model relative to all candidate models (n=23) is denoted by ω and ML is the relative likelihood of the model given the data.

Year	Model ¹	K	AIC _c	ΔAIC_c	ω	ML
2014	Week + Area + Depth + Seed + 5kmDen + (1 Site)	8	209.00	0.00	0.33	1.00
	Week + Area + Depth + Energy + 5kmDen + (1 Site)	8	210.16	1.16	0.19	0.56
	Week + Area + Depth + Seed + 2kmDen + (1 Site)	8	210.50	1.15	0.16	0.47
2015	Week + Energy + (1 Site)	5	201.74	0.00	0.24	1.00
	Week + Energy + 5kmDen + (1 Site)	6	202.04	0.30	0.20	0.86
	Week + Seed + (1 Site)	5	203.13	1.39	0.12	0.50
	Week + Energy + 2kmDen + (1 Site)	6	203.60	1.86	0.09	0.39

¹Potential model variables (n=10) included wetland area (Area), mean water depth (Depth), kilograms/ha of seed (seed), kilocalories/ha derived from seed density (Energy), kilogram/ha of aquatic invertebrates (Invert), number of wetlands greater than 1ha in area within a 2.5km radius of a study site (2kmDen), and number of wetlands greater than 1ha in area within a 5km radius of a study site (5kmDen). The period assessed for an individual observation (week) was included in all models as a fixed effect, and the wetland where reoccurring observations were collected (Site) was included in all models as a random effect.

Table 1.8. Parameter estimates for competing candidate models ($\Delta AICc = 2$) from results of linear mixed effect models of Northern pintail density during spring 2014 and 2015.

Year	Parameter	Model 1		Model 2		Model 3		Model 4	
		Estimate	SE	Estimate	SE	Estimate	SE	Estimate	SE
2014	Week	-0.11	0.12	-0.11	0.12	-0.08	0.13	-	-
	Area	***0.44	0.16	***0.46	0.17	***0.43	0.17	-	-
	Depth	**0.43	0.19	**0.41	0.19	***0.52	0.19	-	-
	Energy	-	-	***0.45	0.17	-	-	-	-
	Seed	***0.48	0.17	-	-	***0.52	0.18	-	-
	2kmDen	-	-	-	-	**0.38	0.17	-	-
	5kmDen	***0.44	0.17	***0.48	0.17	-	-	-	-
	Intercept	***1.74	0.16	***1.74	0.17	***1.73	0.17	-	-
2015	Week	** -0.25	0.10	** -0.22	0.10	** -0.25	0.10	*** -0.27	0.10
	Energy	**0.41	0.19	**0.47	0.18	-	-	**0.38	0.19
	Seed	-	-	-	-	*0.31	0.34	-	-
	2kmDen	-	-	-	-	-	-	-0.13	0.18
	5kmDen	-	-	0.27	0.18	-	-	-	-
	Intercept	***1.70	0.19	***1.71	0.18	***1.69	0.19	***1.69	0.18

*p<0.1; **p<0.05; ***p<0.01

Table 1.9. Competing models explaining other dabbling ducks density from linear mixed effect regression models. Candidate models were ranked using Akaike's Information Criterion (AICc) and models within two AICc ($\Delta AICc$) were considered competing models. The AICc weight of a model relative to all candidate models (n=23) is denoted by ω and ML is the relative likelihood of the model given the data.

Year	Model ²	K	AIC _c	ΔAIC_c	ω	ML
2014	Week + Area + Depth + Energy + 2kmDen + (1 Site)	8	403.79	0.00	0.25	1.00
	Week + Area + Depth + Seed + 2kmDen + (1 Site)	8	403.98	0.20	0.23	0.91
	Week + Area + Depth + Energy + 5kmDen + (1 Site)	8	404.47	0.68	0.18	0.71
2015	Week + Seed + (1 Site)	5	416.60	0.00	0.17	1.00
	Week + 5kmDen + (1 Site)	5	417.24	0.64	0.12	0.72
	Week + Energy + (1 Site)	6	417.42	0.82	0.11	0.66
	Week + 2kmDen + (1 Site)	6	417.74	1.13	0.10	0.57
	Week + Seed + 5kmDen + (1 Site)	5	418.15	1.54	0.07	0.46
	Week + Seed + 2kmDen + (1 Site)	6	418.45	1.85	0.07	0.40

¹Excludes mallards and Northern Pintails

²Potential model variables (n =10) included wetland area (Area), mean water depth (Depth), kilograms/ha of seed (seed), kilocalories/ha derived from seed density (Energy), kilogram/ha of aquatic invertebrates (Invert), number of wetlands greater than 1ha in area within a 2.5km radius of a study site (2kmDen), and number of wetlands greater than 1ha in area within a 5km radius of a study site (5kmDen). The period assessed for an individual observation (week) was included in all models as a fixed effect, and the wetland where reoccurring observations were collected (Site) was included in all models as a random effect.

Table 1.10. Parameter estimates for competing candidate models ($\Delta AICc = 2$) from results of linear mixed effect models of other dabbling ducks density during spring 2014 and 2015.

Year	Parameter	Model 1		Model 2		Model 3		Model 4		Model 5		Model 6	
		Estimate	SE										
2014	Week	***0.61	0.15	***0.60	0.15	***0.58	0.14	-	-	-	-	-	-
	Area	0.17	0.15	0.17	0.15	0.19	0.14	-	-	-	-	-	-
	Depth	***0.85	0.18	***0.82	0.18	***0.77	0.17	-	-	-	-	-	-
	Energy	***0.44	0.16	-	-	***0.45	0.15	-	-	-	-	-	-
	Seed	-	-	***0.42	0.16	-	-	-	-	-	-	-	-
	2kmDen	**0.37	0.15	**0.39	0.15	-	-	-	-	-	-	-	-
	5kmDen	-	-	-	-	**0.35	0.15	-	-	-	-	-	-
	Intercept	***2.51	0.15	***2.51	0.15	***2.51	0.14	-	-	-	-	-	-
2015	Week	***0.45	0.14	***0.44	0.14	***0.45	0.14	***0.46	0.14	***0.44	0.14	***0.44	0.14
	Energy	-	-	-	-	0.26	0.25	-	-	-	-	-	-
	Seed	0.34	0.24	-	-	-	-	-	-	0.29	0.25	0.31	0.25
	2kmDen	-	-	-	-	-	-	-0.22	0.25	-	-	-0.15	0.24
	5kmDen	-	-	-0.28	0.25	-	-	-	-	-0.21	0.25	-	-
	Intercept	***3.01	0.24	***3.00	0.25	***3.01	0.24	***3.00	0.25	***3.01	0.24	***3.01	0.24

*p<0.1; **p<0.05; ***p<0.01

Table 1.11. Competing models explaining dabbling duck species richness from linear mixed effect regression models. Candidate models were ranked using Akaike’s Information Criterion (AICc) and models within two AICc ($\Delta AICc$) were considered competing models. The AICc weight of a model relative to all candidate models (n=23) is denoted by ω and ML is the relative likelihood of the model given the data.

Year	Model	K	AIC _c	ΔAIC_c	ω	ML
2014	Week + Area + Veg + Depth + (1 Site)	7	462.22	0.00	0.12	1.00
	Week + Area + Veg + Depth + 2kmDen + (1 Site)	8	462.93	0.71	0.08	0.70
	Week + Area + Veg + Depth + Seed + 2kmDen + (1 Site)	9	463.21	0.98	0.07	0.61
	Week + Area + Veg + Depth + Seed + (1 Site)	8	463.41	1.18	0.07	0.55
	Week + Area + 2kmDen + (1 Site)	6	463.41	1.19	0.07	0.55
	Week + Area + 5kmDen + (1 Site)	6	463.67	1.45	0.06	0.48
	Week + Area + Veg + Depth + 5kmDen + (1 Site)	8	463.87	1.64	0.05	0.44
	Week + Area + Veg + Depth + Energy + (1 Site)	8	463.94	1.72	0.05	0.42
2015	Week + Area + Veg + Depth + (1 Site)	7	509.89	0.00	0.22	1.00
	Week + Area + Veg + Depth + Invert + (1 Site)	8	510.93	1.04	0.13	0.59

¹Potential model variables (n=10) included wetland area (Area), mean water depth (Depth), kilograms/ha of seed (seed), kilocalories/ha derived from seed density (Energy), kilogram/ha of aquatic invertebrates (Invert), number of wetlands greater than 1ha in area within a 2.5km radius of a study site (2kmDen), and number of wetlands greater than 1ha in area within a 5km radius of a study site (5kmDen). The period assessed for an individual observation (week) was included in all models as a fixed effect, and the wetland where reoccurring observations were collected (Site) was included in all models as a random effect.

Table 1.12. Parameter estimates for competing candidate models ($\Delta AICc = 2$) from results of linear mixed effect models of species richness during spring 2014.

Year	Parameter	Model 1		Model 2		Model 3		Model 4		Model 5		Model 6		Model 7		Model 8		
		Estimate	SE															
2014	Week	***0.74	0.20	***0.84	0.20	***0.96	0.21	***0.81	0.20	***0.49	0.16	***0.49	0.16	***0.78	0.20	***0.79	0.20	
	Area	**0.84	0.34	**0.86	0.34	**0.87	0.31	**0.84	0.33	***1.22	0.38	***1.21	0.37	***0.87	0.34	***0.84	0.34	
	Veg	0.35	0.32	0.27	0.31	-0.03	0.36	0.14	0.37	-	-	-	-	0.30	0.32	0.21	0.36	
	Depth	**0.78	0.31	***0.82	0.30	***1.03	0.30	***0.92	0.31	-	-	-	-	**0.75	0.31	***0.86	0.32	
	Energy	-	-	-	-	-	-	-	-	-	-	-	-	-	-	-	0.33	0.42
	Seed	-	-	-	-	0.61	0.39	0.46	0.42	-	-	0.61	0.39	0.46	0.42	-	-	-
	2kmDen	-	-	0.38	0.3	*0.47	0.28	-	-	0.38	0.33	-	-	-	-	-	-	-
	5kmDen	-	-	-	-	-	-	-	-	-	-	-	0.36	0.35	0.26	0.31	-	-
	Intercept	***4.37	0.36	***4.38	0.34	***4.38	0.31	***4.37	0.33	***4.36	0.44	***4.37	0.43	***4.38	0.35	***4.37	0.34	-
2015	Week	***0.50	0.14	***0.54	0.14	-	-	-	-	-	-	-	-	-	-	-	-	
	Area	***0.86	0.19	***0.87	0.19	-	-	-	-	-	-	-	-	-	-	-	-	
	Veg	***-0.60	0.16	***-0.62	0.16	-	-	-	-	-	-	-	-	-	-	-	-	
	Depth	-0.01	0.18	-0.01	0.18	-	-	-	-	-	-	-	-	-	-	-	-	
	Invert	-	-	0.17	0.16	-	-	-	-	-	-	-	-	-	-	-	-	
	Intercept	***3.86	0.18	***3.86	0.17	-	-	-	-	-	-	-	-	-	-	-	-	

*p<0.1; **p<0.05; ***p<0.01

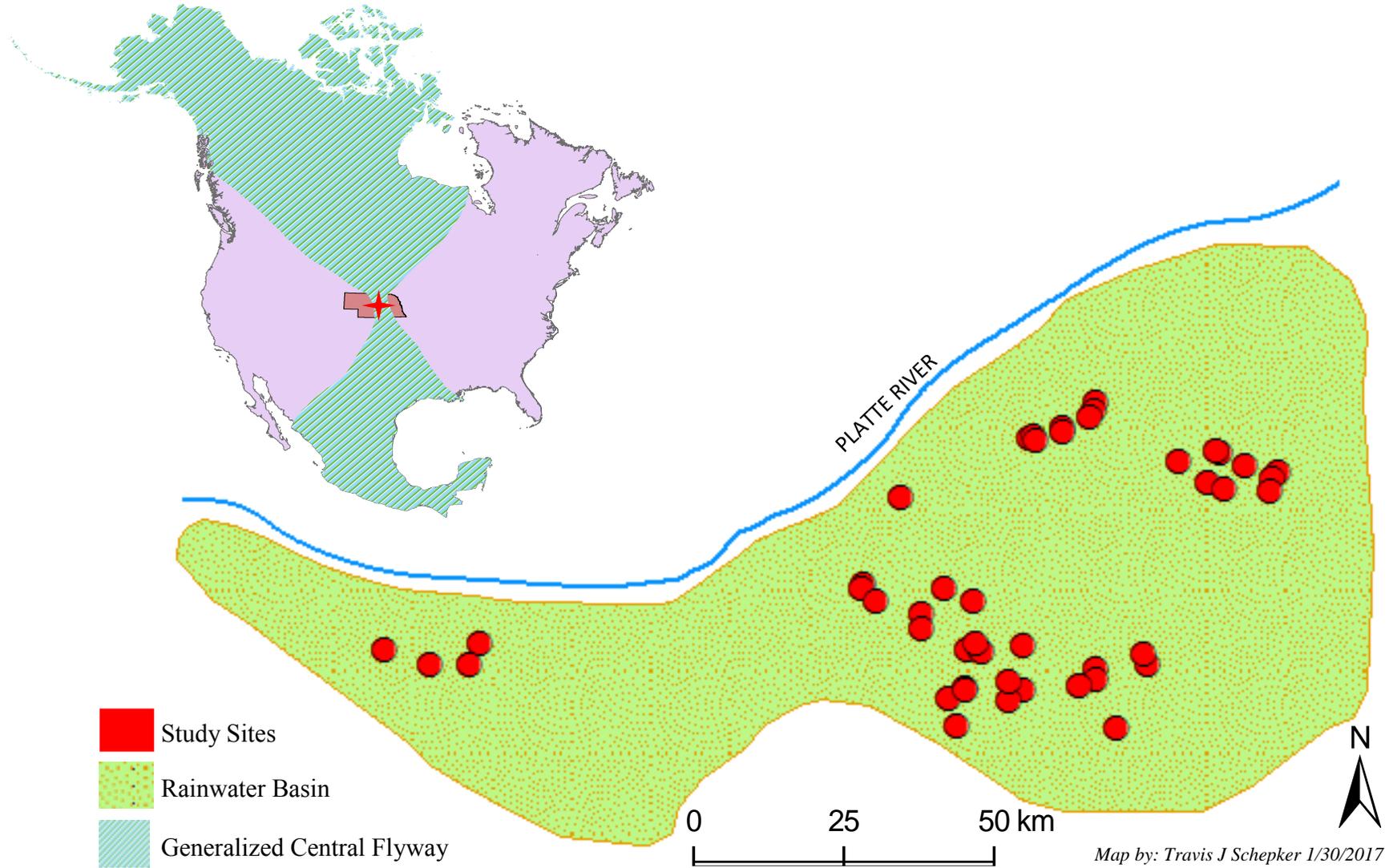


Figure 1.1 Nebraska Rainwater Basin playa wetlands are located in South-Central Nebraska, USA. Influences of local habitat characteristics and landscape structure were used to evaluate habitat use by waterfowl during springs 2014-2015.

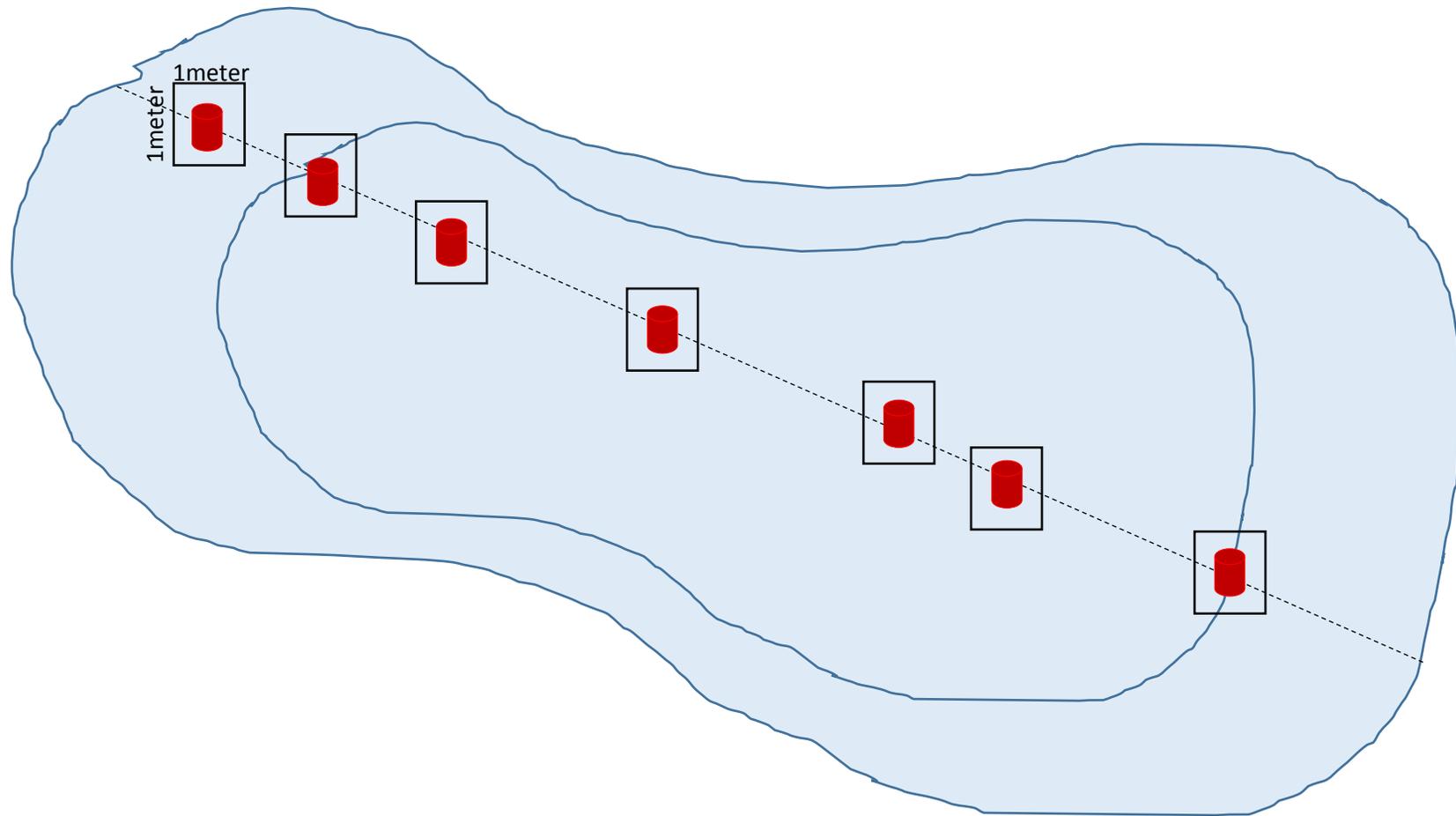


Figure 2.2. Illustration of methods used to collect seed cores, measure water depth, and measure percent vegetative cover at study sites. One meter by one meter quadrats were established at random locations along a transect spanning across the widest portion of a wetland. Seed core samples were extracted from the center of quadrats. Percentage of vegetative cover and water depth were recorded at each quadrat.

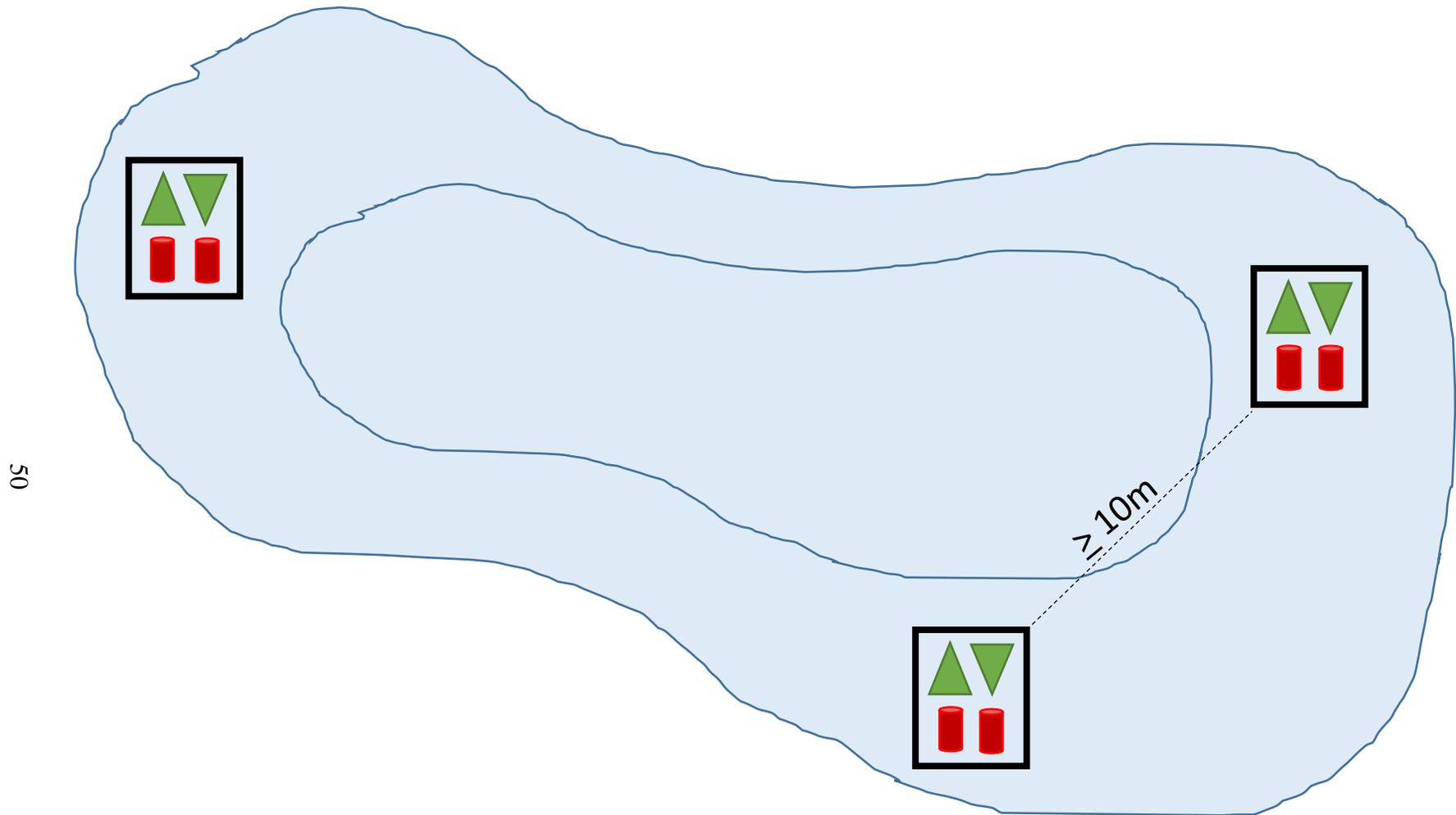


Figure 3.3. Illustration of methods used to collect benthic and nektonic invertebrates, water depth, and percent vegetative cover at study sites. Three to five random sample plots were established at each study site. Sample plots were 3m x 3m in area and spaced $\geq 10m$ apart. Two core samples and 2 sweep samples were collected within each plot. Water depth and vegetative cover were recorded at four locations within each sample plot.

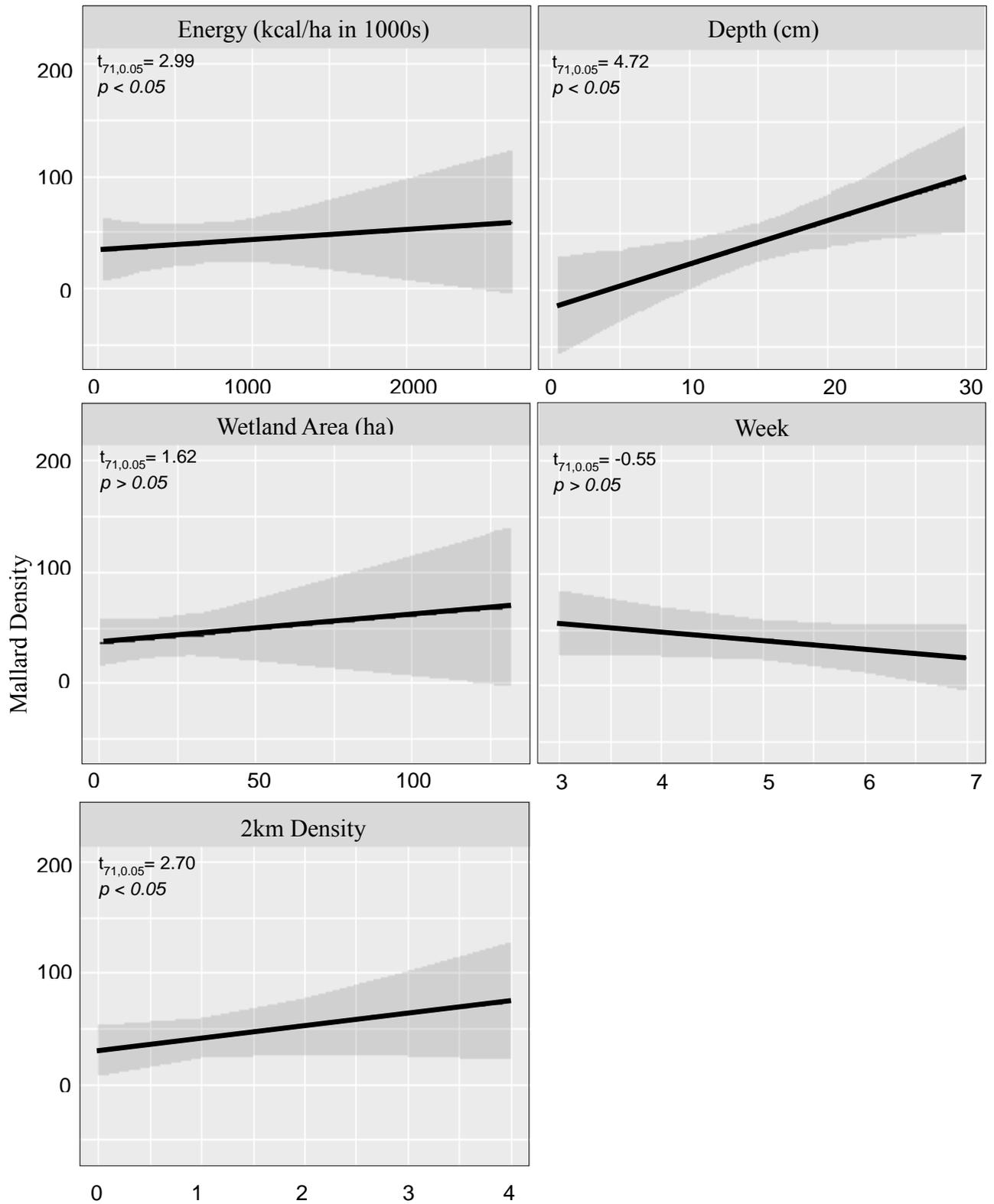


Figure 1.4. Marginal effect of variables used to explain 2014 mallard density at playa wetlands in Nebraska's Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

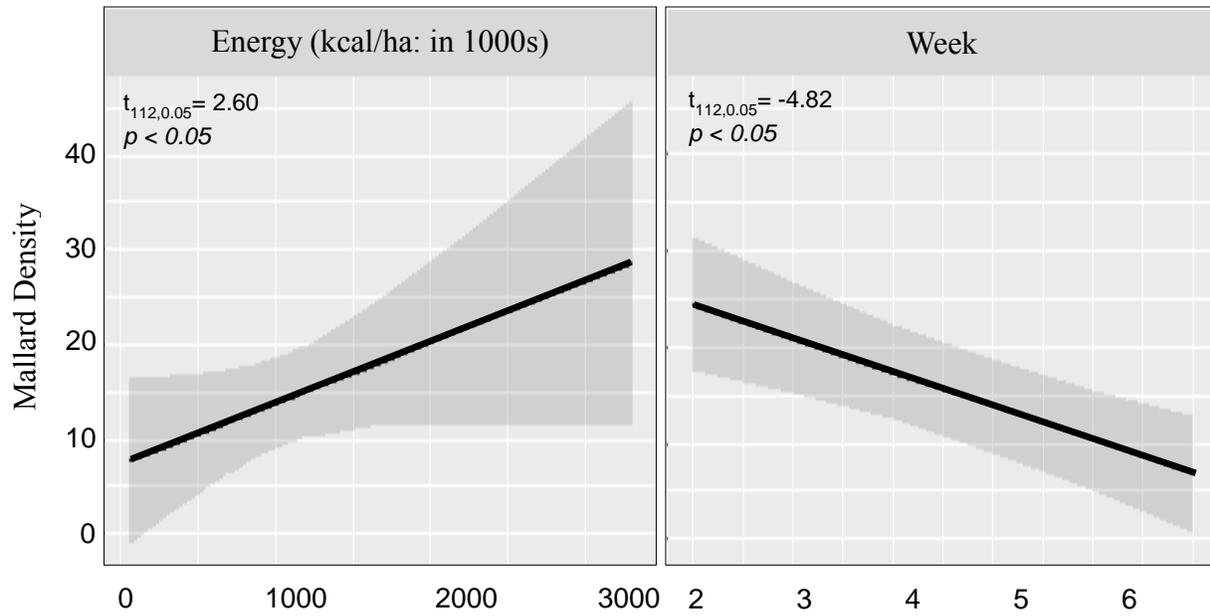


Figure 1.5. Marginal effect of variables used to explain 2015 mallard density at playa wetlands in Nebraska's Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

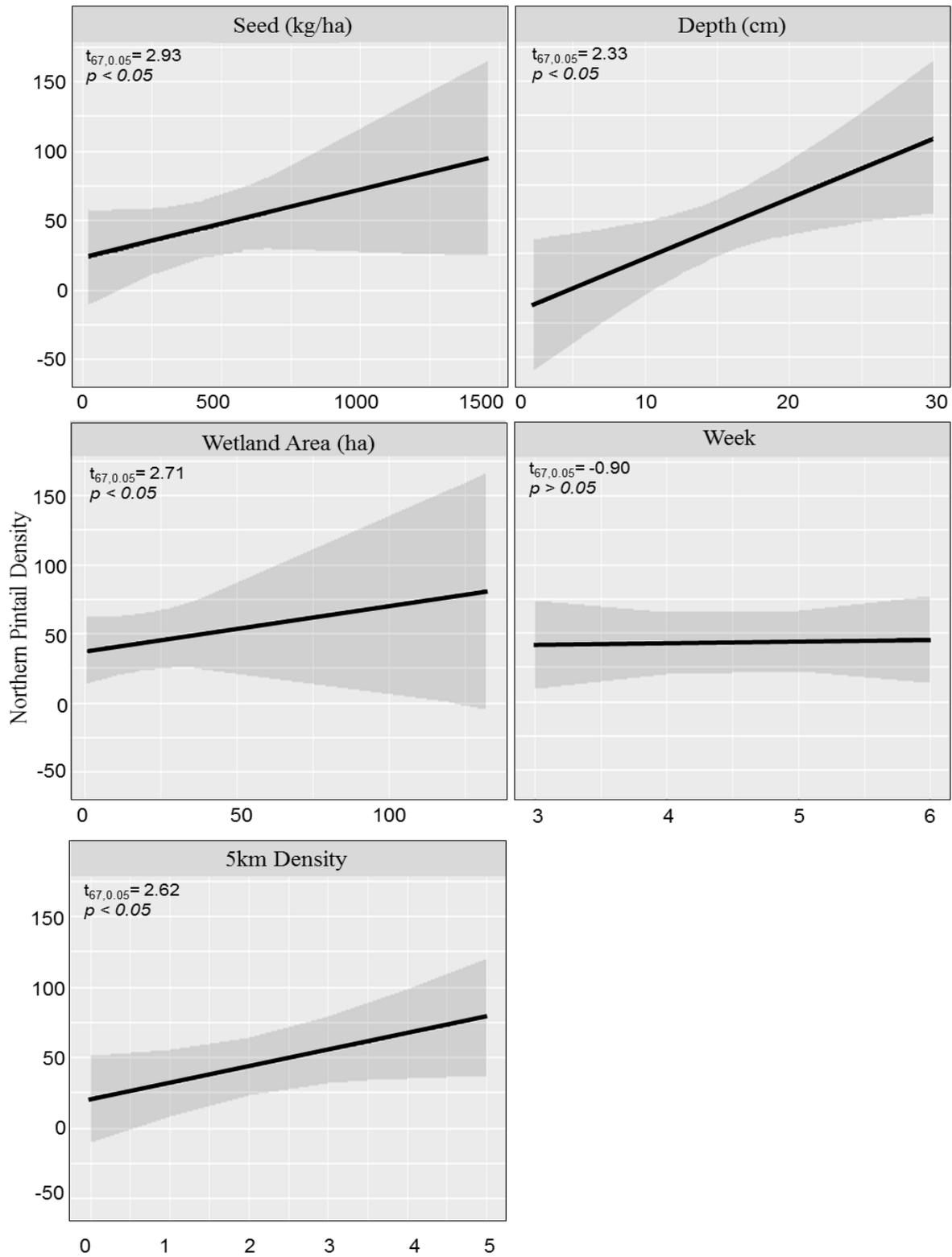


Figure 1.6. Marginal effect of variables used to explain 2014 Northern pintail density at playa wetlands in Nebraska’s Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

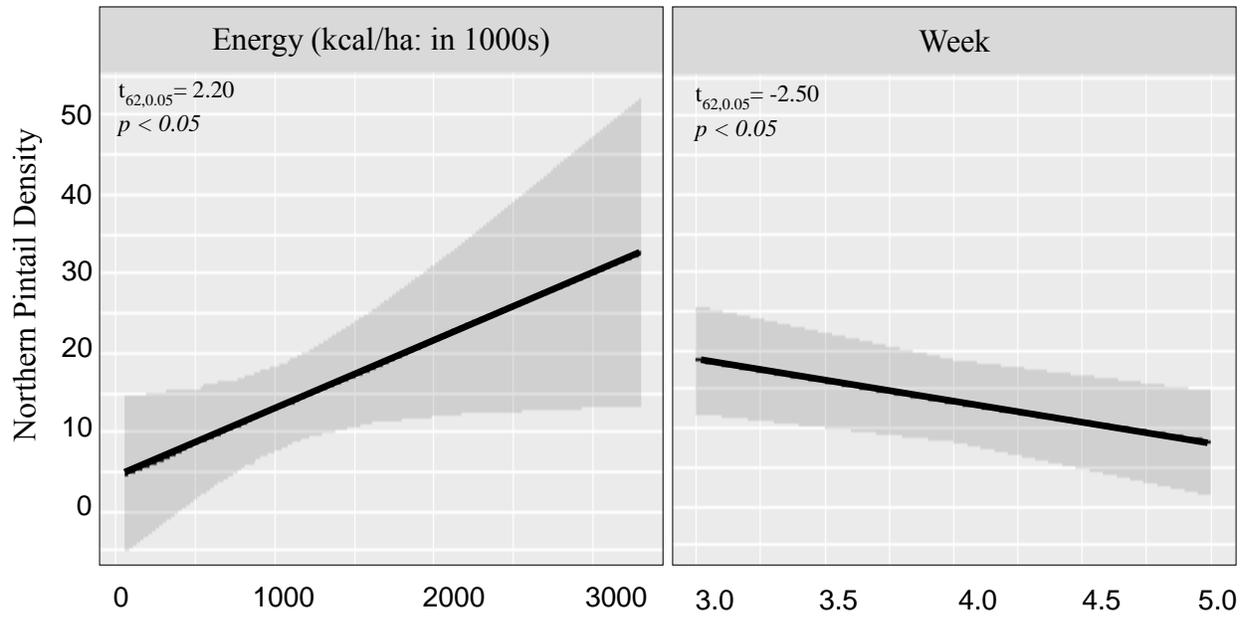


Figure 1.7. Marginal effect of variables used to explain 2015 Northern pintail density at playa wetlands in Nebraska’s Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

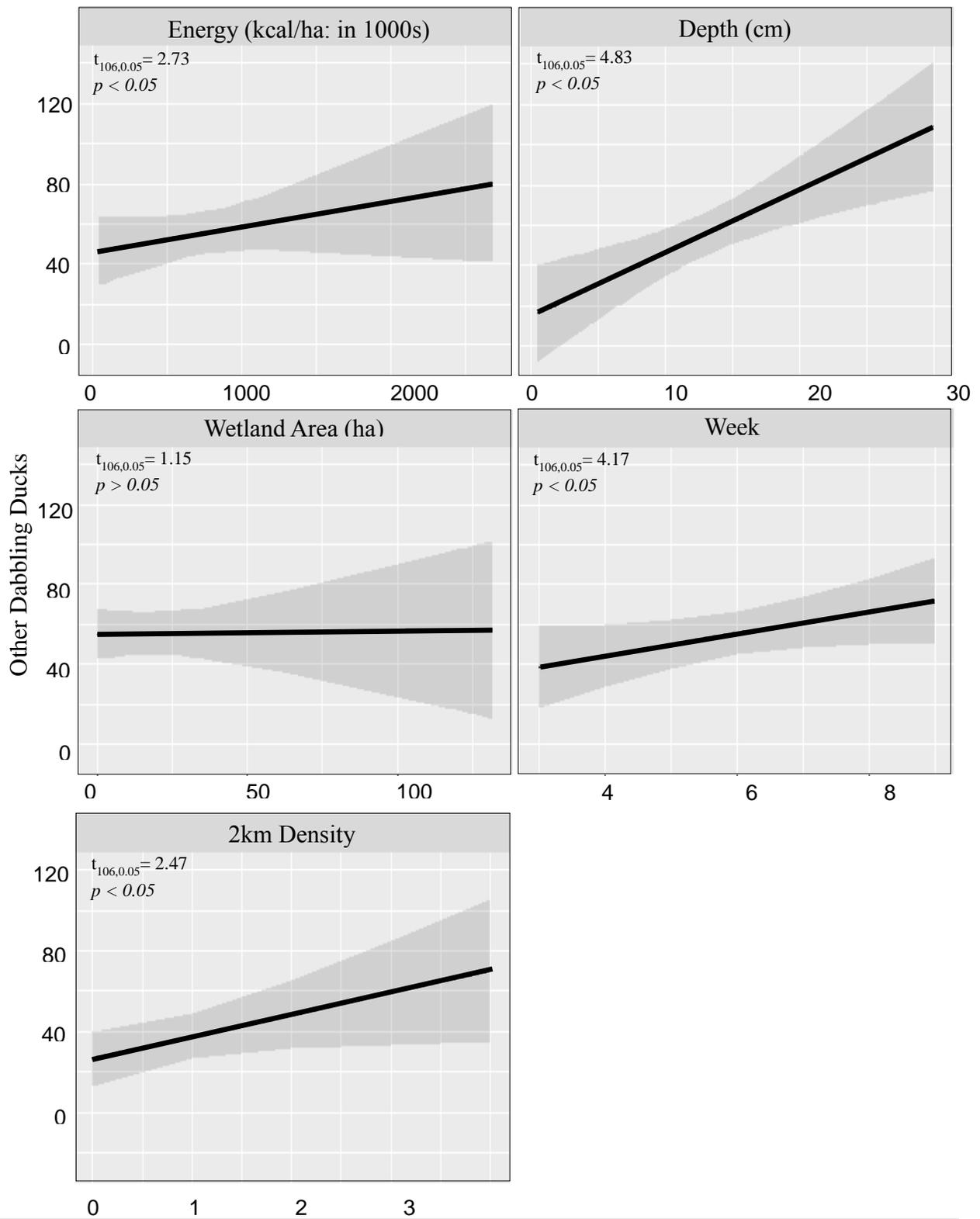


Figure 1.8. Marginal effect of variables used to explain 2014 other dabbling duck density at playa wetlands in Nebraska's Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

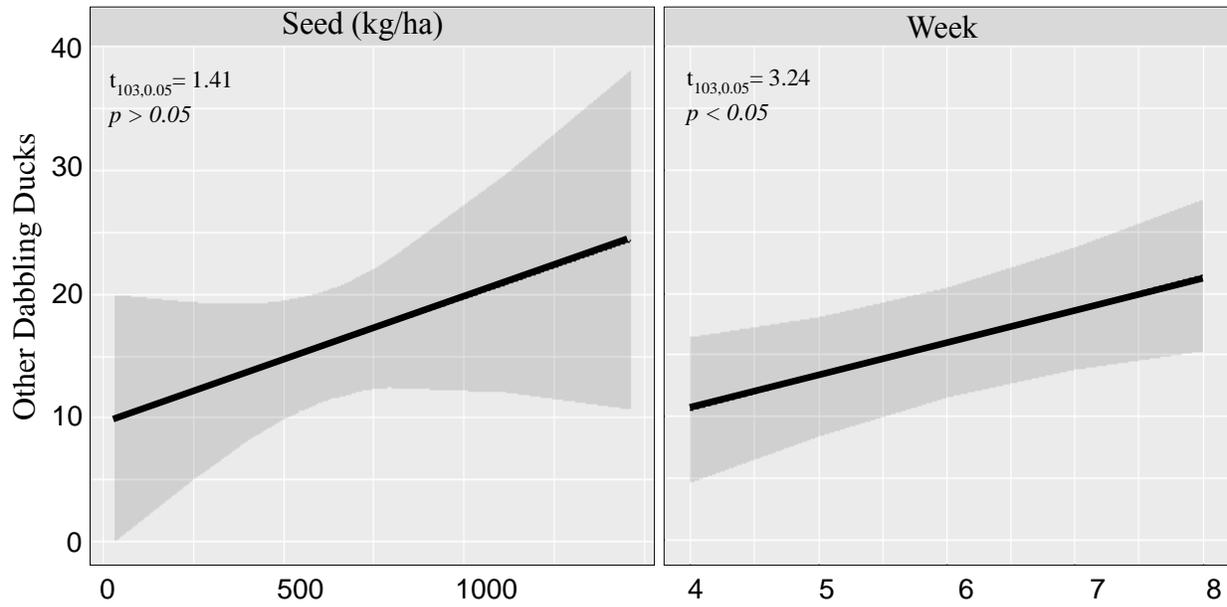


Figure 1.9. Marginal effect of variables used to explain 2015 other dabbling duck density at playa wetlands in Nebraska’s Rainwater Basin, USA. Variables observed in figure represent the most likely AICc model.

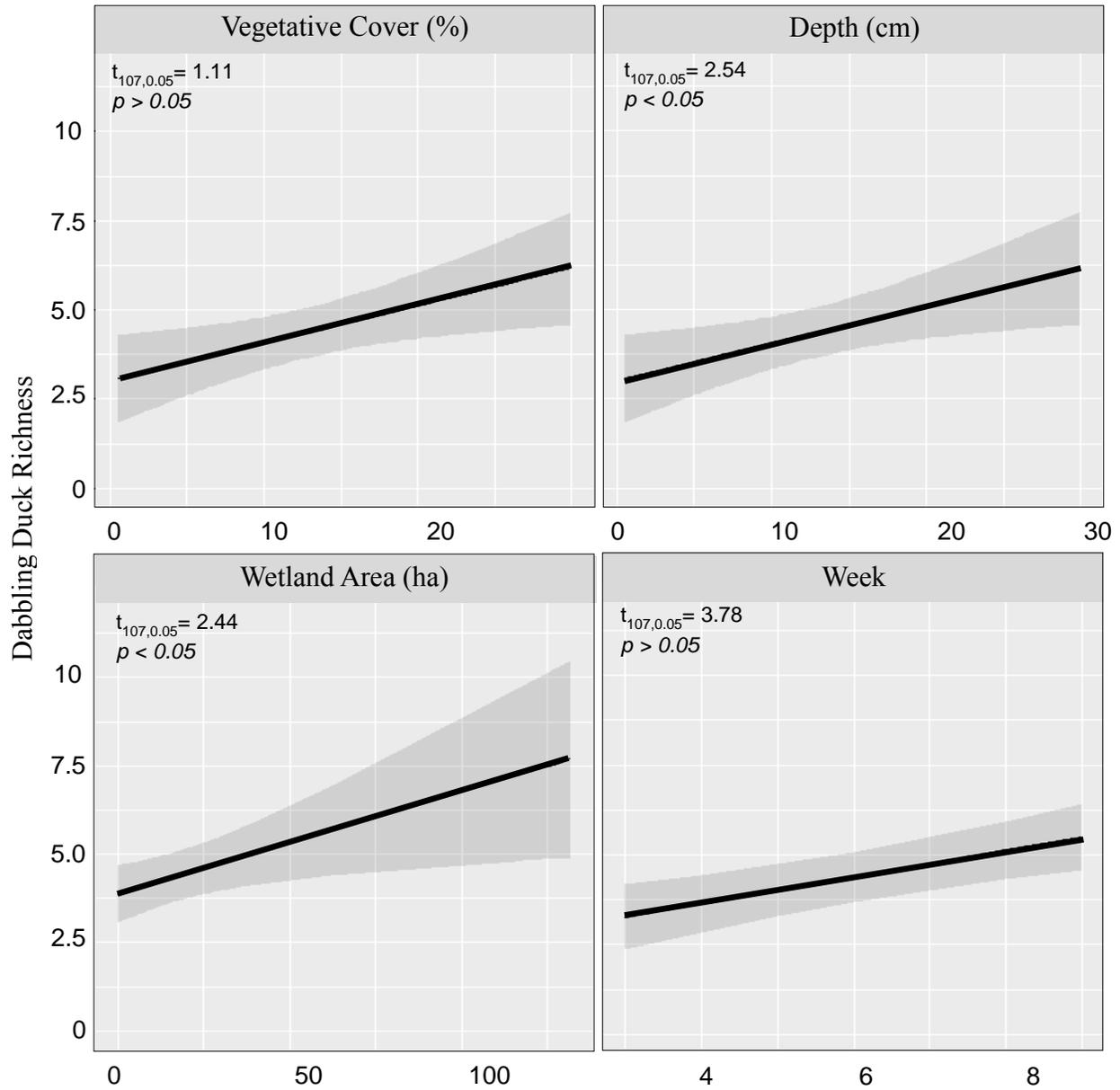


Figure 1.10. Marginal effect of variables used to explain 2014 dabbling duck richness at playa wetlands in Nebraska’s Rainwater Basin, USA. Variables observed in figure represent the top AICc model.

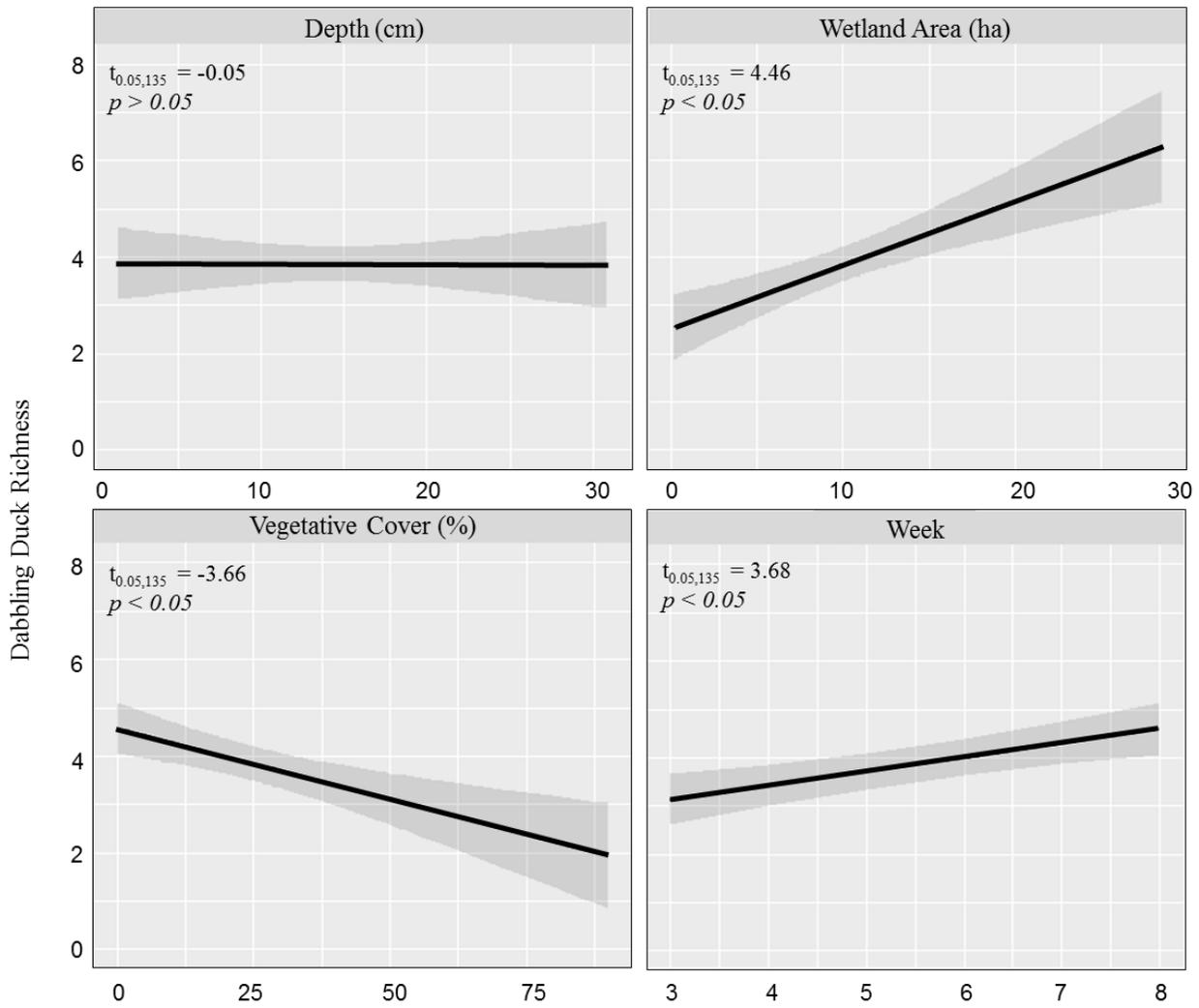


Figure 1.11. Marginal effect of variables used to explain 2015 dabbling duck richness at playa wetlands in Nebraska’s Rainwater Basin, USA. Variables observed in figure represent the top AICc model.

CHAPTER II: IS WATERFOWL FOOD AVAILABILITY LIMITED DURING SPRING MIGRATION? AN ENERGETIC ASSESSMENT OF PLAYAS IN THE RAINWATER BASIN OF NEBRASKA

The spring condition hypothesis acknowledges the potential relationship between wetland food availability at spring migration stopover sites and waterfowl nesting efforts at northern breeding grounds (Afton and Anderson 2001). During spring migration, many wetland birds forage on aquatic invertebrates and plant materials to accumulate the energy and protein needed to complete migration and initiate egg production (LaGrange and Dinsmore 1989; Pearse et al 2011; Straub 2012; Tidwell et al. 2013). When demand for quality food resources is unmet, waterfowl may arrive to breeding grounds in poorer body condition and consequently become less likely to achieve reproductive success (Heitmeyer and Fredrickson 1981; Devries et al. 2008; Anteau and Afton 2009). This cross-seasonal effect became evident during the preceding century when waterfowl foraging opportunities diminished along migration routes due to wetland loss and habitat degradation (Krapu 1981; Dahl 1990; Afton and Anderson 2001). Consequently, poor habitat conditions along spring migratory corridors contributed to North America's waterfowl population plummeting to historical lows, which prompted the protection, restoration, and enhancement of wetland habitats on a continental scale (North American Waterfowl Management Plan 1986).

Given the importance of food resource acquisition at mid-latitude migration stopover sites and subsequent effects on nest success and recruitment, waterfowl managers have adopted a bioenergetics approach to guide habitat conservation planning for remaining wetlands at spring staging areas (North American Waterfowl Management Plan 2012; Straub et al. 2012; Williams et al. 2014). For this approach to be effective, resource planners require accurate estimates of 1)

habitat-specific food resource density (dry weight per unit area), 2) habitat-specific estimates of true metabolizable energy (TME; kcal per dry weight), 3) habitat-specific availability (ha of inundated area), 4) species-specific daily energy expenditure (kcal), and 5) population goals for specific species (Haukos and Smith 1993; Kaminski et al. 2003; Kross et al. 2008; Williams et al. 2014). The utility of this model is likely ambiguous for planning in most regions given the need for more accurate habitat-specific estimates of food resource density and corresponding TME values (Straub et al. 2012; Williams et al. 2014).

The Rainwater Basin Joint Venture (RWBJV) is responsible for developing long-term management strategies and delivery of wetland habitat conservation to migratory waterfowl seeking resting and foraging opportunities in Nebraska's Rainwater Basin (RWB). Currently the RWBJV is tasked with providing habitats sufficient to support ~10 million spring migrants in a region that has experienced a 90% reduction in wetland habitat area (Schildman et al. 1984; LaGrange 2005; Bishop and Vrtiska 2008). Large waterfowl numbers and reduced wetland habitat likely promote intense competition among birds for limited resources (Bishop et al. 2016). One strategy for providing additional wetland resources within the region involves restoring ecological function for a portion of those wetlands converted to other land use during the previous century (Bishop and Vrtiska 2008). The Wetlands Reserve Program (WRP) is a voluntary conservation easement program, which provides private landowners financial and technical assistance to retire farmlands from agricultural production and restore historic wetland hydrology and topography to increase benefits for wildlife (King et al. 2006, Nugent et al. 2015). Although WRP offers financial incentives and technical support for the initial restoration, WRP participants have no contractual obligation to provide perpetual management treatments (Tapp and Webb 2015; Natural Resources Conservation Service 2017a). Cattle grazing and a variety of

mechanical soil disturbance treatments are commonly used to manage wetland vegetation throughout the RWB, with the intended goal of increasing seed production by promoting early successional plant communities (LaGrange 2005; Bishop and Vrtiska 2008; RWBJV 2013). Without sporadic disturbance, usually implemented through management practices, annual vegetative communities may transition into stands of dense perennial monocultures that offer minimal energetic value to migratory waterfowl (Haukos and Smith 1993; Bowyer et al. 2005; Strader and Stinson 2005; Kross et al. 2008). An alternative conservation easement strategy involves restoring the hydrology of non-functional cropped wetlands and encouraging private landowners to continue crop production (Natural Resources Conservation Service 2017b). This strategy provides agricultural producers an additional program for flood-prone cropland, where crops can be harvested during dry years, while maximizing foraging opportunities for migrating birds during years with sufficient precipitation (Nugent et al. 2015). Although conservation easement programs are an integral tool for developing long-term wetland management strategies in the RWB, the food resource estimates and associated energetic value for spring migrants on these wetland habitats have not been assessed.

The current bioenergetic model used for wetland management in the RWB relies on seed density and TME estimates derived from actively managed wetlands in southern United States during autumn (Haukos and Smith 1993; Kaminski 2003; Bishop and Vrtiska 2008). There is likely a spatial bias associated with these estimates as vegetative communities and seed production can vary across regions resulting from length of growing season at different latitudes (Olmstead et al. 2013), annual precipitation (Casanova and Brock 2000), soil type (Reed 1988), and interactions with exotic or endemic species (Houlahan and Findlay 2004). In addition, seed density estimates developed during autumn would likely overestimate actual seed availability

during spring, resulting from decomposition and depletion by autumn migrants (Brasher et al. 2007, Hagy and Kaminski 2012; Williams et al. 2014). Drahota and Reichart (2015) assessed seed density at the most productive public wetlands in the RWB and reported seed density to be as much as four times greater than current estimates used for conservation planning in the RWB. Although these estimates are useful for assessing seed density in actively managed public wetlands, they may not represent of seed densities at private easements and cropped wetlands, and use of public wetland seed density estimates in bioenergetics models may bias estimates of total energy availability within the region (Evans-Peters et al. 2012; Olmstead et al. 2013).

Food density estimates and associated TME currently used by the RWBJV for developing spring bioenergetics models focus exclusively on seed density estimates (Bishop and Vrtiska 2008; Nugent et al. 2015). Although wetland plant seeds provide the greatest caloric benefit and are consistently the most consumed wetland food resource by dabbling ducks during spring migration (Pearse et al. 2011; Tidwell et al. 2013), aquatic invertebrates should also be considered as an essential food resource given their desirable protein value (Anderson and Smith 1998; Tidwell et al. 2013). Spring diet studies have estimated aggregate invertebrate biomass in esophagi to range from 2.2–35% for mallards (*Anas platyrhynchos*; LaGrange 1985; Hitchcock 2008; Tidwell et al. 2013), 36–58% for blue-winged teal (*A. discors*; Hitchcock 2008; Tidwell et al. 2013), and 5% for Northern pintails (*A. acuta*; Pearse et al. 2011). Although the exact timing of when dabbling ducks shift diets from high-calorie to high-protein food sources is unclear, and is likely influenced by a number of factors (e.g., food availability, use of exogenous nutrients in egg formation, annual weather), the shift is thought to be a requirement for optimal reproduction to occur (Swanson et al. 1985; Euliss and Harris 1987; Ankney and Alisauskas 1991).

Quality food resources are potentially limited during spring migration (Brasher et al. 2007; Straub et al. 2012), and given the reduction in wetland habitat in the RWB, it is critical that resource managers accurately determine the regions energetic carrying capacity for spring migrating dabbling ducks. Therefore, I conducted a regional study to: 1) precisely estimate plant seed density and associated TME at public, WRP, and cropped wetlands, 2) precisely estimate aquatic invertebrate density and associated energetic contributions, and 3) conduct an energetic assessment to evaluate the potential annual caloric value of available habitats in the RWB, and determine if the energetic demands of spring migrating waterfowl are being met (estimated at 4.4 billion kilocalories).

STUDY AREA

Nebraska's RWB region spans 21 counties in South-Central Nebraska, adjacent south of the Platte River (Figure: 1.1). A majority of wetlands within the region are classified as playas, which are small depressional sink wetlands, lined with a relatively impermeable clay layer, and located at the low spot of closed basin watersheds (Bolen et al. 1989; Smith 2003). Most playas in the RWB are seasonally or ephemerally inundated, with historic hydrologic processes driven by surface runoff following intense precipitation and accumulated snowmelt (Bolen et al. 1989; Smith 2003; Cariveau et al. 2011). Historically, the RWB included >11,000 playas that would have provided approximately 80,000 ha of wetland habitat (Bishop and Vrtiska 2008). However, drainage ditches, irrigation pits, sedimentation, and agricultural expansion resulted in long-term declines of up to 90% of wetland area within the region (Raines et al. 1990; Bishop and Vrtiska 2008; LaGrange 2005). Over 70% of the RWB upland landscape has been converted from mixed prairie grassland to cropped maize, soybean, and wheat (United States Geological Survey 2014), which corresponds to the approximate 70% of RWB wetlands considered to be degraded

by agricultural runoff (Smith 2003). Precipitation preceding spring of 2014 was minimal, and consequently available (i.e., inundated) wetland habitat was relatively limited (National Oceanic and Atmospheric Administration 2017; Table 1.1). Precipitation preceding spring of 2015 was moderate which corresponded to more frequently observed wetland inundation frequencies. (National Oceanic and Atmospheric Administration 2017; Table 1.1).

METHODS

Study Site Selection

Study sites were located on public lands managed by Nebraska's Game and Parks Commission and the U.S. Fish and Wildlife Service, on private conservation easement lands enrolled in the WRP, and on private lands managed for agricultural crop production. Private lands managed for crop production will pond water following intense precipitation events in modified areas where functional playa wetlands historically occurred. Although moist-soil plant species are not promoted in cropped fields, low densities of dormant seeds in remnant seed banks germinate and mature given appropriate conditions (hydric soil) during the growing season. Albeit highly modified, hereinafter, I classify private lands managed for crop production but capable of producing moist-soil plant species when inundated as cropped wetlands. I stratified potential study sites by county, identified the six counties containing the greatest number of inundated public wetlands (where inundated area \geq 1ha; Nebraska Game and Park Spring Habitat Conditions Summary 2014 and 2015), and randomly selected individual wetlands within these counties (Stafford et al. 2006; Tapp and Webb 2015). In 2014, I selected 12 public, 10 WRP, and 10 cropped wetlands from Phelps, Clay, Fillmore, York, Seward, and Custer counties. In 2015, I selected 12 public, 11 WRP, 10 cropped, and two idle (idle wetlands were not farmed, enrolled in WRP, or publically managed) wetlands located in Phelps, Clay, Fillmore, York,

Seward, and Hamilton counties. Inclusion of WRP and cropped wetlands was limited by minimal inundation (i.e., drought) and landowner cooperation during both years. When I began my field season in 2014, all public sites ponded water, however only three of the selected WRP and three cropped sites were inundated. When I began my field season in 2015, all public and WRP sites ponded water; however, no cropped sites ponded water. For all study sites, I obtained information from land managers on management strategies that occurred during the previous three years, including grazing, mowing/disking, burning, ground water pumping, and idle.

Quantifying Plant Seed Density and Caloric Value

I collected moist-soil seed and waste grains during two sampling events each spring. Initial seed samples were collected at all sites prior to arrival of spring migrating waterfowl (mid-February through the first week of March), regardless of inundation status, and during a second event when I considered the majority of spring migrants had departed the region (last two weeks of April; Drahota and Reichart 2015). I collected 10-20 soil cores (dependent on wetland area) during each sampling event at random locations along a transect located across the widest portion of each wetland (Figure 1.2; Greer et al. 2007; Olmstead et al. 2013; Behney et al. 2014). For sites not inundated at the time of seed collection, I used aerial imagery from previous years of above average precipitation to identify areas where ponding frequently occurred and established sampling transects accordingly. Soil samples were 10cm in diameter (78.54cm²) and removed from the top 5cm of substrate (Evans-Peters et al. 2012; Olmstead et al. 2013). All samples were collected at random locations along the established transect where water depths were < 30cm (preferred range of foraging depths for dabbling ducks; Fredrickson 1991) and where vegetative cover was < 50%. Core samples were stored at -10°C and transported to the University of Missouri for processing.

Soil cores were thawed at room temperature and deflocculated in a hydrogen peroxide and baking soda solution (Hagy and Kaminski 2012). I rinsed soil samples through a series of two graduated sieves (#10[250 μ m], and #50 [500 μ m]) to remove small clay particles, and isolate remaining materials into coarse and fine samples (Greer et al. 2007). Samples were dried to constant dry-mass at 60° C for 48 hours in a convection oven and weighed to the nearest 0.1mg. Seeds were removed from the coarse and fine debris samples, identified to genus, weighed to the nearest 0.1mg, and converted to density estimates (kg/ha; Kross et al. 2008, Straub et al. 2012). I determined available energy (kcal/ha) by multiplying mean biomass of an individual seed taxa by the corresponding published value of true metabolizable energy (TME; gross energy in kilocalories of food consumed minus the gross energy of excreta; Appendix A). Finally, I estimated TME across all plant genera present at a wetland by summing the products of a genus weight (percentage of total biomass for all genera) and its corresponding TME value, and dividing by the sum of weights (Williams et al. 2014);

$$\bar{x} = \frac{\sum_{i=1}^n w_i * x_i}{\sum_{i=1}^n w_i}$$

where,

\bar{x} = The weighted average of TME (kcal/kg) across all genera present at a single wetland

w_i = The weight or percentage of a genus seed biomass contributed to the overall seed biomass of all genera

x_i = The TME value of an individual species/genus

Quantifying Aquatic Invertebrate Density and Caloric Value

I assessed aquatic invertebrate density in alternating weeks at 17 wetlands in 2014 and 24 wetlands in 2015. At each wetland, I established 3-5 (dependent on wetland area) randomly located 3m x 3m sample plots at water depths < 30cm (preferred range of foraging depths for

dabbling ducks; Fredrickson 1991) and where vegetative cover was < 50% (Figure 1.3). Within each plot, I collected two nektonic samples using a 500 μm rectangular sweep net (Murkin et al. 1994; Tapp and Webb 2015). The net was lowered vertically into the water column, pressed firmly against the substrate, and bounced through the water column for a distance of 1.1 m (0.5m^2). Four passes of the net were completed for each sample collected (Klemm et al. 1990; Davis and Bidwell 2008). Within each plot, I also collected two benthic samples using a 10-cm diameter x 5-cm deep benthic core sampler (Swanson 1983; Tapp and Webb 2015) in an undisturbed area adjacent to the corresponding nektonic sampling location. Benthic and nektonic samples were preserved in 70% ethanol to prevent deterioration (Murkin and Kadlec 1986a) and shipped to the University of Missouri for processing.

To increase efficiency in invertebrate sorting and identification, I stained samples with rose bengal for 24 hours prior to sorting (Sherfy et al. 2000; Tapp and Webb 2015). I rinsed invertebrate samples through a series of two graduated sieves (#50 [500 μm] and #10[250 μm]) to remove small clay particles and partition remaining materials into coarse and fine samples (Tapp and Webb 2015). I used a Folsom wheel sample splitter (Aquatic Research Instruments) to subsample (1/4 volumes) material retained by the 250 μm and 500 μm sieve (Meyer et al. 2011; Whiting et al. 2011; Tapp and Webb 2015). Aquatic invertebrates were removed from remaining debris, identified to the lowest taxonomic level practical, measured to the nearest millimeter, and catalogued (Kaminski and Prince 1981; Straub et al 2012). Biomass estimates for individual taxa were obtained using published dry length-mass regressions (Duffy and LaBar 1994; Benke et al. 1999; Appendix B). When a length-mass regression was not available for specific taxa, I used estimates from similar species observed in comparable habitats (Benke and Huryn 2006). I was unable to identify published length-mass regression models for taxon similar

to phylum Nematoda and class Hirudinea. Further, I was unable to obtain published length-mass regression models for taxa within the order Gastropoda that did not include shell mass. Using a subset of specimens collected from my study, I developed dry length-mass regressions for Nematoda and Hirudinea, and length-mass regressions that excluded the shells of families Planorbidae, Physidae, and Bithyniidae following methods by Benke et al. 1999 (Appendix C). Biomass estimates were pooled and averaged for all taxa collected at a wetland for each sampling event and converted to biomass density estimates (kg/ha). I determined available energy (kcal/ha) by multiplying the mean biomass of all invertebrate taxa by a factor of 580kcal/kg (Hohman et al. 1996, Straub et al 2012).

Statistical Analysis

Previous studies have suggested that ~16% of seed biomass is lost during processing (destroyed during sieving or not identified while sorting), thus I corrected for potential seed loss in this study by multiplying all raw density measurements by 1.16 (Hagy et al. 2011; Drahota and Reichart 2015). I used a two sample paired t-test to determine if seed depletion occurred at inundated wetlands by comparing estimates of pre-migration seed density with post migration seed density. To determine the percentage of wetlands that offered limited food availability to spring migrants, I performed a one-sided t-test to evaluate whether mean seed density exceeded a foraging threshold of 200kg/ha. Foraging thresholds occur when food density falls below a level at which time the energetic cost of foraging exceeds the energetic benefit (Reinecke et al. 1989; Hagy and Kaminski 2012; Williams et al. 2014). Although there has been debate as to what an appropriate foraging threshold should be for dabbling ducks foraging on moist-soil seeds, 200kg/ha is regarded as a relatively conservative threshold (Hagy and Kaminski 2012; Williams et al. 2014).

I compared plant seed density, TME, and energy produced for each wetland type using measures of central tendency (e.g., median), as each metric followed a non-normal distribution, and there is considerable uncertainty whether spring-migrating waterfowl distribute themselves in an ideal free manner (Straub et al. 2012). I used a non-parametric Kruskal-Wallis H test in program R (R Development Core Team 2017) to test for differences in food resource availability between wetland types (Kruskal and Wallis 1952). Proceeding each Kruskal-Wallis H test, I tested data for all general Kruskal-Wallis H test assumptions including homogeneity of distributions. Raw distributions were not homologous (Lavene 1960; Fox et al. 2016) and were thus log transformed. Following a statistically significant result, I conducted Dunn's multiple comparison test to assess differences between independent wetland types (Dinno 2015).

Total potential area of wetland habitat in RWB for public, WRP, and cropped wetlands is approximately 11,582ha, 2,117ha, and 65,874ha respectively, however actual wetland area available for waterfowl use is dependent on a wetland becoming inundated (RWBJV 2013; Bishop et al. 2016; Table 2.1). I corrected for actual wetland area available for duck use by multiplying potential wetland area by an inundation frequency factor specific to each wetland type (RWBJV 2013; Bishop et al., 2016; Table 2.1). To determine the overall caloric contribution of each wetland type, I multiplied inundated wetland area by the corresponding median caloric estimate (kcal/ha). Finally, spring carrying capacity was estimated by summing available energy for each wetland type.

RESULTS

I collected and processed 1,202 soil core samples from 24 public (Appendix J), 21 WRP (Appendix K), and 20 cropped wetlands (Appendix L), respectively, during springs 2014 and 2015. I identified seeds from 42 native, exotic, and agricultural plant genera. *Polygonum spp.*,

Echinochloa spp, *Sparganium spp*, *Potamogeton spp*, and *Scirpus spp* were the most common moist-soil plant genera observed, and accounted for 73% of the total native seed density during both years. *Glycine spp* and *Zea spp* were the most common agriculture plant genera observed and accounted for 97% of the total agricultural seed density observed during both years. Pre and post-migration seed density estimates were not statistically different in 2014 or 2015 ($t_{17, 0.05} = 1.22$ and $t_{24, 0.05} = 1.57$), indicating minimal seed depletion.

Seed density estimates (kg/ha) did not differ among the three wetland types in 2014 ($H_{2,30,0.05} = 5.76$; Table 2.2). Wetland-specific estimates of mean seed density in 2014 were greater than the 200kg/ha threshold at 83% of public sites, 90% of WRP sites, and 70% of cropped sites (Figure 2.1). Among wetlands sampled in 2014, mean seed density was significantly greater than 200kg/ha at public ($t_{11,0.05} = 3.32$) and WRP wetlands ($t_{9,0.05} = 2.97$), but did not exceed the 200kg/ha threshold at cropped wetlands ($t_{9,0.05} = 1.48$). Mean ranks of TME (kcal/kg) for seeds collected in 2014 varied among wetland types ($H_{2,30,0.05} = 7.38$; Table 2.2). Overall TME was statistically greater at cropped (median = 2405kcal/kg) than public wetlands (median = 1750kcal/kg; $Z_{21,0.05} = 2.64$), but TME did not differ among other pairwise comparisons ($p > 0.05$; Figure 2.2). Energy produced by seeds collected in 2014 did not vary among wetland types ($H_{2,30,0.05} = 2.68$; Figure: 2.3).

Seed density estimates did not differ among the three wetland types in 2015 ($H_{2,31,0.05} = 1.60$ (Table 2.3). Wetland-specific estimates of mean seed density in 2015 were greater than the 200kg/ha threshold at 100% of public sites, 90% cropped sites, and 91% WRP sites (Figure 2.4). Overall mean seed density exceeded the 200kg/ha threshold at all wetland types ($p < 0.05$). Mean ranks of TME for seeds collected in 2015 varied among wetland types ($H_{2,31,0.05} = 11.83$; Table 2.3). Overall TME was greater at cropped wetlands (median = 2587kcal/kg) than public

(median = 1714kcal/kg: $Z_{21,0.05} = 3.23$) and WRP wetlands (median = 1804kcal/kg: $Z_{20,0.05} = 2.73$), however TME did not differ between WRP and public wetlands ($Z_{22,0.05} = 0.46$; Figure 2.5). Energy produced by seeds collected in 2015 did not vary among wetland types ($H_{2,31,0.05} = 2.80$; Figure: 2.6).

I collected and processed 1,524 invertebrate samples from 41 wetlands during springs 2014 and 2015. Given the limited amount of inundated habitat at WRP and cropped wetlands, I chose to combine all wetland types when summarizing aquatic invertebrates. I identified 45 invertebrate families from nektonic and benthic communities (Appendices H and I). Among the most frequently observed taxonomic orders during both years were *Diptera*, *Gastropoda*, *Odonata*, *Copepoda*, and *Anostraca*. During both years, invertebrate density (kg/ha) was greater in benthic communities, however taxonomic richness was greater in nektonic communities. Combined benthic and nektonic invertebrate density also varied substantially between years, with mean biomass of 10.26kg/ha (SE = 2.09) in 2014, and 16.17kg/ha (SE = 1.09) in 2015 (Table 2.4). Plant seed density estimates exceeded aquatic invertebrate density at all sites (n=41) in 2014 and 2015. Assuming a TME value 580kcal/kg (Hohman et al. 1996), I estimated the caloric value of aquatic invertebrates at playa wetlands in the RWB to be 7,958kcal/ha.

From 2006-2015, I estimated annual mean energy at inundated wetlands in the RWB to be 1.2 billion kcal (Table 2.1). During the 10-year period, mean annual energy produced varied among wetland types ($F_{2,27,0.05}=3.35$). Inundated area was greatest for public wetlands, and consequently public wetlands generated the greatest caloric contribution to carrying capacity in the RWB. Although inundated area was greater at cropped wetlands than WRP wetlands, there was no difference in available caloric production between the two wetland types ($t_{18,0.05} = 0.08$).

Based on these estimates, the RWBJVs annual objective for providing 4.4 billion kilocalories to spring migratory waterfowl was achieved in only one year during the 10-year period.

DISCUSSION

Food resources are considered limited during spring migration, resulting from seed depletion by fall migrants and decomposition (Nelms and Twedt 1996; Barney 2008; Foster et al. 2010). However, my estimates for spring food density estimates were similar to seed densities reported by studies conducted prior and/or during autumn migration (Table 2.5; Kross et al. 2008; Evan-Peters et al. 2012; Olmstead et al. 2013). Further, over 90% of the public and WRP wetlands sampled in 2014 and 2015 had seed densities exceeding 200kg/ha, suggesting that the vast majority of wetlands in the RWB offer beneficial spring foraging opportunities (Hagy and Kaminski 2012; Williams et al. 2014). Although wetland food availability in the RWB appears to be comparable to other regions (see also Drahotka and Reichart 2015), my estimates may not be applicable for management planning in other regions. Straub et al. (2012) sampled spring wetlands in the Upper Mississippi River and Great Lakes Region, and reported mean seed density as 208kg/ha in palustrine emergent systems, 87kg/ha in palustrine forested systems, and only 52kg/ha at lacustrine-riverine systems. Brasher et al. (2007) reported even lower spring estimates for actively and passively managed wetlands in Central and Northern Ohio to be 14kg/ha, following an 80% reduction by fall migrants.

The Wetlands Reserve Program appears to be a viable conservation strategy for increasing suitable waterfowl foraging habitat in the RWB. I estimated overall mean seed density at WRP wetlands to be 566kg/ha, which was similar to seed estimates for public wetlands (621kg/ha). In addition to quantity, the nutritional quality of seeds in terms of TME was also comparable between the two wetland types (Table 2.6). Active management techniques used at WRP

wetlands in the RWB could explain these similar seed density estimates at the two wetland types. Vegetative disturbance techniques such as prescribed burning, disking, and mowing are commonly associated with moist-soil management, with the objective of maintaining plant communities in early-successional stages (Fredrickson and Taylor 1982). The cost and labor required to implement these disturbance techniques are a concern for landowners with no perpetual obligation to manage dabbling duck foraging habitats on WRP easements (King et al. 2006; Tapp and Webb 2015). Consecutive years of passive or no management can promote dense stands of late-successional plant species that often produce fewer seeds and are associated with less desirable TME values (Fredrickson and Taylor 1982; Olmstead et al. 2013; Appendix A). Conversely, WRP enrollees in the RWB also have the option of cattle grazing during certain months of the year, which mimics a natural vegetative disturbance (Stutheit et al. 2004; LaGrange 2005; Marty 2005). This is a unique selling point for landowners in the RWB with livestock (Personal communication with Nathan Walker, United State Department of Agriculture). Among the private landowners who permitted me access to their WRP sites, approximately 80% used cattle grazing as a management tool to control persistent emergent vegetation and promote early successional plant communities.

Although less common and still in the beginning stages of implementation, the Agricultural Land Easement (ALE) option within the Agricultural Conservation Easement Program could serve as an alternative to, or complement the WRP (Nugent et al. 2015). I did not collect samples from ALEs; however, seed density estimates derived from cropped wetlands are likely equivalent and may offer some guidance when assessing the program's potential for meeting waterfowl foraging habitat goals. Site-specific seed density at cropped wetlands was highly variable, ranging from 23-2680kg/ha over both years (Table 2.6). The majority of this

variation was explained by harvest during the preceding fall. Over both years, mean seed density at harvested sites (n=15) was 480kg/ha, compared to 991kg/ha at sites (n=5) that went unharvested during the preceding fall (Table 2.7). In addition to biomass, the caloric value of waste grains commonly produced in the RWB (*Zea mays* and *Glycine max*) were twice that of annual smartweed (*Polygonum spp.*), which was the most frequently observed native seed in this study (Appendices J-L). Despite the caloric benefit waste grains offer, they fail to provide dabbling ducks with all nutrients needed to complete life history requirements and maintain optimal body condition (Baldassare et al. 1983; Loesch and Kaminski 1989; Miller et al. 2000). Regardless, estimated mean density for native seeds at cropped wetlands was 414kg/ha (data skewed right, consider median = 333kg/ha), which is far above the proposed foraging threshold of 200kg/ha and indicates that cropped wetlands have potential to provide dabbling ducks with food energy from both waste grains and recruitment moist-soil seeds.

Given their relatively low abundance and patchy distributions, it is a challenge for wetland and waterfowl biologists to identify appropriate target densities for aquatic invertebrates (Murkin and Kadlec 1986a; Drahota and Reichart 2015; Stafford et al. 2016). My estimates for mean aquatic invertebrate density were approximately 40-fold less than estimates for mean seed density (Table 2.4). Davis and Bidwell (2008) also conducted a spring aquatic invertebrate assessment in the RWB and estimated total invertebrate density to be 12.70kg/ha at actively managed wetlands, which corresponds to estimates from this study; however, both estimates are considerably lower than those reported by studies outside the RWB (Table 2.8). In an effort to evaluate the role of invertebrates in influencing dabbling duck habitat use, I developed species distribution models for explaining dabbling duck densities using several metrics of food resource availability as explanatory variables (see Chapter I). I observed a significant relationship

between dabbling duck density and seed density; however, I did not observe a relationship between dabbling duck density and aquatic invertebrate density. Tapp (2013) observed similar results when assessing dabbling duck abundance in the Lower Mississippi Alluvial Valley during autumn and winter; however, Murkin and Kadlec (1986b) observed a strong positive relationship between dabbling duck density and invertebrate density during late spring/early summer on Manitoba breeding grounds. Despite consistent lack of evidence that invertebrate densities influence dabbling duck densities during the non-breeding season, diet studies have emphasized the importance of invertebrates during spring migration (Hitchcock 2008; Tidwell et al. 2013). To my knowledge, there is no defined benchmark for invertebrate availability for waterfowl management, which would likely vary across space and time. Thus, managers should recognize best management practices that promote both seed and invertebrate production to ensure sufficient availability of all food resources consumed by waterfowl and other wetland-dependent species (Tidwell et al. 2013; Stafford et al. 2016).

The amount of energy available to waterfowl at playa wetlands in the RWB is dependent on seed production, TME, and inundated wetland area (RWBJV 2013; Nugent et al. 2015). At the landscape scale, accurate estimates for predicting inundation frequency specific to the RWB during spring are available (Bishop et al. 2016); however, seed density and TME estimates used in energetic assessments are currently based on conditions at actively managed public wetlands in the southern United States during autumn (Haukos and Smith 1993; Kaminski et al. 2003; Bishop and Vrtiska 2008). Based on my results from springs 2014 and 2015, the current RWBJV energetic model underestimates seed density and overestimates TME (Table 2.6). Seed density estimates currently used for energetic modeling are based on three habitat types: early-successional (250kg/ha), late-successional (25kg/ha), and cropped wetlands (100kg/ha; Bishop

and Vrtiska 2008; RWBJV 2013). Most of the vegetative communities observed during my field seasons were dormant, knocked down and submerged, or deceased, making it difficult to differentiate between early and late-successional stands. Thus, I chose to develop general seed density estimates for public, WRP, and cropped wetlands, regardless of vegetative communities present in these wetlands. It is likely that my general estimates for seed density under value early-successional stands and over value late-successional stands, but remain consistent with estimates at cropped wetlands. Regardless of wetland type, seed density estimates developed in this study for public, WRP, and cropped wetlands greatly exceed those currently assumed for highly productive early-successional stands during spring by the RWBJV (Also see Drahota and Reichart 2015).

Mean TME provided by a gram of seed across all moist-soil plant species is generally estimated as 2.47kcal/g (Kaminski 2003; Bowyer et al. 2005; Kross et al. 2008; RWBJV 2013); however, variation in seed composition resulting from vegetative manipulations and geographic location may influence exact estimates (Dugger et al. 2007; Williams et al. 2014). For instance, a late-successional stand dominated by bulrush (*Scirpus spp.*) and spike rush (*Eleocharis spp.*) would likely have an overall TME less than 1.00kcal/g, whereas an early-successional stand of smartweed (*Polygonum spp.*) and barnyard grass (*Echinochloa spp.*) would be associated with an overall TME of ~2.00kcal/g (Appendix A). Further, TME may exceed 3.00kcal/g when assessing agricultural waste grains in cropped wetlands. For public and WRP wetlands assessed in this study, I estimated TME to be significantly less than the 2.47kcal/g ($p < 0.05$; table 2.6). Thus, continuing to use 2.47kcal/g when modeling energetics in the RWB will significantly overestimate actual accessible energy to waterfowl when all other metrics (seed density and inundated area) remain constant.

MANAGEMENT IMPLICATIONS

My spring bioenergetics assessment indicated that available energy to waterfowl in spring wetlands exceeded the RWBJVs energetic objective of 4.4 billion kilocalories in just 10% of the years 2006-2015 (Table 2.1). Based on my observations at moist-soil wetlands in the RWB, seed density does not appear to be limiting this objective from being achieved on a more consistent basis. Although TME was far below expectations in moist-soil units, its potential to increase in the RWB is improbable considering most wetlands already receive management for early-successional plant communities. Increasing inundated area of unharvested cropped fields would be the most efficient method for increasing available energy during spring; however, the utility of this approach may not benefit all wetland-dependent wildlife (Gray et al. 2004; Riens et al. 2013; Casanova and Casanova 2016). Although beyond the scope of this study, developing methods to increase inundation frequency during years with minimal precipitation would likely increase accessible food resources for waterfowl. Despite the potential WRP has in providing waterfowl with optimal foraging opportunities, only 3% and 5% of their wetland area ponded water in 2014 and 2015 (Bishop et al. 2016). In consideration of the forgoing, installing additional groundwater-pumping capabilities at public and WRP wetlands would offset unfavorable conditions during dry years. Additional opportunities to increase natural inundation frequencies at wetlands within closed basin systems (e.g. playas and potholes) will require hydrological modifications beyond the local level (Bishop and Vrtiska 2008). Surface drains and irrigation reuse pits located in the upland watershed often intercept water that would have otherwise runoff into a wetland, and leveed road-fragmentation can disconnect a wetland from its watershed entirely (Stutheit et al. 2004; Bishop and Vrtiska 2008). Regardless of how and where, increasing inundation frequency must become a priority for ephemeral wetlands in arid

regions to insure adequate wetland-derived energy is consistently available to spring migrants (RWBJV 2013; Nugent et al. 2015; Petrie et al. 2016).

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Table 2.1. Wetland habitat summary and energetic production in the Rainwater Basin from 2006-2015. Inundation frequency, potential habitat, and inundated area were all derived from annual habitat surveys conducted by the Rainwater Basin Joint Venture. Available energy at each wetland type was calculated by multiplying inundated area by median energetic density (kcal/ha) for each wetland type from 2014 and 2015. Caloric estimates used to develop model are defined in table 2.6. Wetland type listed as other in table was not assessed in this study.

Inundation Frequency											
Wetland Type	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10yr Avg
Public	0.0520	0.2246	0.1363	0.1404	0.2681	0.0629	0.0589	0.0296	0.0460	0.0617	0.1080
WRP	0.0666	0.3043	0.2282	0.1878	0.3946	0.0583	0.0444	0.0177	0.0296	0.0560	0.1388
Cropped	0.0006	0.0183	0.0071	0.0028	0.0210	0.0009	0.0009	0.0003	0.0001	0.0001	0.0052
Other	0.0051	0.0169	0.0136	0.0114	0.0235	0.0048	0.0057	0.0035	0.0035	0.0050	0.0093

Potential Wetland Area (ha)											
Wetland Type	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10yr Avg
Public	11,634	11,540	11,549	11,476	11,672	11,626	11,631	11,610	11,542	11,534	11,582
WRP	1,712	1,907	1,893	1,985	2,031	2,363	2,325	2,246	2,345	2,363	2,117
Crop/Other	66,169	66,067	66,073	66,055	65,918	65,632	65,665	65,745	65,715	65,704	65,874
<i>Total</i>	79,515	79,515	79,515	79,516	79,622	79,622	79,622	79,601	79,601	79,601	79,573

Inundated Area (ha)											
Wetland Type	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10yr Avg
Public	605	2,591	1,575	1,611	3,130	732	685	343	531	711	1,251
WRP	114	580	432	373	802	138	103	40	69	132	278
Cropped	38	1,206	472	188	1,382	58	61	18	8	4	344
Other	340	1,119	900	750	1,549	318	376	231	227	330	614
<i>Total</i>	1,097	5,497	3,379	2,923	6,863	1,246	1,225	632	836	1,177	2,487

Available Energy (kcal in THOUSANDS)											
Wetland Type	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	10yr Avg
Public	510,314	2,185,006	1,327,580	1,358,528	2,638,816	617,044	577,327	289,585	447,395	599,608	1,055,120
WRP	124,321	634,945	471,068	406,591	874,217	150,315	112,581	43,294	75,775	144,224	303,533
Cropped	32,464	1,017,516	397,865	158,804	1,166,080	48,918	51,342	15,020	7,135	3,243	289,839
Other	-	-	-	-	-	-	-	-	-	-	-
<i>Total</i>	667,098	3,835,467	2,196,512	1,923,922	4,679,111	816,276	741,251	347,899	530,304	747,076	1,648,942

Table 2.2. Estimates of median, mean, standard error (SE), and 95% confidence intervals (CI) for seed density (kg/ha dry mass), TME (kcal/kg), and Energy (kcal/ha) collected in public, WRP (Wetlands Reserve Program) and cropped wetlands during spring 2014.

Metric	<u>Public Wetlands</u>			<u>WRP Wetlands</u>			<u>Cropped Wetlands</u>		
	Density	TME	Energy	Density	TME	Energy	Density	TME	Energy
Median	582	1750	768,020	569	1776	1,112,588	310	2405	632,648
Mean	546	1652	898,015	594	1832	1,013,186	281	2256	613,029
SE	104	78	191,492	133	112	183,466	55	137	127,309
Lower CI	229	1480	476,544	300	1579	598,158	125	1945	325,037
Upper CI	229	1824	1,319,486	300	2085	1,428,214	125	2567	901,021

Table 2.3. Estimates of median, mean, standard error (SE), and 95% confidence intervals (CI) for seed density (kg/ha dry mass), TME (kcal/kg), and Energy (kcal/ha) collected in public, WRP (Wetlands Reserve Program) and cropped wetlands during spring 2015.

Metric	Public Wetlands			WRP Wetlands			Cropped Wetlands		
	Density	TME	Energy	Density	TME	Energy	Density	TME	Energy
Median	651	1714	892,954	560	1804	1,090,553	637	2587	1,413,891
Mean	695	1665	1,192,197	540	1741	964,815	934	2511	2,448,435
SE	93	97	226,274	91	86	195,223	243	207	727,825
Lower CI	491	1452	694,172	337	1549	529,831	384	2042	801,981
Upper CI	899	1878	1,690,222	743	1933	1,399,798	1484	2980	4,094,889

Table 2.4. Estimates of mean, standard error (SE), median, and 95% confidence intervals (CI) for Invertebrate density (kg/ha dry mass) at wetlands in Nebraska's Rainwater Basin during springs 2014 (n=17) and 2015 (n=24).

Metric	<u>2014 Invertebrate Density</u>			<u>2015 Invertebrate Density</u>			<u>Average Density</u>		
	Benthic	Nektonic	Total	Benthic	Nektonic	Total	Benthic	Nektonic	Total
Mean	8.45	1.81	10.26	9.88	6.29	16.17	9.29	4.43	13.72
SE	1.62	0.55	2.09	1.79	0.84	1.99	1.24	0.64	1.50
Median	6.80	0.93	7.69	6.42	5.04	14.39	6.66	3.58	11.22
Lower CI	5.02	0.63	5.84	6.18	4.54	12.06	6.79	3.13	10.68
Upper CI	11.88	2.98	14.68	13.59	8.03	20.28	11.79	5.73	16.76

Table 2.5. Comparison of WRP and public seed density estimates at various geographic study locations in the United States.

Type	Season	Region	Density	Reference
WRP	Spring	Nebraskas Rainwater Basin	569kg/ha	This Study
WRP	Fall	Mississippi Alluvial Valley	528kg/ha	Olmstead et al. 2013
WRP	Fall	Willamette & Columbia River Valley	441kg/ha	Evan-Peters et al. 2012
WRP	Fall/Winter	Lower Mississippi Alluvial Valley	188kg/ha	Tapp 2013
Public	Spring	Nebraska Rainwater Basin	593kg/ha	This Study
Public	Spring	Nebraska Rainwater Basin	686kg/ha	Drahota and Reichart 2015
Public	Fall	Mississippi Alluvial Valley	496kg/ha	Kross et al. 2008
Public	Fall	Willamette & Columbia River Valley	545kg/ha	Evan-Peters et al. 2012
Public	Fall/Winter	Lower Mississippi Alluvial Valley	211kg/ha	Tapp 2013

*Estimates for Tapp (2013) is an average of initial and post seed estimates

Table 2.6. Estimates of median, mean, standard error (SE), and 95% confidence intervals (CI) for seed density (kg/ha dry mass), TME (kcal/kg), and Energy (kcal/ha) collected in public, WRP (Wetlands Reserve Program) and cropped wetlands during springs 2014 and 2015.

Metric	<u>Public Wetlands</u>			<u>WRP Wetlands</u>			<u>Cropped Wetlands</u>		
	Density	TME	Energy	Density	TME	Energy	Density	TME	Energy
Median*	593	1740	843,164	561	1781	1,090,553	419	2431	843,555
Mean	621	1659	1,045,106	566	1784	987,849	608	2384	1,530,732
SE	70	61	148,165	77	69	131,239	142	125	416,684
Lower CI	476	1533	738,604	400	1640	714,089	310	2123	658,602
Upper CI	766	1785	1,351,608	727	1928	1,261,608	906	2645	2,402,863

*Median estimates from this table were used to develop bioenergetic model (see Table 2.1).

Table 2.7. Seed composition observed at cropped wetlands in 2015 where row crops were not harvested during the previous fall (n=5). Crops went unharvested when land managers were unable to operate equipment in hydric soils.

	<u>Waste Grain</u>			<u>Moist-Soil Seed</u>		
	Density	TME	Energy	Density	TME	Energy
Mean	653	3058	1,996,874	339	2045	693,255
Median	497	2650	1,317,050	340	2422	823,480
SE	230	559	128,570	22	313	6,886
Upper CI (95%)	1291	3751	4,843,547	399	2914	1,162,686
Lower CI (95%)	15	2365	34,841	279	1176	328,104

Table 2.8. Comparison of aquatic macroinvertebrate density estimates in various geographic study locations in the United States.

Season	Wetland Type	Location	(kg/ha)	Reference
Spring	Playa Wetlands	Rainwater Basin	14	This Study
	Actively Managed Wetlands	Rainwater Basin	13	Davis and Bidwell 2008
	Passively Managed Wetlands	Rainwater Basin	12	Davis and Bidwell 2009
	Palustrine Emergent Wetland	Upper Mississippi Great Lakes	69	Straub et al. 2012
	Palustrine Forested Wetland	Upper Mississippi Great Lakes	28	Straub et al. 2012
	Lacustrine Wetland	Upper Mississippi Great Lakes	18	Straub et al. 2012
	Rice Field	Louisiana	22	Hohman et al. 1996
Autumn	Moist-Soil Impoundments	Mississippi Alluvial Valley	12	Tapp and Webb 2015
	Actively Managed Wetlands	Great Salt Lake Marshes	40	Johnson 2007
	Passively Managed Wetlands	Great Salt Lake Marshes	74	Johnson 2007
	Actively Managed Wetlands	Central Kansas ¹	2	Kostecke et al. 2005
	Passively Managed Wetlands	Central Kansas ¹	6	Kostecke et al. 2005
Winter	Greentree Reservoir	Mississippi Interior Flatwoods	2	Wehrle et al. 1995
	Clear-cut Areas	Mississippi Interior Flatwoods	2	Wehrle et al. 1995
	Greentree Reservoir	Mississippi Alluvial Valley	10	Wehrle et al. 1995
	Clear-cut Areas	Mississippi Alluvial Valley	3	Wehrle et al. 1995

¹Results only displayed for autumn 1999 when both wetland types were assessed.

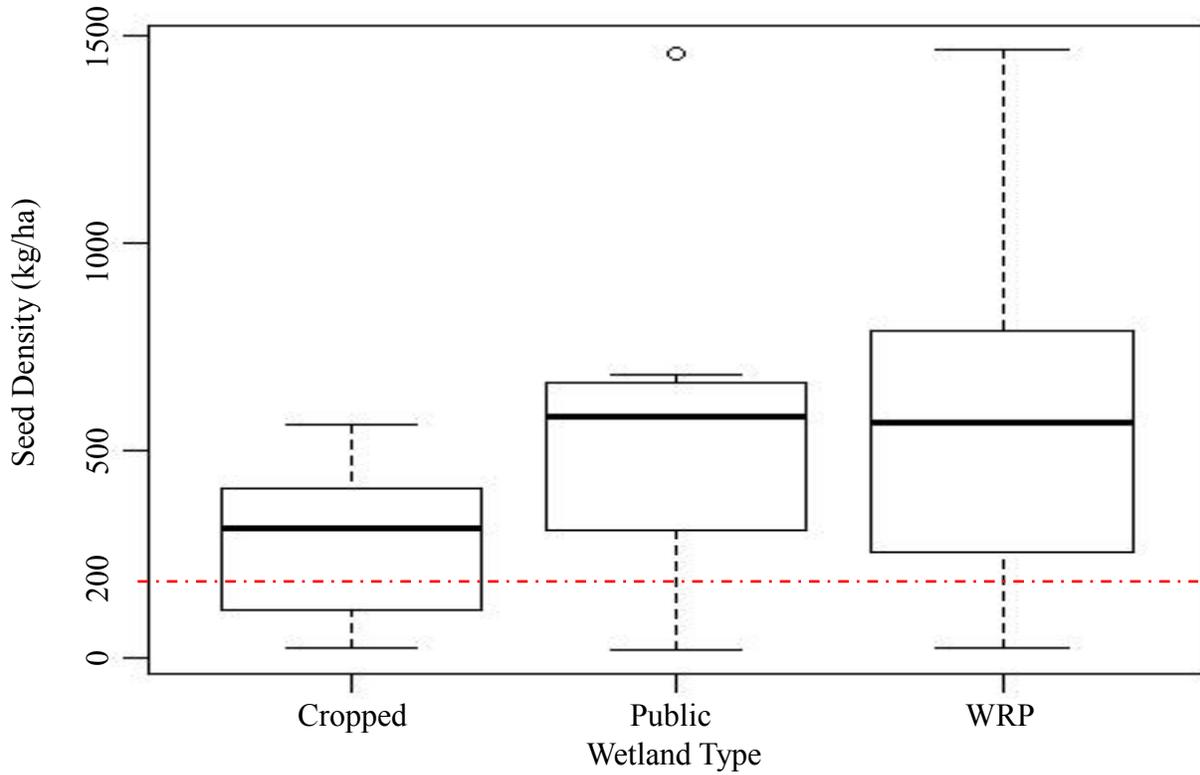


Figure 2.1. Seed density at cropped, public, and WRP wetlands during the spring of 2014. The dashed line (200kg/ha) represents a foraging threshold which recognizes the point at which a wetland would have little to no forage value to dabbling ducks. The foraging threshold was exceeded at 70% of cropped, 83% of public, and 90% of WRP wetlands sampled.

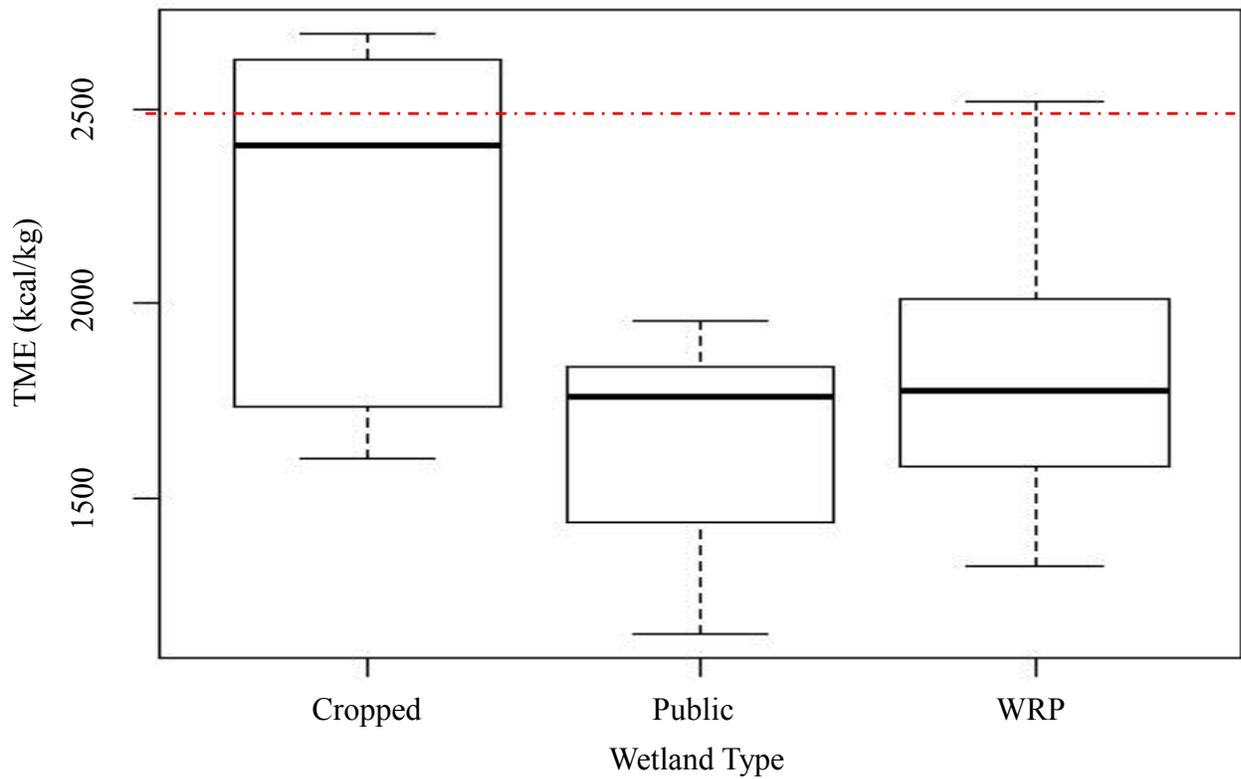


Figure 2.2. True Metabolizable Energy (TME) at cropped, public, and WRP wetlands during the spring of 2014. TME is an average of pre and post migratory seed collections. The dashed line (2470 kcal/kg) represents the current estimate used by the Rainwater Basin Joint Venture for developing bioenergetic models.

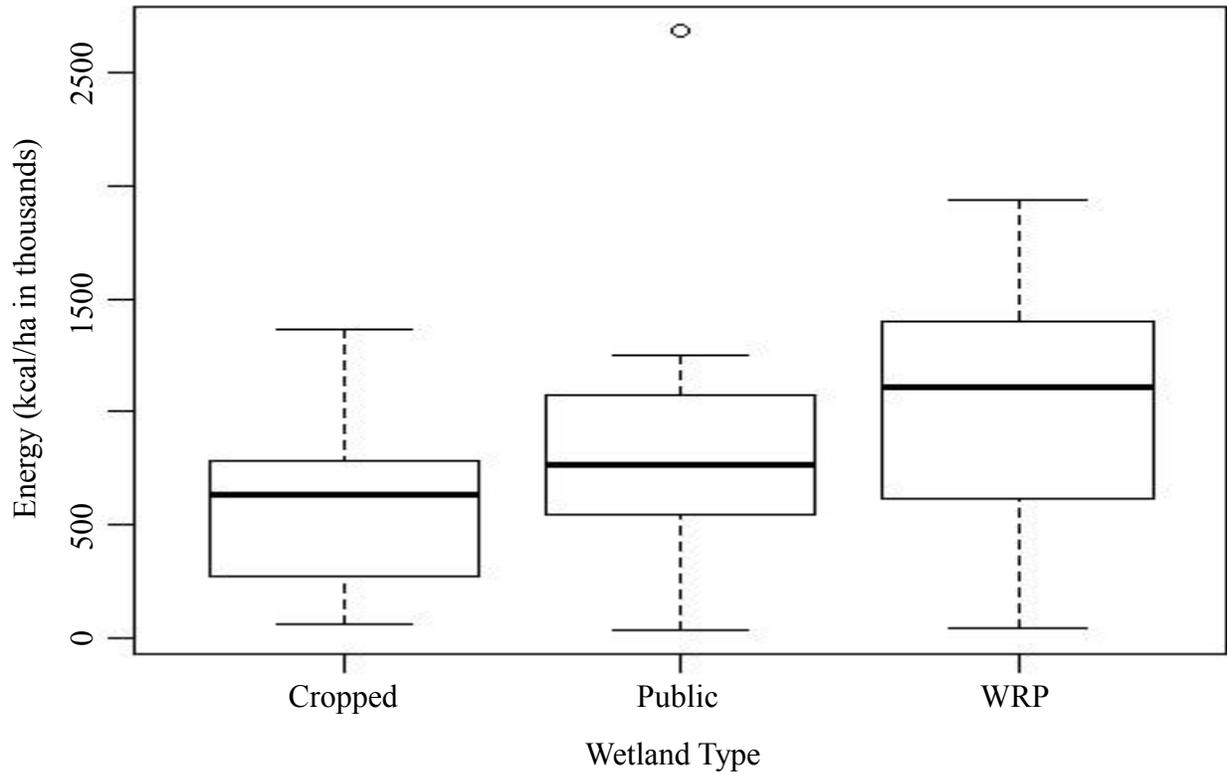


Figure 2.3. Energy produced (kcal/ha) at cropped, public, and WRP wetlands during the spring of 2014.

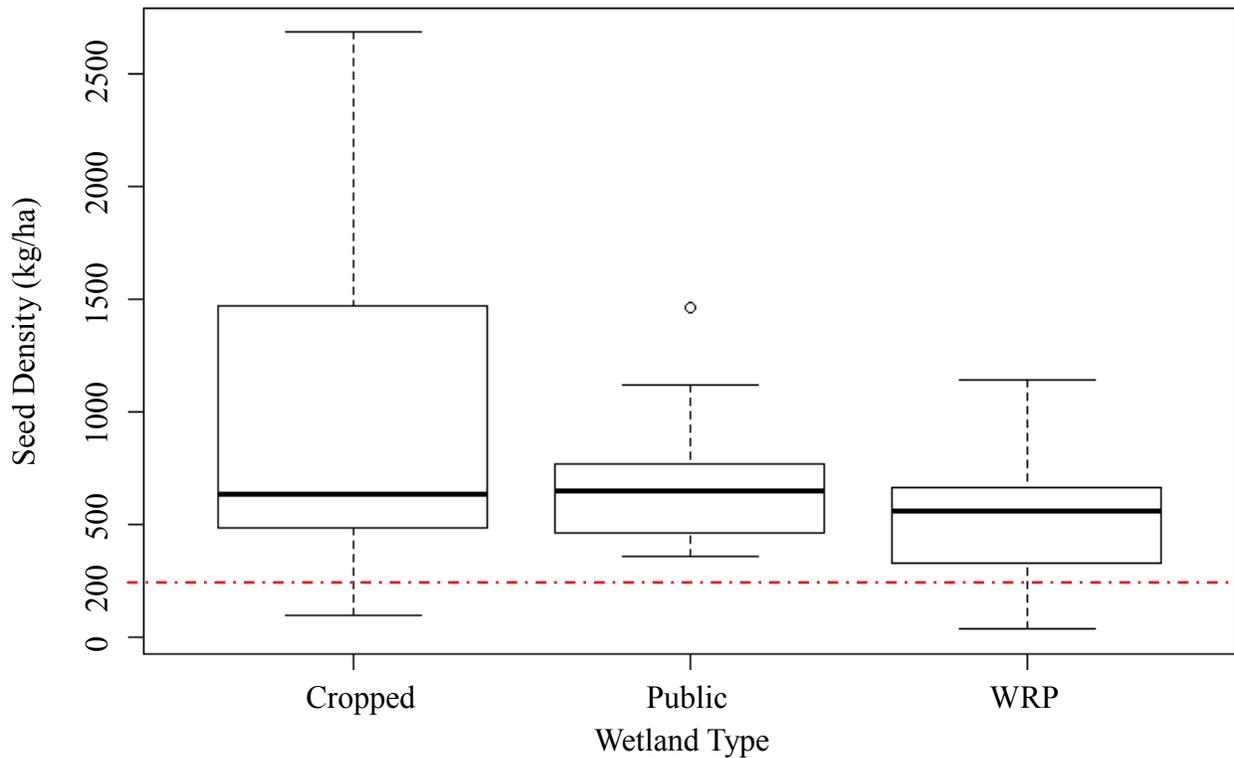


Figure 2.4. Seed density at cropped, public, and WRP wetlands during the spring of 2015. The dashed line (200kg/ha) represents a foraging threshold which recognizes the point at which a wetland would have little to no forage value to dabbling ducks. The foraging threshold was exceeded at 90% of cropped, 100% of public, and 91% of WRP wetlands sampled.

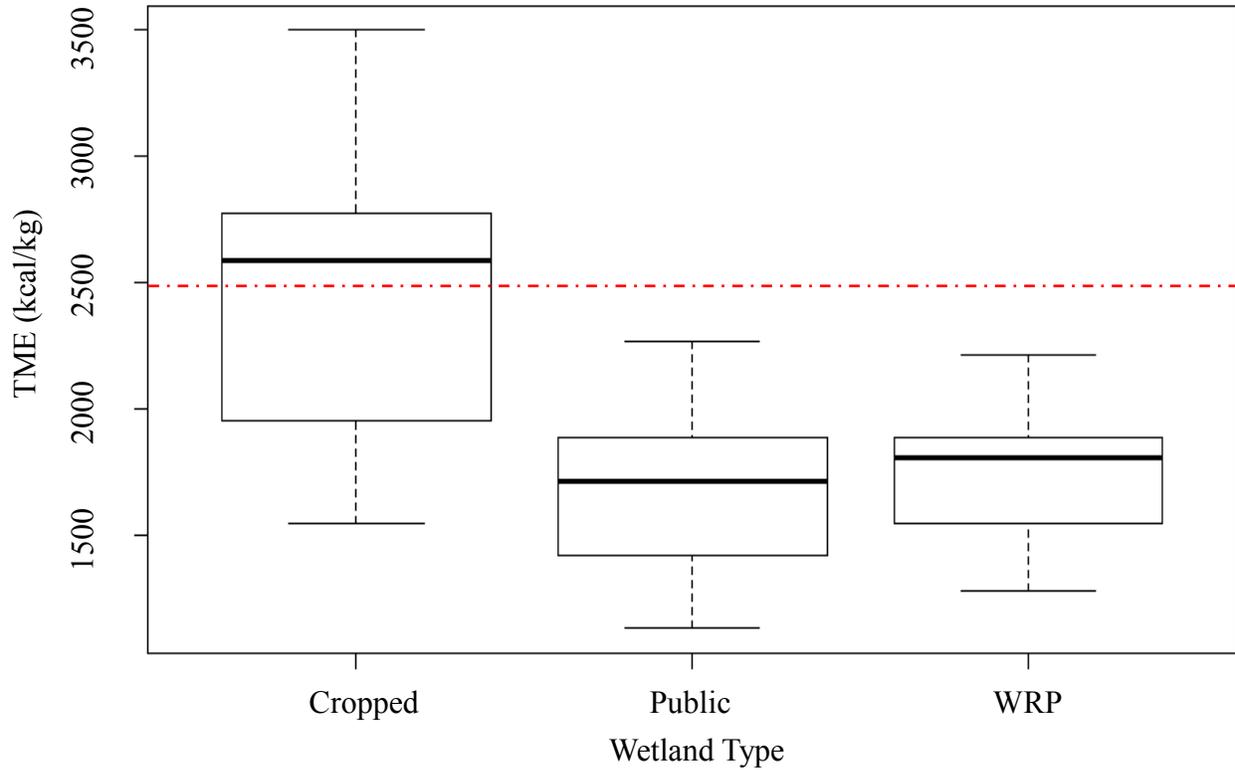


Figure 2.5. True Metabolizable Energy (TME) at cropped, public, and WRP wetlands during the spring of 2015. TME is an average of pre and post migratory seed collections. The dashed line (2470 kcal/kg) represents the current estimate used by the Rainwater Basin Joint Venture for developing bioenergetic models.

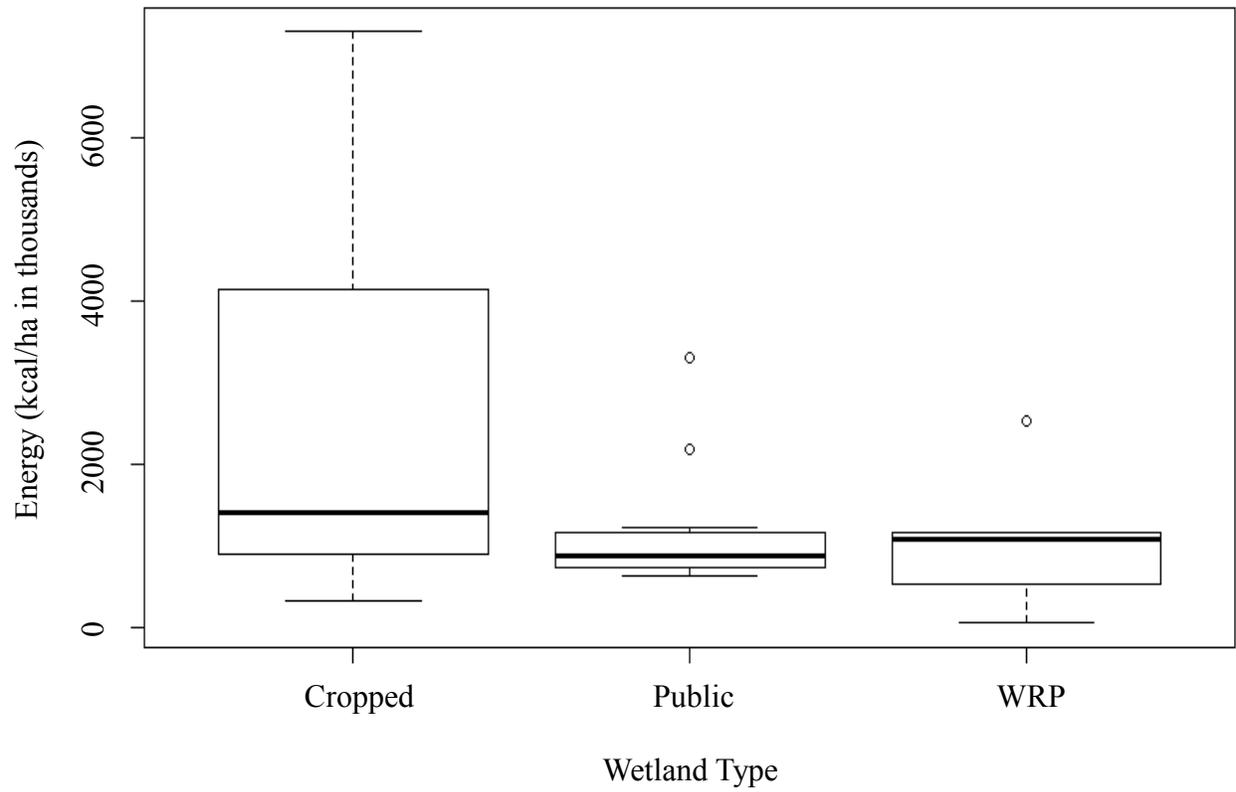


Figure 2.6. Energy produced (kcal/ha) at cropped, public, and WRP wetlands during the spring of 2015.

CHAPTER III: NEONICOTINOIDS IN AGRICULTURAL LANDSCAPES: DETECTION FREQUENCY, MITIGATION STRATEGIES, AND EFFECTS ON AQUATIC INVERTEBRATE COMMUNITIES

Persistence of healthy and sustainable aquatic invertebrate communities depends in large part on the abiotic and biotic characteristics present in aquatic systems, especially in wetlands (Cairns and Pratt 1993; Davis and Bidwell 2008; Riens et al. 2013). Although aquatic invertebrates are not generally the main focus of wetland management practices, they provide an essential link for energy flow between primary producers and most vertebrate wetland-dependent taxa, including fish, amphibians, and waterbirds (Covich et al. 1999; Batzer and Boix 2016). In addition to channeling energy into upper trophic levels, benthic invertebrates provide essential ecosystem services by accelerating rates of decomposition for detrital and dead organic matter (Covich et al. 1999). These ecosystem processes redistribute bound nutrients to the water column, which can then be absorbed by bacteria, fungi, algae, and aquatic angiosperms (Wallace and Webster 1996; Covich et al. 1999; Kalff 2002). The broad range of sensitivity to contaminants and sedimentation exhibited by aquatic macroinvertebrate communities means they also serve as excellent bioindicators for assessing wetland ecosystem health (Cairns and Pratt 1993; Spieles and Mitsch 2000; Reins et al. 2013).

Components of agricultural runoff from cropped fields thought to negatively impact aquatic invertebrate communities include phosphorous and nitrogen that can increase nutrient loading and lead to reductions in dissolved oxygen, sedimentation that stunts invertebrate reproduction by smothering eggs, and synthetic pesticides (Kreutzweiser 1997; Cuffney et al. 2000; Kalff 2002; Frost et al. 2009). Neonicotinoids are currently among the most widely used class of synthetic insecticides, and projected to increase in agricultural application during the coming years (Jeschke and Nauen 2008; Jeschke et al. 2010; USGS National Synthesis Project

2014). Rapid increase in neonicotinoid use has been attributed to versatility of application, relatively low risk to non-target vertebrate organisms such as mammals, birds, and fish, and superior effectiveness in pest management (Jeschke and Nauen 2008; Jeschke et al. 2010). The intended targets for most neonicotinoid applications include aphids and other agricultural pest insects; however, concerns have developed that neonicotinoids might also affect non-target invertebrate taxa, and wildlife dependent on these invertebrate taxa as a food source (Hallmann et al. 2014; Gibbons et al. 2014; Stafford et al. 2016). In recent years, neonicotinoids in surface waters adjacent to agricultural areas were detected in concentrations exceeding the lethal concentration required to kill 50% of the population (LC50) for several common aquatic invertebrate species (Van Dijk et al. 2013; Hladik et al. 2014; Main et al. 2014). Several hypotheses have been proposed for evaluation of aquatic community exposure and response to neonicotinoids (Goulson 2014; Hallmann et al. 2014; Morrissey et al. 2015); however, many of these results were obtained from laboratory settings and a simultaneous assessment of aquatic invertebrate communities and neonicotinoid concentrations in a field setting is needed.

Approximately 75% of the historic land cover in Nebraska's Rainwater Basin (RWB) has been converted to agricultural land use, including maize, soybean, and other agricultural management regimes (Figure 3.1; USGS National Synthesis Project 2014), which corresponds to the approximate 75% of wetlands in the region considered to be impacted by agricultural runoff (Smith 2003). Wetlands in the RWB are classified as playas, which are ephemeral, closed basin systems whose hydrology is primarily driven by precipitation and surface runoff (Smith 2003). Most playa wetlands do not receive ground water inflow; therefore, the chemical, physical, and biological integrity of playas is influenced predominately by upland land use (Belden et al. 2012). Thus, water-soluble contaminants and sediments transported into playas are more likely

to persist as water subsides and accumulates between hydroperiods (Belden et al. 2012). Accumulation of neonicotinoids in RWB wetlands is potentially a matter of conservation concern due to the insecticides extensive half-lives ranging from 200 to >1000 days in soil, and two to 420 days in water for compounds commonly used in pest management (Rexrode et al. 2003; Goulson 2013; Morrissey et al. 2015).

Runoff and overflow from agricultural fields within a watershed drains into some playas directly (point source) via drainage ditches and culverts (non-buffered), whereas other wetlands receive nonpoint source runoff after it has been filtered through buffers of native vegetation surrounding the wetland (Anderson et al. 2013; Riens et al. 2013; Rainwater Basin Joint Venture 2013). Riens et al. (2013) reported that vegetative wetland buffers could intercept sediments bound with nitrogen, phosphorous, chlorophyll α , and atrazine, thereby improving water quality within a wetland. In 2000, Prince Edward Island passed legislation that required wetlands contiguous to agricultural fields be buffered with 10m to 30m vegetation strips. Dunn et al. (2011) assessed the effectiveness of the Prince Edward Island initiative, and found that 10m and 30m buffers removed 52% and 78% of the pesticides analyzed, respectively, however neonicotinoids were not evaluated. Currently, limited information is available on the potential of vegetative buffers for mitigating neonicotinoid concentrations in wetlands (Main et al. 2017). Previous studies in the RWB have evaluated the effectiveness of buffers in reducing sediments and pesticides other than neonicotinoids and suggested that vegetative buffer strips surrounding wetlands would be an appropriate management strategy to improve water quality issues associated with agricultural runoff (Dabney et al. 2006; Rainwater Basin Joint Venture 2013; Riens et al. 2013).

Although several studies have identified the impacts of neonicotinoids on aquatic invertebrates in controlled lab studies, few studies have evaluated the relationship between neonicotinoids and invertebrate communities under actual field conditions. Therefore, the primary objectives of this study were to: (1) quantify water concentrations of neonicotinoid pesticides at playa wetlands in the RWB and compare values with recently established toxicity benchmarks, (2) determine the effectiveness of vegetative buffers for reducing neonicotinoids in agricultural runoff water, and (3) simultaneously compare aquatic macroinvertebrate assemblages with local water quality parameters and land cover in the surrounding landscape.

STUDY AREA

Nebraska's Rainwater Basin (RWB) spans across 21 counties in South Central Nebraska, positioned south to the Platte River (Figure 1.1). Wetlands within the RWB are classified as playas and characterized as small depressional sinks, lined with a relatively impermeable clay layer positioned at the low spot of a closed basin watershed (Bolen et al. 1989; Smith 2003). Most RWB playas are seasonally or ephemerally inundated, with historic hydrologic processes driven by surface runoff following intense precipitation and accumulated snowmelt (Bolen et al. 1989; Smith 2003; Cariveau et al. 2011). Historically the RWB contained >11,000 playas and approximately 80,000ha of wetland habitat (McMurtrey et al. 1972). However, drainage ditches, concentration pits, sedimentation, and agricultural expansion within the region resulted in long-term declines in wetland area by as much as 90% (Raines 1990; LaGrange 2005). Nevertheless, playa wetlands in the RWB provide habitat to wetland-dependent invertebrates, amphibians, fish, and waterbirds, and deliver water quality services in the form of flood storage, nutrient retention, and sediment trapping (Gersib et al. 1992; Mollhoff 2001; Smith 2003; LaGrange 2005).

METHODS

Wetland Selection

Study sites were located on public lands managed by the Nebraska Game and Parks Commission and the U.S. Fish and Wildlife Service, private lands enrolled in the Wetlands Reserve Program, and agricultural wetlands embedded in cropped fields (Figure 1.1). I stratified potential study sites by county and identified the six counties containing the greatest number of inundated wetlands (with inundated area \geq 1ha) and randomly selected individual wetlands within these counties (Stafford et al. 2006; Tapp and Webb 2015). During the spring of 2015, I selected 12 public and 14 private wetlands throughout Phelps, Clay, Fillmore, York, Seward, and Hamilton counties. Precipitation preceding and during spring 2015 was minimal, and consequently, inundated wetland habitat was relatively scarce (National Oceanic and Atmospheric Administration 2017; Table 1.1). Four of the selected private sites became dry halfway through the study and were therefore excluded from the aquatic invertebrate component of my analysis. I obtained information from land managers on management strategies that occurred on study sites during the previous three years. Specific management techniques included grazing, mowing/disking, prescribed burning, ground water pumping, and idle.

Local Variables

I collected composite one-liter water samples during two sampling events (March and April 2015; prior to planting) at each wetland to quantify concentrations of neonicotinoid insecticides, chlorophyll *a*, total phosphorous, total nitrogen, and turbidity. Neonicotinoids analyzed in this study were those commonly used for crop production in the RWB (Water Science Laboratory at the University of Nebraska, personal communication) and included

acetamiprid, clothianidin, dinotefuran, imidacloprid, thiacloprid, and thiamethoxam.

Neonicotinoid concentrations were measured at the Water Science Laboratory at the University of Nebraska-Lincoln using solid phase extraction and liquid chromatography-tandem mass spectrometry (concentration-based limits of detection was 0.002ug/L; Xie et al. 2011). The Limnology Laboratory at the University of Missouri-Columbia measured chlorophyll *a*, total phosphorous, and total nitrogen in water samples. To process chlorophyll *a*, a water sample was rinsed through a glass-fiber filter to separate residue containing chlorophyll *a* pigments (within 8 hours of water collection). The filter was placed in an opaque container and stored at -10 degrees Celsius. Chlorophyll *a* was extracted from the filter using a solution of concentrated ethanol (95%) and measured using a spectrophotometer. Total nitrogen and total phosphorous were processed with digestion and analyzed using a UV-Vis spectrometer. Turbidity was measured (within 8 hours of collection) using a LaMotte 2020we/wi turbidimeter. Conductivity, pH, depth, and vegetative cover were measured in the field bi-monthly at each wetland.

Landscape Structure Variables

Proximity to nearest wetland may have an effect on aquatic macroinvertebrate species diversity and recolonization potential at ephemeral wetlands following natural drought cycles (Euliss et al. 1999; Delettre and Morvin 2000; Gledhill et al. 2008). In addition to emergence from local cysts and eggs, invertebrates also repopulate ephemeral systems from external sources by wind, in the digestive tracts of birds, and by adhering to larger vertebrate and invertebrate fauna (Proctor 1964; Swanson 1984; Pennak 1989; Euliss and Mushet 1999). To account for this potential association, I established three explanatory variables describing landscape context: (1) number of inundated wetlands ≥ 1 ha within 2.5km of a study site, (2) inundated wetland area

within 2.5km of a study site, and (3) distance from wetland perimeter to the nearest cropped field. I used multiple sets of aerial and satellite imagery to define and measure wetland inundation. I downloaded Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) data sets from www.earthexplorere.usgs.gov using the USGS Bulk Download Application. I downloaded 2015 imagery from two scenes using WRS during dates ranging from February 24 – April 13 (Appendix D.1). Scenes were visually inspected and considered unusable when atmosphere disturbance occurred (Hansen and Loveland 2012). Supplemental imagery was obtained from multispectral orthophotography collected by aircraft during the week of March 8, 2015 (Rainwater Basin Joint Venture annual spring habitat survey). For weeks when no imagery was available, I projected wetland area based on the mean rate of change during the preceding two weeks.

I processed Landsat 8 imagery in ArcMap 10.3 (ESRI 2015) using ModelBuilder to develop, edit, and manage model workflow (Appendix D.2). I combined bands 1-7 into a single RGB layer; Band 1 (Blue_0.45–0.52 μm) was used to identify water, and Bands 5 (SWIR_1.55-1.75 μm) and 6 (SWIR_1.55-1.75 μm) offered the best contrast to blue (Li et al. 2013; Yang et al. 2015). Bands 1, 5, and 6 were stacked to differentiate inundated wetlands from upland agricultural fields, prairie grasslands, and urban areas. Band 8 (15-meter resolution panchromatic raster) was fused with the stacked composite RGB raster to improve resolution using the Pan Sharpening Function (Narma et al. 2010).

I performed a supervised classification in ArcMap that allowed me to create specific land classes based on the location of known land cover types (Ozesmi and Bauer 2002; MacAlister and Mahaxay 2009; Liu et al. 2016). I apportioned four specific land cover classifications:

water, cropped, upland grassland, and urban areas. I used wetlands inundated during my field seasons as training data for water (polygons developed by walking wetted edge of study sites) and high-resolution multispectral orthophotography aerial imagery to visually develop training data for cropped fields, grasslands, and urban centers. I used a Maximum Likelihood Classification to assign each raster pixel with a unique value that corresponded with pixels observed in my training data (Ozesmi and Bauer 2002; MacAlister and Mahaxay 2009). Pixels classified as water were extracted using the Raster Calculator function and converted to shapefiles.

I designated 2.5km and 5km buffers around each study site, and clipped all inundated polygons within the buffers. The clipped shapefile included wetland polygons, as well as impoundments, streams, and false surface water polygons (e.g. shadows casted by cloud cover). To distinguish wetland polygons from other aquatic non-wetland polygons, I superimposed the constructed shapefile with a layer provided by the National Wetlands Inventory (NWI; USFWS 2011). I visually inspected contradictions between the two layers, and eliminated discrepancies based on personal judgement and knowledge of the area.

I conducted a paired t-test to validate Landsat methods developed in this study by comparing known wetland area (ground truthed by walking wetted edge of each wetland with handheld GPS unit) and projected wetland area obtained from processed Landsat data. Area estimates were only included if Landsat imagery was recorded within 7 days of a wetland being ground truthed (Appendix D.3). Additionally, wetland sites concurrently pumped with ground water were not included in the analysis. No statistical differences in known wetland area and projected wetland area were observed ($t_{76,0.05} = 0.45$).

Invertebrate Collection and Processing

I assessed benthic and nektonic aquatic invertebrate communities in alternating weeks from 22 February through 18 April 2015. At each wetland, I established three randomly located 3m x 3m sample plots at water depths < 30cm and where vegetative cover was < 50% (Figure 1.3). Within each plot, I collected two nektonic samples using a 500 μm rectangular sweep net (Murkin et al. 1994, Tapp and Webb 2015). I lowered the net vertically into the water column, pressed firmly against the substrate, and bounced it through the water column over a distance of 1.1 m (0.5m^2). Four passes were completed (Klemm et al. 1990, Davis and Bidwell 2008). Within each plot, I also collected two benthic samples using a 10cm diameter x 5cm deep benthic core sampler (Swanson 1983; Tapp and Webb 2015) in an undisturbed area adjacent to the corresponding nektonic sampling location. Benthic and nektonic samples were preserved in 70% ethanol to prevent deterioration (Murkin and Kadlec 1986) and shipped to the University of Missouri for processing.

To increase efficiency in invertebrate sorting and identification, I stained samples with rose bengal for 24 hours (Sherfy et al. 2000; Tapp and Webb 2015). I rinsed invertebrate samples through a series of two graduated sieves (#50 [500 μm] and #10[250 μm]) to remove small clay particles and partition remaining materials into coarse and fine samples (Tapp and Webb 2015). I used a Folsom wheel sample splitter (Aquatic Research Instruments) to subsample (1/4 volumes) debris retained by the 250 μm and 500 μm sieves (Whiting et al. 2011; Tapp and Webb 2015). Aquatic invertebrates were removed from remaining debris, identified to the lowest taxonomic level possible, measured to the nearest millimeter, and catalogued. I calculated biomass estimates for individual taxa using published dry length-mass regressions (Duffy and Labar 1994; Benke et al. 1999; Appendix B). When a length-mass regression was

not available for specific taxa, I used estimates from similar species observed in comparable habitats (Benke and Huryn 2006). I was unable to identify published length-mass regression models for taxon similar to phylum Nematoda and class Hirudinea. Further, I was unable to obtain published length-mass regression models for families within the order Gastropoda that did not include shell mass. Using a subset of specimens collected from my study, I developed dry length-mass regressions for Nematoda and Hirudinea, and length-mass regressions that excluded the shells of families Planorbidae, Physidae, and Bithyniidae following methods by Benke et al. 1999 (Appendix C). Biomass estimates were pooled and averaged for all taxa collected at a wetland for each sampling event and converted to biomass density estimates (g/m²). Production was then calculated using the P:B method for each taxonomic group (Equation 1; Benke 1984, Duffy and LaBar 1994) from published P:B values (Waters 1977; Duffy and LaBar 1994; Stagliano and Whiles 2002; Benke and Huryn 2006; Whitting et al. 2011; Butkas 2011).

Equation 1:
$$\text{Secondary Production}_2 = \text{biomass}_2 \times \frac{\text{Production}_1}{\text{biomass}_1}$$

where:

- Production₁ is the annual production value from a previous study
- Biomass₁ is the total biomass from a previous study
- Biomass₂ is the biomass from the current study

Statistical Analysis

Neonicotinoid benchmarks used for comparison were derived from the Environmental Protection Agency (EPA, USA), and Canadian Environmental Quality Guidelines (CCME, Canada). Neonicotinoid benchmarks proposed by the EPA and CCME are specific to imidacloprid, however at the time data were collected, clothianidin was the most commonly applied class of neonicotinoid in the RWB (USGS National Synthesis Project 2014). I accounted

for toxic equivalence using an equation developed by Cavallaro et al. (2016) to convert imidacloprid-based benchmarks to clothianidin benchmarks. I conducted a one-sample t-test for each benchmark to determine if mean pre-planting concentrations observed in the RWB were equal to proposed maximum chronic concentrations (Anderson et al. 2013). I used multispectral orthophotography imagery collected by aircraft during the week of March 8, 2015 to identify buffered and non-buffered wetlands. I defined a wetland as buffered when the entire perimeter was surrounded by $\geq 50\text{m}$ of vegetation other than row crops. I classified a wetland as non-buffered when $< 50\text{m}$ of vegetation other than row crops lied between the wetted edge of a wetland and a cropped field (Rainwater Basin Joint Venture 2013). I also classified sites with predominant drainage structures leading from a cropped field into a wetland as non-buffered (Riems et al. 2013). Buffered and non-buffered concentrations did not follow normal distributions, thus I conducted a two-sample Mann–Whitney rank sum test with continuity correction to determine if the difference between neonicotinoid concentrations at buffered and non-buffered sites were statistically different (Van Dijk et al. 2013).

I developed a set of *a priori* candidate models for explaining nektonic and benthic invertebrate rarefied richness, Shannon Wiener's index of species diversity, density (g/m^2), and invertebrate community production based on ecologically reasonable scenarios supported by published literature and personal experience (Tables 3.1 and 3.2; Gleason et al. 2003; Riems et al. 2013; Van Dijk et al. 2013). Rarefied richness is a method used to assess species richness that accounts for sampling effort (Gotelli and Colwell 2011). Rarefaction curves can be used to determine if sampling efforts were adequate in accurately identifying species richness at a study site (Gotelli and Colwell 2011). In program R (R Development Core Team 2017), I calculated rarefied richness and Shannon diversity (Package 'vegan'; Oksanen et al. 2016). I used an

information theoretic approach to evaluate a priori models for explaining response variables (Burnham and Anderson 2002) using a linear mixed model procedure (AIC_c, AIC_{modavg}; Mazerolle 2016). Density and secondary production did not exhibit normal distributions and were normalized with a Yeo-Johnson Power Transformation (Yeo and Johnson 2000). To prevent heteroscedasticity in fitted residuals, I scaled and centered all independent variables on zero (Gelman 2008). Pearson's r-correlation indices were used to test for multicollinearity between independent variables in candidate models (Dormann et al. 2013). When evaluating response variables for nektonic communities, I assigned water depth to all *a priori* models, given the strong relationship it had with all response variables. I analyzed *a priori* models for nektonic and benthic communities separately because of differences in sampling device and predicted community structure (Tapp and Webb 2015). For each response variable, I fit models using maximum likelihood estimation and calculated output statistics including second order Akaike's Information Criterion (AIC_c) values and model weights (AIC_{modavg}; Mazerolle 2016). All models within two Δ AIC_c values of the most likely model were considered competitive, and included in a competing model set (Richards 2005). For each competing model, I performed a Shapiro-Wilk test on the distribution of fitted residuals for normality, and discarded models when the resulting p-value was < 0.05 . I used a coefficient of determination adjusted for the number of predictors (R^2) to measure the goodness of fit for each model selected. Models were removed when an adjusted $R^2 < 0.10$, since these models did not explain a meaningful percentage of variability for response variables (Stephens et al. 2005). Finally, I calculated the natural average for each parameter estimates observed in the *remaining* set of competing candidate models to determine if it had a statistical influence on invertebrate communities

(Burnham and Anderson 2002; Mazerolle 2016). I used model-averaged parameter estimates and associated 95% confidence intervals to guide hypothesis testing.

RESULTS

During spring 2015, I detected trace levels of neonicotinoids in water samples from 20 of 26 wetlands in March and 16 of 22 wetlands in April (Appendix P and Q). Combining collection periods, total neonicotinoid concentrations at wetlands ranged from 0.00 – 23.67ng/L (\bar{x} = 5.53ng/L). Concentrations were greater in April than March (\bar{x} = 6.18 and 5.16ng/L, respectively), however there were no statistical differences between sampling periods ($t_{21, 0.05} = 1.72$). Clothianidin was detected in 100% of samples containing trace levels of neonicotinoids, while imidacloprid was detected in only 11% of samples. Acetamiprid, dinotefuran, thiacloprid, and thiamethoxam went undetected in all samples. The maximum observed concentration was approximately 15x less than the CCME benchmark of 373 ng/L (Canadian Council of Ministers of the Environment. 2007), and samples collected from three wetlands had observed concentrations exceeding the most conservative benchmark of 16.20 ng/L proposed by the EPA (Environmental Protection Agency 2017; Figure 3.2). Mean neonicotinoid concentrations across all observations were significantly less than proposed benchmarks (CCME: $t_{47, 0.05} = -477.00$ and EPA: $t_{47, 0.05} = -13.85$). Mean total neonicotinoid concentrations in water differed between wetland buffer treatments (Mann–Whitney $U_{48, 0.05} = 176$; Figure 3.2). Mean total neonicotinoid concentration measured in water samples collected from wetlands buffered by ≥ 50 m of native vegetation ($n=20$) was 3.82ng/L (SE = 0.90) and ranged from 0-16.52ng/L. Mean total neonicotinoid concentration measured in water samples collected from wetlands buffered by < 50 m of native vegetation ($n=28$) was 6.97ng/L (SE = 1.14) and ranged from 0-23.67.

Nektonic rarefied richness ranged from 12 – 24 species (mean = 17.98, SE = 0.77; Appendix O) at 22 wetlands sampled during spring 2015 (Appendix M). I identified four competing models for explaining nektonic rarefied species richness, which accounted for 55% of the total AICc weight (Table 3.4). Explanatory variables included in the most likely model were water depth (+) and pH (-). Conductivity (-), total dissolved solids (-), and total nitrogen (-) appeared in subsequent competing models. Water depth was the only significant when averaged over all competing models using AICc weights ($p > 0.05$; Figure 3.3a). Benthic rarefied richness ranged from 4 – 9 (mean = 6.27, SE = 0.34), which was significantly less than what was observed in nektonic communities ($t_{21, 0.05} = 13.84$; Appendix N). I identified five competing models to explain benthic rarefied richness, which accounted for 52% of the total AICc weight (Table 3.4). I omitted three models with adjusted $R^2 < 0.10$ since these models did little to explain variation in rarefied richness. The only explanatory variable observed in the most likely model was pH (-). Total neonicotinoid concentration (+) was observed in the only subsequent competing model. No variables from my analysis had a significant association with benthic rarefied richness when averaged over all competitive models ($p > 0.05$; Figure 3.3b).

Nektonic diversity ranged from 0.68 – 1.87 (mean = 1.41, SE = 0.07; Appendix O) at 22 wetlands sampled during spring of 2015. Explanatory variables in the best-fit model for explaining nektonic diversity included water depth (+), and pH (-) and the model accounted for 35% of the total AICc weight (Table 3.5). Both explanatory variables observed in the model were significant ($p > 0.05$; Figure 3.4a). Benthic diversity ranged from 0.25 – 1.19 (mean = 0.81, SE = 0.05), which was significantly less than what was observed in nektonic communities ($t_{21, 0.05} = 7.06$). I identified one model within $\Delta AICc=2$ that accounted for 41% of the total AICc

weight (Table 3.5). Chlorophyll α (+) was the only explanatory variable observed in the most likely model and had a significant association with invertebrate diversity ($p > 0.05$; Figure 3.4b).

Nektonic density (g/m^2) ranged from 0.18 – 2.04 (mean = 0.64, SE = 0.09; Appendix O) at 22 wetlands sampled during the spring of 2015. I identified three competing models for explaining nektonic density, which accounted for 39% of the total AICc weight (Table 3.6). Fitted residuals for all competing models followed a normal distribution, and did not display patterns when plotted. Explanatory variables included in the most likely model were water depth (+) and total nitrogen (-). Percent emergent vegetation (+), chlorophyll α (+), and total neonicotinoid concentrations (-) were observed in subsequent competing models. Water depth and total neonicotinoid concentration were significant when explaining density averaged over all competing models using AICc weights (Figure 3.5a). Benthic invertebrate density (g/m^2) ranged from 0.13 – 4.11 (mean = 1.05, SE = 0.19), which was significantly greater than what was observed in nektonic communities ($t_{21, 0.05} = 1.98$). I identified three competing models for explaining benthic density, which accounted for 58% of the total AICc weight (Table 3.6). Total dissolved solids (-) was the only explanatory variable observed in the most likely model. Conductivity and pH were observed in subsequent competing models. Total dissolved solids, conductivity, and pH were significant when explaining benthic density averaged over all competing models using AICc weights (Figure 3.5b).

Nektonic secondary production ($\text{g}/\text{m}^2/\text{year}$) ranged from 1.88 – 18.91 (mean = 6.83, SE = 0.85; Appendix O) at 22 wetlands sampled in spring 2015. I identified two competing models for explaining nektonic production, which accounted for 38% of the total AICc weight (Table 3.7). Variables included in the most likely model were water depth (+) and percentage of emergent vegetation (+). Distance to nearest cropped field (-) was observed in the competing

model; however, only water depth was significant for explaining production when averaged over both competing models using AICc weights (Figure 3.6a). Benthic secondary production ($\text{g}/\text{m}^2/\text{year}$) ranged from 1.13 – 20.86 (mean = 5.05, SE = 0.95), which was not statistically different from production estimates observed in nektonic communities ($t_{21, 0.05} = 1.39$). I identified two competing models for explaining benthic production, which accounted for 57% of the total AICc weight (Table 3.7). Conductivity (-) was the only explanatory variable observed in the most likely model. Total dissolved solids (-) was observed in the only subsequent competing model. Conductivity and total dissolved solids were significant when explaining production averaged over both competing models using AICc weights (Figure 3.6b).

DISCUSSION

Previous studies at playa wetlands in the RWB have focused on elemental contaminants, nutrient inputs, and several classes of agricultural pesticides (Foster 2010; Belden et al. 2012; Reins et al. 2013); however, the presence and concentration of neonicotinoid insecticides were previously unknown. I detected trace levels of neonicotinoids at 92% of all playa wetlands sampled in the RWB during the spring of 2015. I predicted a relatively high detection rate given the intensity of row crop production in the region, although concentrations were lower than expected (USGS Pesticide National Synthesis Project 2014; Main et al. 2014). Concentrations at all 26 wetlands sampled fell below toxicity benchmarks proposed by the CCME, and only 11% of wetlands sampled had concentrations exceeding the most conservative benchmark proposed by the EPA. Although imidacloprid has been the most widely studied class of neonicotinoid (Cavallaro et al. 2016), its application in the RWB was minimal relative to clothianidin and thiamethoxam (USGS Pesticide National Synthesis Project 2014). Accordingly, I estimated mean concentrations for imidacloprid and clothianidin as 0.55ng/L and 4.98ng/L, respectively.

Although thiamethoxam went undetected at all study sites, it is important to note that formation of clothianidin often occurs as thiamethoxam degrades (Zabar et al. 2012), thus, a decrease in thiamethoxam could prompt an increase in clothianidin.

I collected water samples preceding spring planting during months March and April; therefore, all observations are considered chronic toxicity levels (Schaafsma 2015; Main et al. 2017). Chronic neonicotinoid concentrations and detection frequencies reported by previous studies have been somewhat conflicting, and were likely influenced by precipitation and runoff, upland crop production, and application period (Anderson et al. 2013; Hladik et al. 2014; Main et al. 2014). For instance, pre-planting detections by Main et al. (2014) included trace levels of aqueous neonicotinoid concentrations at 36% of Prairie Pothole wetlands and a mean concentration of 8.3ng/L during a dry year, whereas in a wet year detection frequency and mean concentration increased to 91% and 52.7ng/L. Neonicotinoids have greater half-lives in soils than water, thus upland runoff can effectively recharge concentrations following adequate rainfall (Zabar et al. 2012; Hladik et al. 2014; Main et al. 2014). Precipitation was minimal in the RWB leading into spring 2015 (Table 1.1; National Oceanic and Atmospheric Administration 2017); therefore, concentrations observed in my study were likely reduced residuals from runoff accumulated during late summer and fall 2014 (Main et al. 2014).

Upland native grasses facilitate trapping and deposition of sediment-absorbed contaminants in runoff by increasing infiltration and sorbing dissolved phase substances to vegetation and soil surfaces in the buffer strip (Dunn et al. 2011). This relationship was evident in my study, as I observed significantly greater total neonicotinoid concentrations at wetlands without vegetative buffers (Table 3.3). Imidacloprid concentrations were approximately equal among buffered and non-buffered wetlands; however, clothianidin concentrations at non-

buffered wetlands were approximately 2-fold greater than wetlands with vegetative buffers. Prior studies have reported equivocal results concerning the utility of vegetative buffers. Dunn et al. (2011) reported pesticides (excluding neonicotinoids) underwent a 52% reduction in concentrations for aqueous and filter phases in 10m grass buffers at operational farms in Prince Edward Island. In contrast, Main et al. (2017) found buffer zone width to be irrelevant when predicting aqueous neonicotinoid concentrations. There are a number of interacting processes influencing the utility of vegetative buffer strips, and determining effectiveness will require accurate information about a system of interest and complex modeling (Lacas et al. 2005; Dunn et al. 2011). In the RWB, the size of the vegetative buffer strip may be influential in mediating neonicotinoid concentrations in surface waters.

Acute and chronic toxicity thresholds for neonicotinoids vary greatly among aquatic invertebrate taxa (Morrissey et al. 2015). The time of year my water samples were collected in conjunction with minimal precipitation preceding my field season likely limited my assessment to focusing on the effects of chronic neonicotinoid exposures. Chronic LC₅₀ for invertebrate taxa in the RWB ranged from 195ng/L for Ephemeroptera (Roessink et al. 2013) to >4.3 X 10⁷ng/L for Cladocera (Sanchez-Bayo 2006; Morrissey et al. 2015). I observed a total maximum concentration of 23.67ng/L, which is far below the LC₅₀ suggested for Ephemeroptera by a magnitude of eight-fold (Roessink et al. 2013). Therefore, it is unlikely that observed concentrations had an influence on spring macroinvertebrate mortality; however, sub-lethal effects might explain the strong negative association I observed between neonicotinoid concentrations and nektonic density (g/m²; Alexander et al. 2007; Morrissey et al. 2015; Cavallaro et al. 2016). Chronic exposure to concentrations as low as 10ng/L may have adverse effects on the life cycles of freshwater invertebrates (Environmental Protection Agency 2017).

Sub-lethal effects include, but may not be limited to, growth, reproduction, immobility, feeding, swimming, and emergence (Alexander et al. 2007; Morrissey et al. 2015; Cavallaro et al. 2016).

Sub-lethal effects may also explain why I did not observe a relationship between neonicotinoid concentrations and nektonic secondary production. Secondary production accounts for the rate at which invertebrate biomass increases through growth, reproduction, and survivorship in space and time (Waters 1977; Benke and Huryn 2006), making it an excellent metric for assessing sub-lethal effects of contaminants. Accurately calculating secondary production for a species within a system requires intense sampling of a cohort during their entire life history cycle (Benke and Huryn 2006). Given limitations in time and resources, I used an abbreviated method that relied on published P:B values developed outside of the RWB to estimate secondary production (Equation 1: Waters 1969; Benke 1984; Benke and Huryn 2006). Habitat conditions (e.g. pH, dissolved oxygen, temperature, contaminants) vary across systems and the physiological response to changing environments alter precise estimates of secondary production (Benke and Huryn 2006). A meaningful percentage of P:B values used in my study were developed for stream invertebrates and may not be applicable for wetland communities (Waters 1977; Krueger and Waters 1983). Given its utility, future studies assessing the sub-lethal effects of neonicotinoids should consider measuring secondary production of specific taxa using a cohort technique (Krueger and Waters 1983).

MANAGEMENT IMPLICATIONS

Toxicity benchmarks are commonly used to estimate risk and identify potential for adverse effects of pesticide concentrations in surface waters (Anderson et al. 2013). Overall, neonicotinoid concentrations measured in this study were below chronic toxicity benchmarks proposed by the EPA and CCME. Although results from this study are encouraging, it is likely I

assessed wetlands during a time of year when concentrations were minimal. Moving forward, there is a need to measure post-planting concentrations at wetlands in the RWB following intense precipitation events to identify acute levels. Subsequent sampling would also allow researchers to determine duration of acute pulse and more accurately define intermediate concentrations.

While my results support previous studies suggesting vegetative buffers will improve quality of surface waters, there is still an economic need to determine the precise length of an effective and efficient vegetative buffer strip. Based on current management planning recommendations (Rainwater Basin Joint Venture 2013), I designated a 50m buffer for all sites, which is equivalent to 6.40ha of economically valuable farmland surrounding a circular wetland of 10ha. Although I observed a 50% reduction in neonicotinoid concentrations at buffered wetlands, previous studies have observed significant reductions of other pollutants using buffer strips 10m in length (Dunn et al. 2011). While increasing vegetative buffer area can decrease unwanted contaminants, doing so might also have a negative influence wetland water budgets by decreasing surface runoff (Castelle et al. 1994). Although beyond the scope of this study, development of predictive models based on, but not limited to, seasonal precipitation, precipitation intensity, topography, and ratio of the area farmed to area of inundated wetland would be useful for determining appropriate buffer strip length at wetlands under varying field conditions.

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Table 3.1. Designations, descriptions, and methods used to collect local and landscape habitat data for assessing aquatic macroinvertebrate communities in the Rainwater Basin, spring 2015.

Variable	Description	Methods
Veg	% emergent vegetation	Percentage of wetland area where emergent vegetation occurred opposed to open water.
Depth	Mean water depth	Water depth (cm) was measured bi-monthly by recording depths at 12-16 random locations and averaged for the wetland.
Neonic	Neonicotinoids	Neonicotinoid ($\mu\text{g/L}$) was measured and averaged from monthly composite grab samples.
NeonicMax	Neonicotinoids Max	Maximum neonicotinoid concentration observed between the two sample periods.
TN	Nitrogen	Total Nitrogen (mg/L) was measured and averaged from monthly composite grab samples.
TP	Phosphorus	Total Phosphorus ($\mu\text{g/L}$) was measured and averaged from monthly composite grab samples.
Turb	Turbidity	Turbidity (NTU) was measured and averaged from monthly composite grab samples.
pH	pH	pH was measured bi-monthly at 12-16 locations in each wetland and averaged.
Cond	Conductivity	Conductivity ($\mu\text{S/cm}$) was measured bi-monthly at 12-16 locations in each wetland and averaged.
TDS	Total Dissolved Solids	Total dissolved solids (mg/L) were measured bi-monthly at 12-16 locations in each wetland and averaged.
2kmDen	2.5km wetland complex	Wetland density within a 2.5km radius of a study site. Wetland density was assessed every ~16 days using satellite and aerial imagery.
2kmUnits	2.5km wetland complex	Number of wetlands >1ha in area within a 2.5km radius of a study site. Wetland density was assessed every ~16 days using satellite and aerial imagery.
DistRC	Distance to Cropped Field	Distance (meters) from perimeter of wetland to the nearest cropped field.

Table 3.2. *A priori* candidate models used to explain variation in benthic and nektonic invertebrate communities at wetlands (n=22) in Nebraska’s Rainwater Basin during spring 2015.

Level	Model
Local_Non Neonic Models	Depth + Chlorophyll <i>a</i>
	Depth + Phosphorus
	Depth + Nitrogen
	Depth + Turbidity
	Depth + Total Dissolved Solids
	Depth + Conductivity
	Depth + pH
	Depth + Vegetative Cover
Local_Neonic Models	Depth + Neonic
	Depth + Neonic Maximum Observed
	Depth + Neonic + Chlorophyll <i>a</i>
	Depth + Neonic + Phosphorus
	Depth + Neonic + Nitrogen
	Depth + Neonic + Turbidity
	Depth + Neonic + Total Dissolved Solids
	Depth + Neonic + Conductivity
	Depth + Neonic + pH
Depth + Neonic + Vegetative Cover	
Landscape_Non Neonic Models	Depth + 2kmDensity
	Depth + 2kmUnits
	Depth + Cropped Distance
Landscape_Neonic Models	Depth + Neonic + 2kmDensity
	Depth + Neonic + 2kmUnits
	Depth + Neonic + Cropped Distance

*Depth was only considered when analyzing nektonic communities.

Table 3.3. Summary statistics for neonicotinoid concentrations and water quality measures at buffered (n=20) and non-buffered (n=28) study sites, Rainwater Basin, Nebraska, 2015. Buffered wetland were defined as wetlands with >50m of native vegetation (includes introduced plant species not used in crop management) between inundated area and cropped field.

	<u>Buffered Wetlands</u>					<u>Non-Buffered Wetlands</u>				
	Mean	S.E.	Median	Min	Max	Mean	S.E.	Median	Min	Max
<u>Neonicotinoids</u>										
Clothianidin (ng/L)	3.23	0.84	2.46	0.00	16.52	6.45	0.97	5.92	0.00	22.82
Imidacloprid (ng/L)	0.59	0.40	0.00	0.00	6.60	0.53	0.40	0.00	0.00	9.90
Total Neonicotinoid (ng/L)	3.82	0.90	2.71	0.00	16.52	6.97	1.14	6.15	0.00	23.67
<u>Nutrient</u>										
Chlorophyll-a (ug/L)	150.60	29.58	125.08	47.71	356.28	236.79	64.92	179.80	30.81	803.61
Total Phosphorus (ug/L)	1391.10	379.11	919.00	342.00	3485.50	1225.79	158.18	1039.75	656.50	2431.50
Total Nitrogen (mg/L)	5.68	1.14	4.24	1.47	11.77	5.38	1.03	4.65	1.79	14.72
<u>Additional Water Quality</u>										
pH	7.92	0.12	7.97	7.38	8.41	7.77	0.11	7.81	6.97	8.28
Conductivity (uS/cm)	374.08	101.27	254.44	105.44	1155.33	209.84	34.28	167.22	107.78	452.92
Total Dissolved Solids (mg/L)	187.06	50.55	127.67	51.56	577.08	104.75	17.08	83.65	53.78	226.17
Turbidity (NTU)	1381.62	484.36	737.00	30.10	3912.00	616.90	292.56	110.85	18.00	3126.00

Table 3.4. Competing models for explaining rarefied nektonic and benthic macroinvertebrate richness from linear model regression. Competing models were ranked using Akaike's Information Criterion (AICc) and models within two AICc ($\Delta AICc$) were considered as competing models. The AICc weight of a model relative to all candidate models (n=25) is denoted by ω . Likelihood that an individual model is the best-fitted model relative to all candidate models assessed is denoted by ω . ML is the relative likelihood of the model given the data. The adjusted coefficient of determination (R^2 Adj) indicates the proportion of the variance in the response variable explained by the explanatory variable(s).

Rarefied Richness: Competing Candidate Models							
Community	Model	K	AIC _c	ΔAIC_c	ω	ML	R ² Adj
Nektonic	RareN ~ 1 + Depth + pH	4	109.69	0.00	0.17	1.00	0.55
	RareN ~ 1 + Depth + Cond	4	109.82	0.13	0.15	0.94	0.53
	RareN ~ 1 + Depth + TDS	4	109.86	0.17	0.15	0.92	0.53
	RareN ~ 1 + Depth + Nitrogen	4	111.05	1.35	0.08	0.50	0.50
Benthic	RareB ~ 1 + pH	3	86.35	0.00	0.14	1.00	0.10
	RareB ~ 1 + Cond*	3	86.96	0.61	0.10	0.74	0.07
	RareB ~ 1 + 2kmDen*	3	86.99	0.64	0.10	0.72	0.07
	RareB ~ 1 + TDS*	3	87.02	0.67	0.10	0.71	0.07
	RareB ~ 1 + Neonic + pH	4	87.38	1.03	0.08	0.60	0.13

* Models omitted from final analysis given minimum required R² adjusted value

Table 3.5. Competing candidate models for explaining invertebrate diversity from linear model regression. Competing models were ranked using Akaike's Information Criterion (AICc). Models within 2 AICc ($\Delta AICc$) were considered as competing candidate models. The AICc weight of a model relative to all candidate models (n=25) is denoted by ω . Likelihood that an individual model is the best-fitted model relative to all candidate models assessed is denoted by ω . ML is the relative likelihood of the model given the data. The adjusted coefficient of determination (R^2 Adj) indicates the proportion of the variance in the response variable explained by the explanatory variable(s).

Shannon Wiener's Diversity Index: Competing Candidate Models							
Community	Model	K	AIC _c	ΔAIC_c	ω	ML	R ² Adj
Nektonic	ShannonN ~ 1 + Depth + pH	4	1.50	0.00	0.35	1.00	0.54
Benthic	ShannonB ~ 1 + Chl_a	3	1.99	0.00	0.41	1.00	0.22

Table 3.6. Competing candidate models for explaining invertebrate density from linear model regression. Competing models were ranked using Akaike's Information Criterion (AICc). Models within 2 AICc ($\Delta AICc$) were considered as competing candidate models. The AICc weight of a model relative to all candidate models (n=25) is denoted by ω . Likelihood that an individual model is the best-fitted model relative to all candidate models assessed is denoted by ω . ML is the relative likelihood of the model given the data. The adjusted coefficient of determination (R^2 Adj) indicates the proportion of the variance in the response variable explained by the explanatory variable(s).

Invertebrate Density: Competing Candidate Models							
Community	Model	K	AIC _c	ΔAIC_c	ω	ML	R ² Adj
Nektonic	Density ~ Depth + TN	4	-57.49	0.00	0.17	1.00	0.40
	Density ~ Depth + Veg	4	-57.14	0.34	0.14	0.84	0.39
	Density ~ Depth + Neonic + Chl_a	5	-55.88	1.61	0.08	0.45	0.42
Benthic	Density ~ 1 + TDS	3	-2.41	0.00	0.24	1.00	0.17
	Density ~ 1 + Cond	3	-2.40	0.01	0.24	1.00	0.17
	Density ~ 1 + pH	3	-0.70	1.71	0.10	0.43	0.11

Table 3.7. Competing candidate models for explaining annual invertebrate production from linear model regression. Competing models were ranked using Akaike's Information Criterion (AICc). Models within 2 AICc ($\Delta AICc$) were considered as competing candidate models. The AICc weight of a model relative to all candidate models ($n=25$) is denoted by ω . Likelihood that an individual model is the best fitted model relative to all candidate models assessed is denoted by ω . ML is the relative likelihood of the model given the data. The adjusted coefficient of determination (R^2 Adj) indicates the proportion of the variance in the response variable explained by the explanatory variable(s).

Invertebrate Production: Competing Candidate Models							
Community	Model	K	AIC _c	ΔAIC_c	ω	ML	R ² Adj
Nektonic	Production ~ 1 + Depth + Veg	4	3.27	0.00	0.24	1.00	0.36
	Production ~ 1 + Depth + DistRC	4	4.35	1.08	0.14	0.58	0.33
Benthic	Production ~ 1 + Cond	3	9.75	0.00	0.29	1.00	0.20
	Production ~ 1 + TDS	3	9.84	0.09	0.28	0.96	0.20

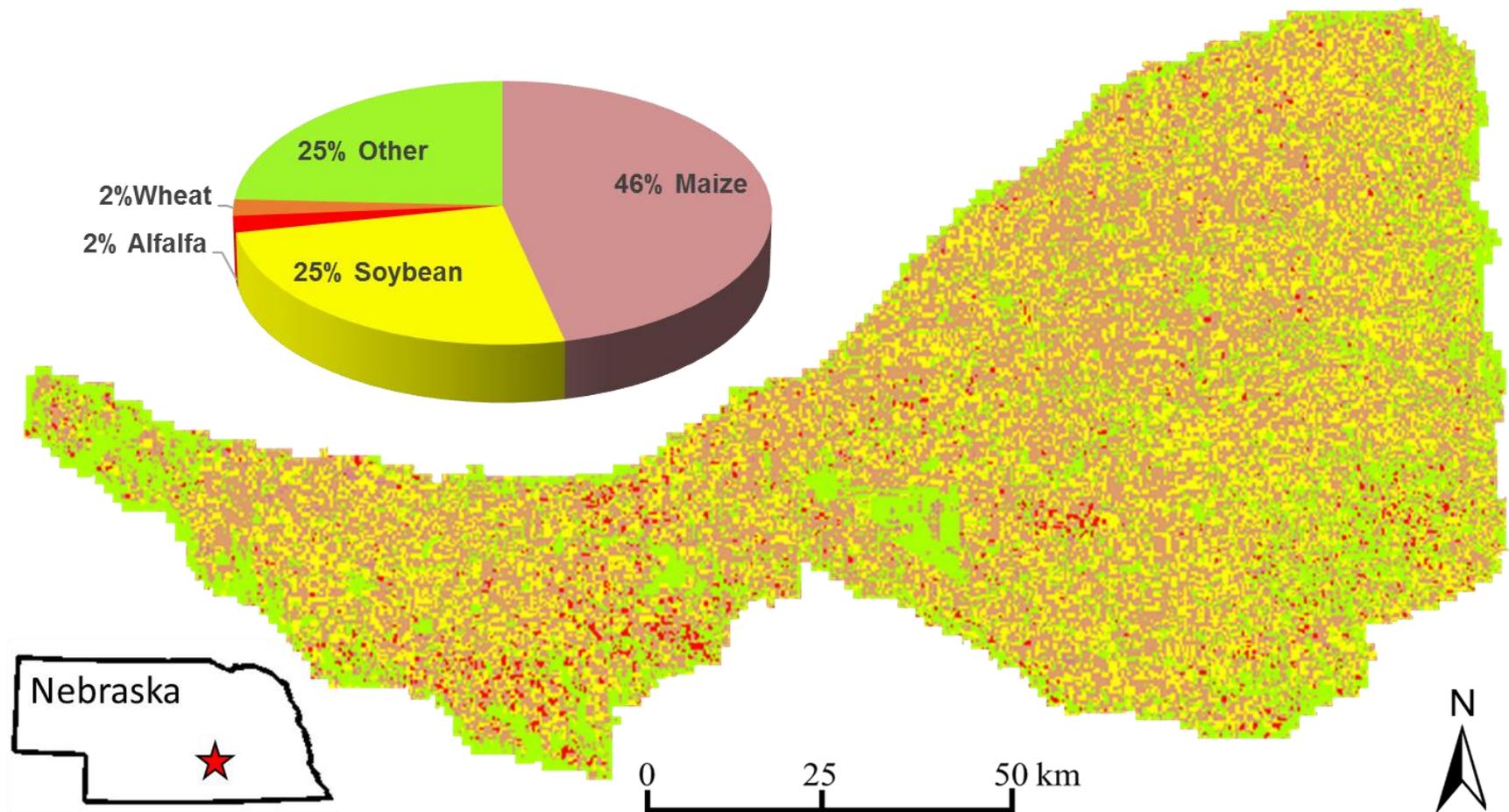


Figure 3.1 Approximately 75% of land cover in Nebraska's Rainwater Basin has been converted from mixed prairie grassland to agricultural production. Accordingly, 75% of the regions wetlands receive contaminated runoff from upland crop fields (United States Geological Survey 2014)

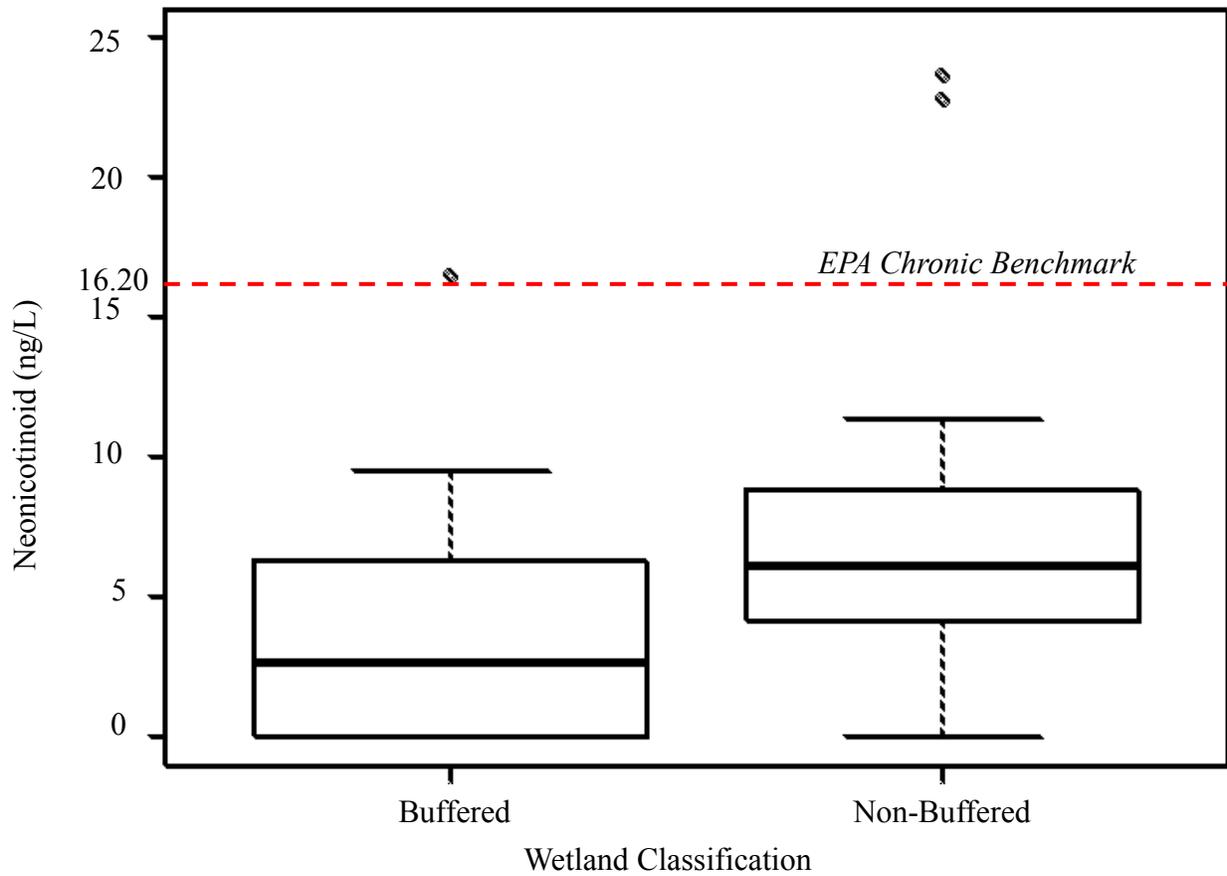
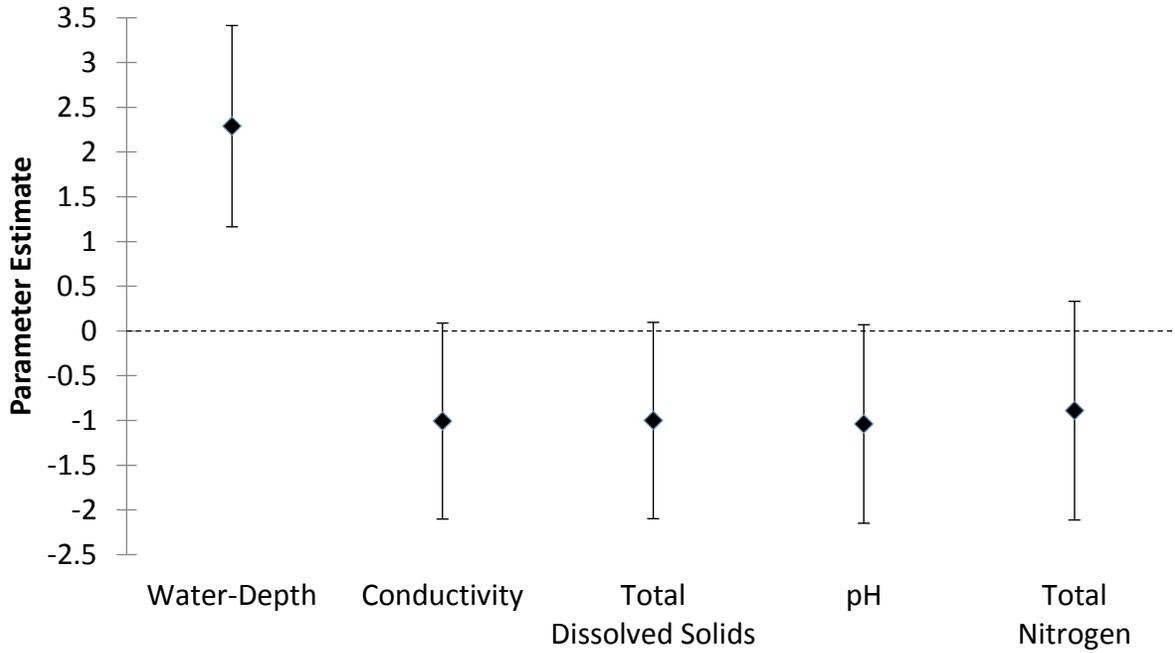


Figure 3.2 Total neonicotinoid concentrations (including imidacloprid and clothianidin) at buffered and non-buffered wetlands in Nebraska’s Rainwater Basin, USA. Chronic concentrations exceeded benchmark toxicity standards set by the Environmental Protection Agency at three wetlands. Median neonicotinoid concentrations among the two treatment types were statistically different (Mann–Whitney $U_{48,0.05} = 176$).

Figure 3.3 Model-averaged parameter estimates and associated 95% confidence intervals for parameters observed in competing model set for explaining rarefied richness. Model-averaged parameter estimates and standard errors were averaged over four competing models using their AICc weights for nektonic communities (A), and two competing models for benthic communities (B).

A)



B)

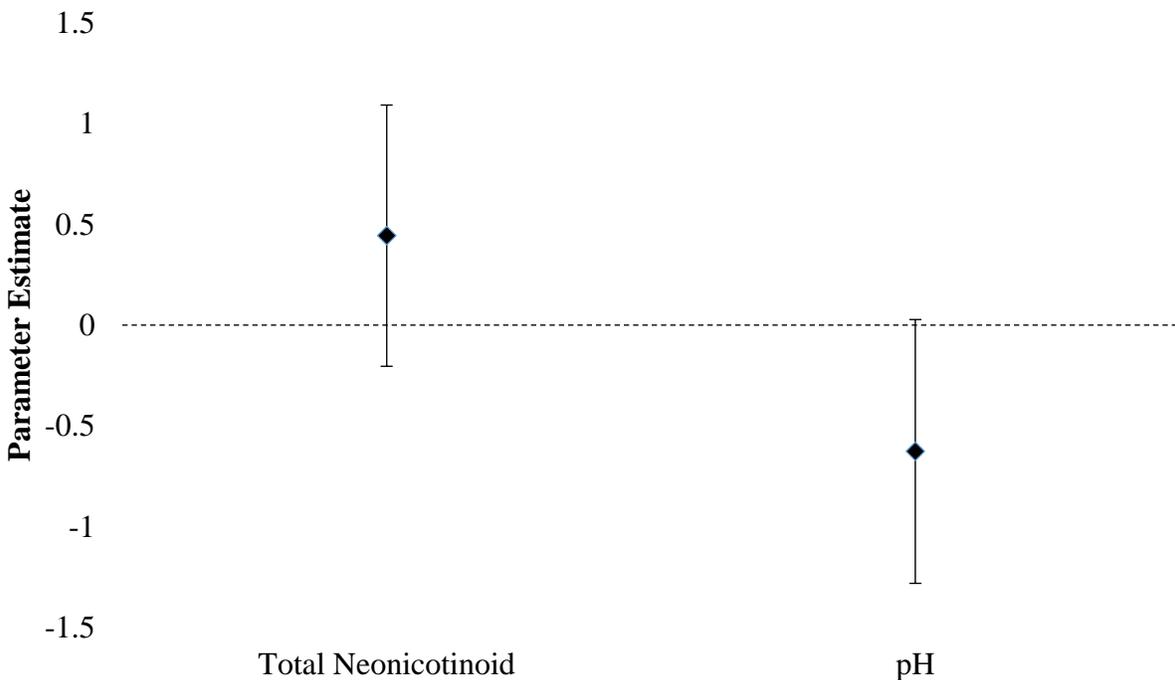
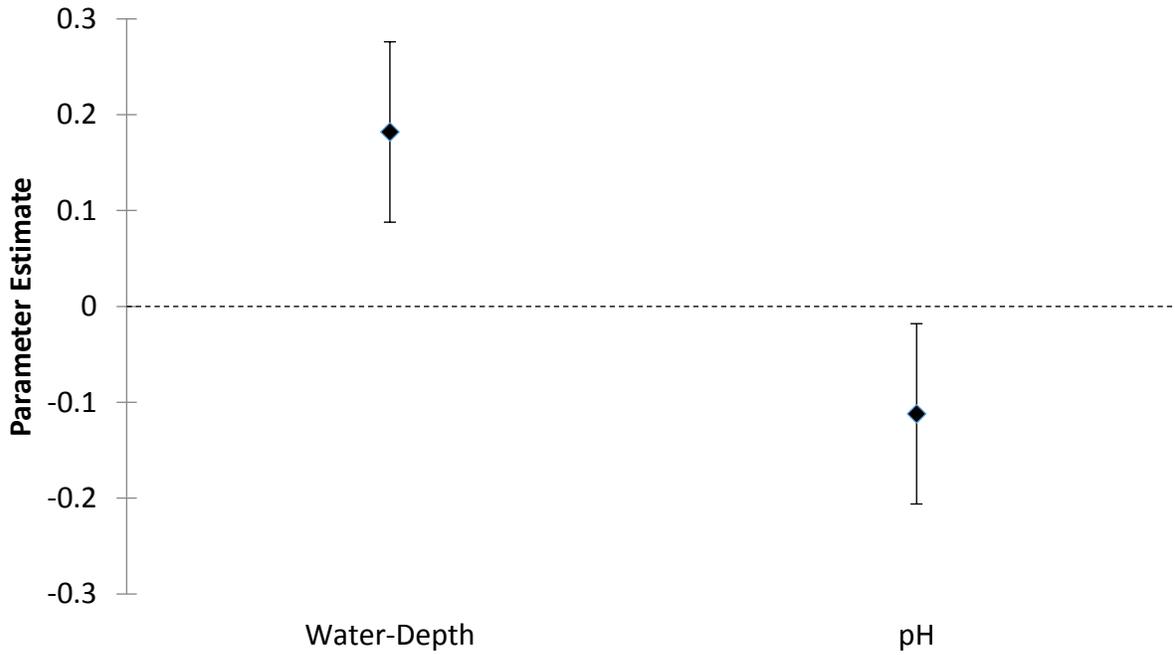


Figure 3.4 Parameter estimates and associated 95% confidence intervals for parameters observed in the most likely model for explaining invertebrate diversity. There were no models within two AICc of the most likely model when explaining diversity in nektonic communities (A), and benthic communities (B).

A)



B)

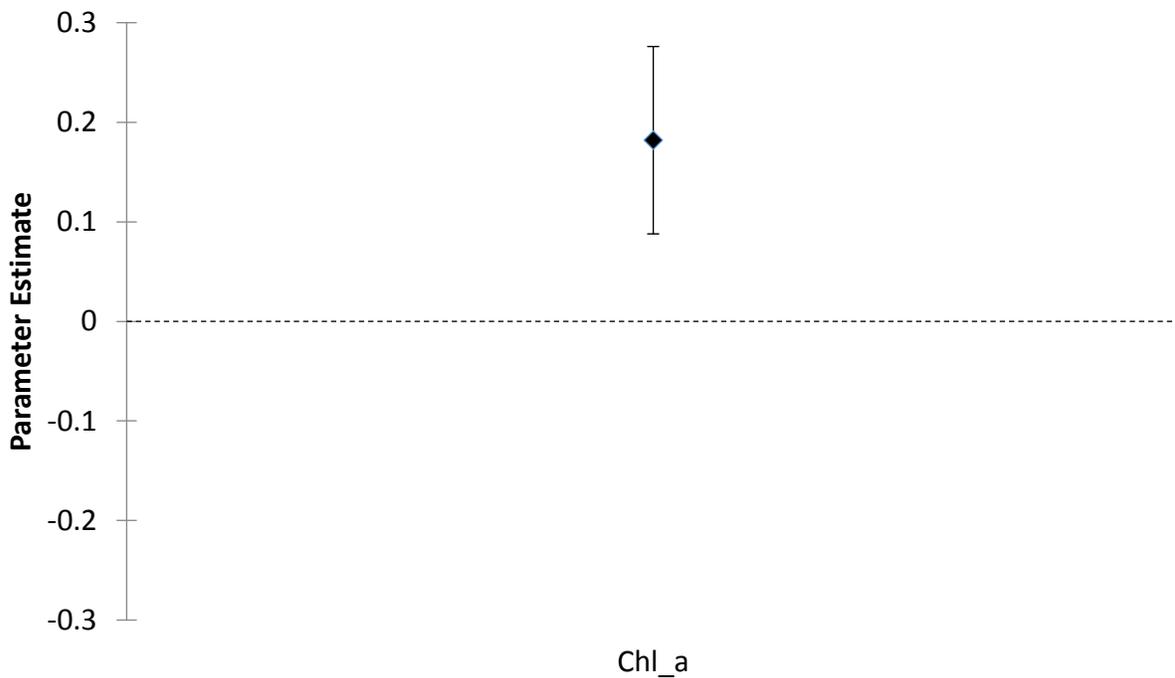
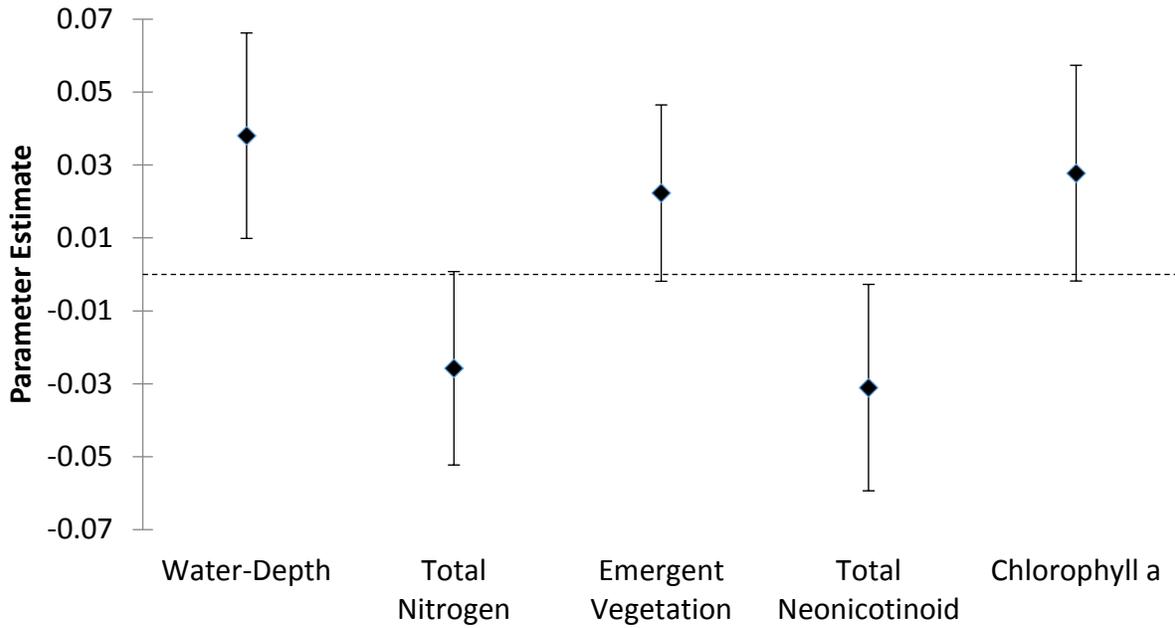


Figure 3.5 Model-averaged parameter estimates and associated 95% confidence intervals for parameters observed in competing model set for explaining invertebrate density. Model-averaged parameter estimates and standard errors were averaged over three competing models using AICc weights for nektonic communities (A), and three competing model for benthic communities (B).

A)



B)

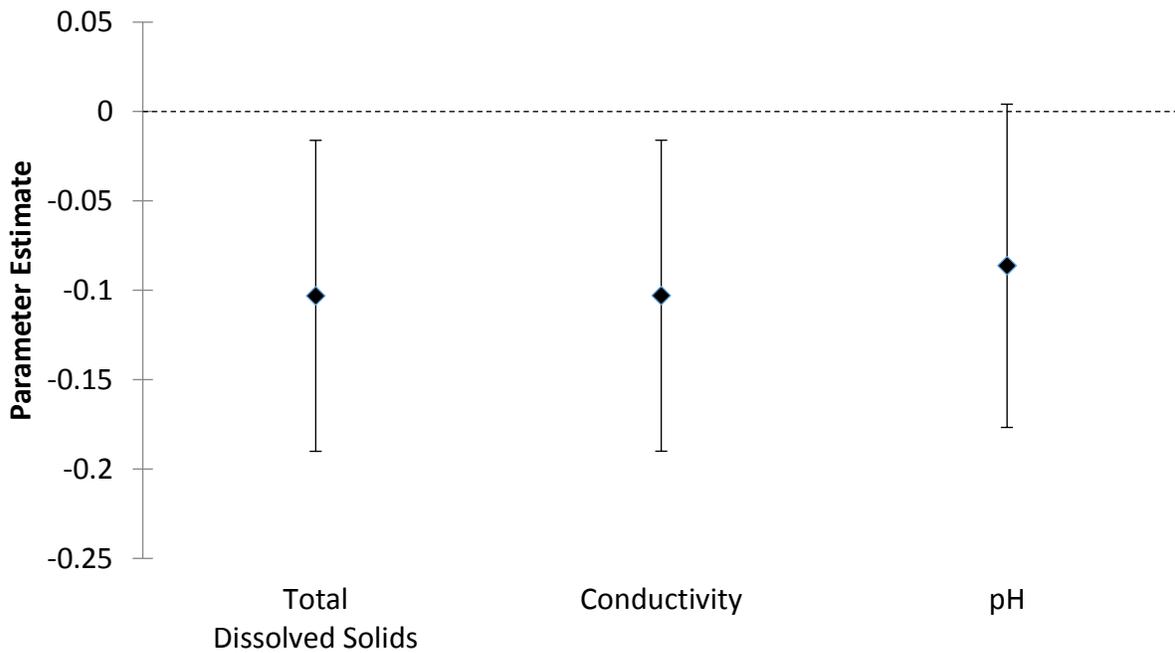
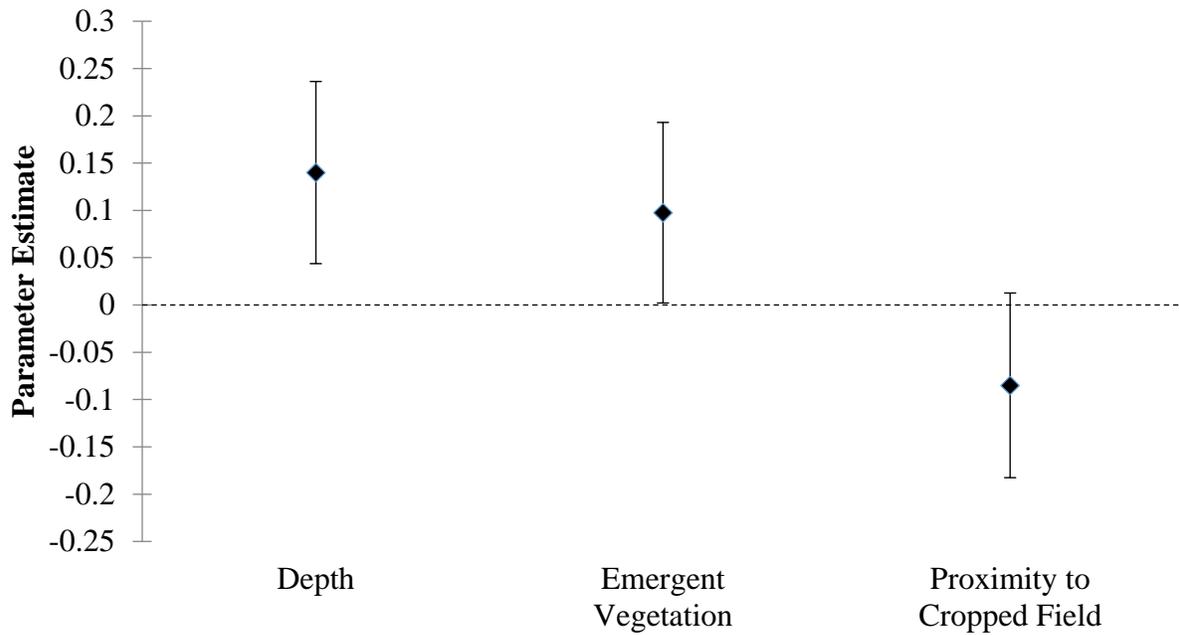
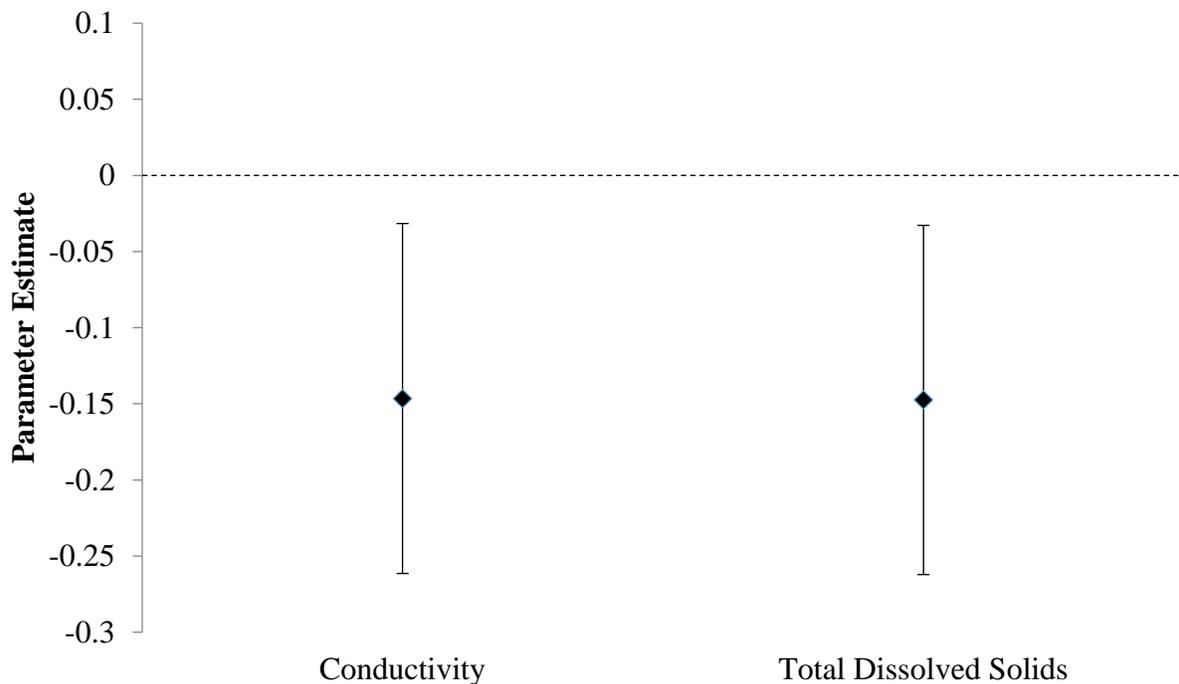


Figure 3.6 Model-averaged parameter estimates and associated 95% confidence intervals for parameters observed in competing model set for explaining invertebrate production. Model-averaged parameter estimates and standard errors were averaged over two competing models using AICc weights for nektonic communities (A), and two competing model for benthic communities (B).

A)



B)



APPENDIX

Appendix A: True metabolizable energy values (kcal/kg) used to estimate energy for seeds identified from core samples at wetlands in Nebraska's Rainwater Basin during spring 2014 and 2015.

Family	Binomial Name	Common	Stand	TME	Source
Poaceae	Echinochloa spp	Barnyard-Grass	Annual	2720	Sherfy 1999
	Panicum dichotomiflorum	Panic-Grass	Annual	2450	Straub 2008
	Setaria spp	Bristle-Grass	Annual	2880	Checkett et al. 2002
	Leptochloa fascicularis	Sprangle Top	Annual	2730	Straub 2008*
Polygonaceae	Polygonum spp	Smartweed	Annual	1590	Ballard et al. 2004
	Fagopyrum esculentum	Buckwheat	Annual	1450	Staub 2008*
Amaranthaceae	Amaranthus spp	Pigweed	Annual	2970	Checkett et al. 2002
Asteraceae	Ambrosia spp	Ragweed	Annual	1260	Straub 2008*
	Bidens spp	Bidens	Annual	550	Sherfy 1999
	Helianthus spp	Sunflower	Annual	905	Average of Asteraceae
	Iva spp	Sumpweed	Annual	905	Average of Asteraceae
Chenopodiaceae	Chenopodium spp	Lamb Quarters	Annual	2520	Dugger et al. 2007
Malvaceae	Abutilon theophrasti	Velvet Leaf	Annual	2190	Straub 2008*
	Sida spinosa	Prickly Fanpetals	Annual	2190	Straub 2008*
Sparganiaceae	Sparganium spp	Bur-Reed	Perennial	960	Straub 2008*
Poaceae	Andropogon spp	Blue stem	Perennial	2730	Straub 2008*
	Phalaris spp	Reed Canary	Perennial	2730	Straub 2008*
	Leersia oryzoides	Rice Cut Grass	Perennial	2910	Hoffman and Bookhout 1985
	Leersia virginica	White-Grass	Perennial	2910	Hoffman and Bookhout 1985 (Oryzoides TME)
	Poa pratensis	Blue-Grass	Perennial	2730	Straub 2008*
	Hordeum jubatum	Fox tail	Perennial	2730	Straub 2008*
	Paspalum spp	Knot-Grass	Perennial	1570	Straub 2008*
	Cyperaceae	Eleocharis spp	Spike Rush	Perennial	960
Carex spp		Sedge	Perennial	1360	Straub 2008*
Cyperus spp		Flat Sedge	Perennial	1690	Straub 2008*
Cyperus esculentus		Yellow Nutsedge	Perennial	1690	Straub 2008*
Scirpus spp		Bulrush	Perennial	930	Dugger et al. 2007
Polygonaceae	Polygonum coccineum	Longroot smartweed	Perennial	1590	Ballard et al. 2004
	Rumex Crispus	Curly Dock	Perennial	2680	Checkett et al. 2002
Alismataceae	Alisma plantago-aquatica	Water Plantain	Perennial	3060	Hoffman and Bookhout 1985 (Sargittaria TME)

Appendix A: Continued

Alismataceae	Sagittaria spp	Arrowhead	Perennial	3060	Hoffman and Bookhout 1985
Asteraceae	Symphyotrichum spp	Aster	Perennial	1260	Straub 2008*
Ceratophyllaceae	Ceratophyllum spp	Coons Tail	Perennial	1980	Straub 2008*
Haloragaceae	Myriophyllum spp	watermilfoil	Perennial	610	Straub 2008*
Potamogetonaceae	Potamogeton	Pondweed	Perennial	1420	Ballard et al. 2004
Zannichelliaceae	Zannichellia palustris	Horned Pondweed	Perennial	3200	Straub 2008*
Marsileaceae	Marsilea vestita	Hairy Waterclove	Perennial	2500	This Study*
Poaceae	Sorghum bicolor	Sorghum (RC)	WCrop	2730	Straub 2008
	Sorghum halepense	Johnson-Grass (RC)	WCrop	2730	Straub 2008
	Zea Mays	Maize	WCrop	3670	Reineke et al. 1989
	Triticum aestivum	Common Wheat	WCrop	3460	Reineke et al. 1989
Fabaceae	Glycine max	Soy Beans	WCrop	2650	Reineke et al. 1989

*True Metabolizable Energy assigned using regression equation developed by Straub 2008.

Appendix B: Length-mass regression equations based on total body length. Equations are of the form dry-mass (mg) = $a \cdot L^b$, where L = body length (mm), and a and b are fitted constants. Intended Taxon is the taxon/group that a length-mass regression was constructed. For more details, see Benke et al. 1999.

Order	Family	a	b	Intended Taxon	Source
Amphipoda	Gammaridae	0.012	2.740	Gammaridae	Basset and Glazier 1995
Anostraca	Artemiidae	0.279	1.363	Streptocephalidae	Nagorskaya et al. 2004
	Branchinectidae	0.279	1.363	Streptocephalidae	Nagorskaya et al. 2004
Cladocera	Daphniidae	0.009	2.630	Daphniidae	Burns 1969
Coleoptera	Anthicidae	0.008	2.910	Coleoptera Average	Benke and Huryn 2006
	Carabidae	0.008	2.910	Coleoptera Average	Benke and Huryn 2006
	Curculionidae	0.013	2.180	Curculionidae	Smock 1980
	Dytiscidae	0.062	2.502	Dytiscidae	Smock 1980
	Gyrinidae	0.008	2.910	Coleoptera Average	Benke and Huryn 2006
	Haliplidae	0.008	2.910	Coleoptera Average	Benke and Huryn 2006
	Hydrophilidae	0.008	2.910	Coleoptera Average	Benke and Huryn 2006
Copepoda	Copepoda	0.000	2.408	Copepoda	Leeper and Taylor 1998
Diptera	Cecidomyiidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Ceratopogonidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Chaoboridae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Chironomidae	0.002	2.617	Diptera Average	Benke et al. 1999
	Culicidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Empididae	0.007	2.436	Empididae	Benke et al. 1999
	Muscidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Stratiomyiidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Tabanidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
	Tipulidae	0.003	2.692	Diptera Average	Benke and Huryn 2006
Entomobryomorpha	Isotomidae	0.006	2.809	Isotomidae	Gruner 2003
Ephemeroptera	Ephemerellidae	0.007	2.832	Ephemeroptera Average	Benke and Huryn 2006
	Leptohyphidae	0.007	2.832	Ephemeroptera Average	Benke and Huryn 2006
	Potomanthidae	0.007	2.832	Ephemeroptera Average	Benke and Huryn 2006
Gastropoda	Bithyniidae	0.050	2.026	Physidae	This Study (Appendix C)
	Physidae	0.050	2.026	Physidae	This Study (Appendix C)

Appendix B: Continued

	Planorbidae	0.070	2.208	Planorbidae	This Study (Appendix C)
Haplotaxida	Haplotaxida	0.002	1.875	Haplotaxida	Stoffels et al. 2003
Hemiptera	Corixidae	0.003	2.904	Corixidae	Smock 1980
	Notonectidae	0.003	2.904	Corixidae	Smock 1980
	Pentatomidae	0.003	2.904	Corixidae	Smock 1980
	Pyralidae	0.003	2.918	Pyralidae	Benke et al. 1999
Lepidoptera	Aeshnidae	0.008	2.813	Aeshnidae	Benke et al. 1999
Odonata	Coenagrionidae	0.005	2.785	Coenagrionidae	Benke et al. 1999
	Gomphidae	0.009	2.787	Gomphidae	Benke et al. 1999
	Libellulidae	0.008	2.809	Libellulidae	Benke et al. 1999
	Lestidae	0.005	2.785	Coenagrionidae	Benke et al. 1999
	Glossosomatidae	0.008	2.958	Glossosomatidae	Benke et al. 1999
Trichoptera	Leptoceridae	0.003	3.212	Leptoceridae	Benke et al. 1999
	Sphaeriidae	0.016	2.477	Sphaeriidae	Benke et al. 1999
Veneroida	Ostracoda	0.016	2.477	Sphaeriidae	Benke et al. 1999
(Class)	Nematoda	0.006	0.964	Nematoda	This Study (Appendix C)
(Phylum)	Hirudinea	0.012	2.037	Hirudinea	This Study (Appendix C)
(Sub Class)					

Appendix C: Length-mass regression equations based on methods from Benke et al. 1999. The power equation of the form $DM = aL^b$ was used to explain density where DM = dry mass (mg), L = body length (mm), and a and b are fitted constants. Measurement = the method of measurement used to determine length, n = the number of specimens included in regression model, and range = range of body lengths (mm) and mass (mg) of individuals included. A shell measurement was used to determine the length of families in class Gastropoda, however the shell was discarded when determining an individual's mass.

Taxon	Measurement	n	Range (mm)	Range (mg)	a + SE	b + SE	R ²
Gastropoda							
Physidae/Bithyniidae	Shell Length	133	3 - 33	0.20 - 47.40	0.05 ± <0.01	2.03 ± 0.08	0.94
Planorbidae	Shell Length	61	3 - 23	0.85 - 67.60	0.07 ± <0.01	2.21 ± 0.10	0.91
Hirudinea	Body Length	91	3 - 28	0.11 - 17.70	0.01 ± <0.01	2.04 ± 0.11	0.89
Nematoda	Body Length	166	2 - 5	0.01 - 0.03	0.01 + <0.01	0.96 + 0.09	0.78

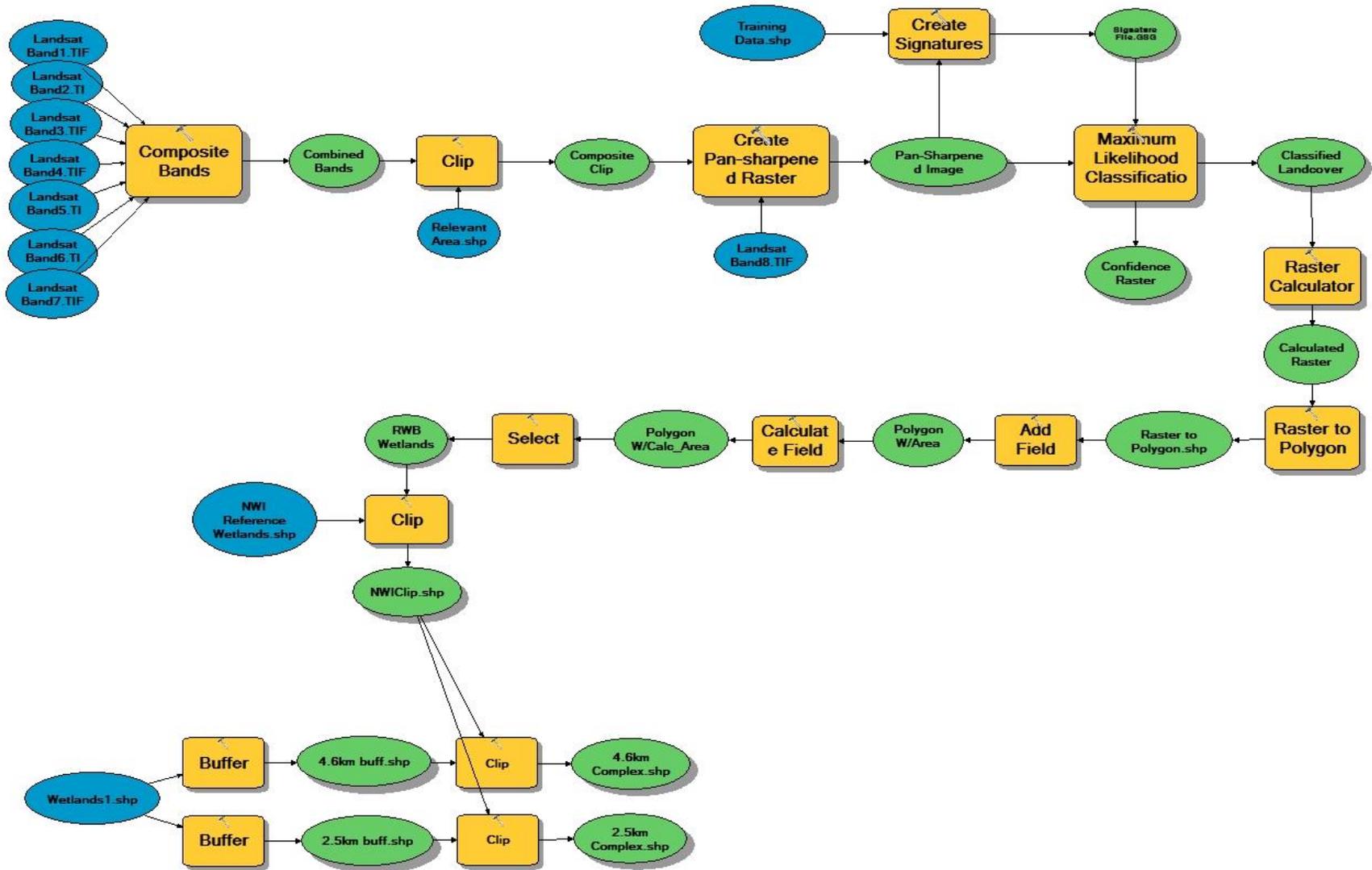
Appendix D.1: Landsat 8 Operational Land Imager (OLI) and Thermal Infrared Sensor (TIRS) data sets were acquired from www.earthexplorere.usgs.gov. Perimeter Date = date that a wetlands area was ground proofed, Imagery Date = the date that satellite imagery was recorded, Actual Area = known area from ground proofing, Projected Area= area estimated from computer processing, and % Difference = difference between known and projected areas. Supplemental satellite imagery was obtained from Google Earth on 17 April 2014, and multispectral orthophotography collected by aircraft during the weeks of March 9, 2014, and March 8, 2015.

Site	Perimeter Date	Imagery Date	Imagery Type	Path/Row	Actual Area	Proj_Area	% Difference
Alt01	3/9/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	18.11	16.53	9%
Alt01	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	12.64	11.75	7%
Alt02	2/18/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	2.67	2.42	10%
Alt03	2/19/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	6.47	8.57	28%
Alt03	3/23/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	4.37	5.21	17%
WRP01	3/26/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	2.12	2.20	4%
WRP05	2/25/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	2.06	2.79	30%
WRP08	2/28/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	9.15	10.25	11%
WRP08	3/26/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	6.49	6.10	6%
WRP09	3/11/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	5.29	4.66	13%
WRP10	3/8/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	8.91	6.50	31%
WRP10	3/26/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	5.13	4.90	5%
WRP11	3/8/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	14.99	15.39	3%
WRP11	4/2/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	11.11	10.58	5%
WRP12	3/17/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	14.42	14.63	1%
WRP14	3/8/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	12.00	8.40	35%
WRP14	3/28/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	7.50	7.35	2%
WRP15	3/8/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	0.76	0.35	74%
WRP17	2/28/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	2.19	2.24	2%
WRP18	3/11/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	8.22	7.61	8%
WRP18	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	6.03	5.96	1%
Eckhardt	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	16.88	17.47	3%
Kissinger	3/2/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	16.75	23.80	35%

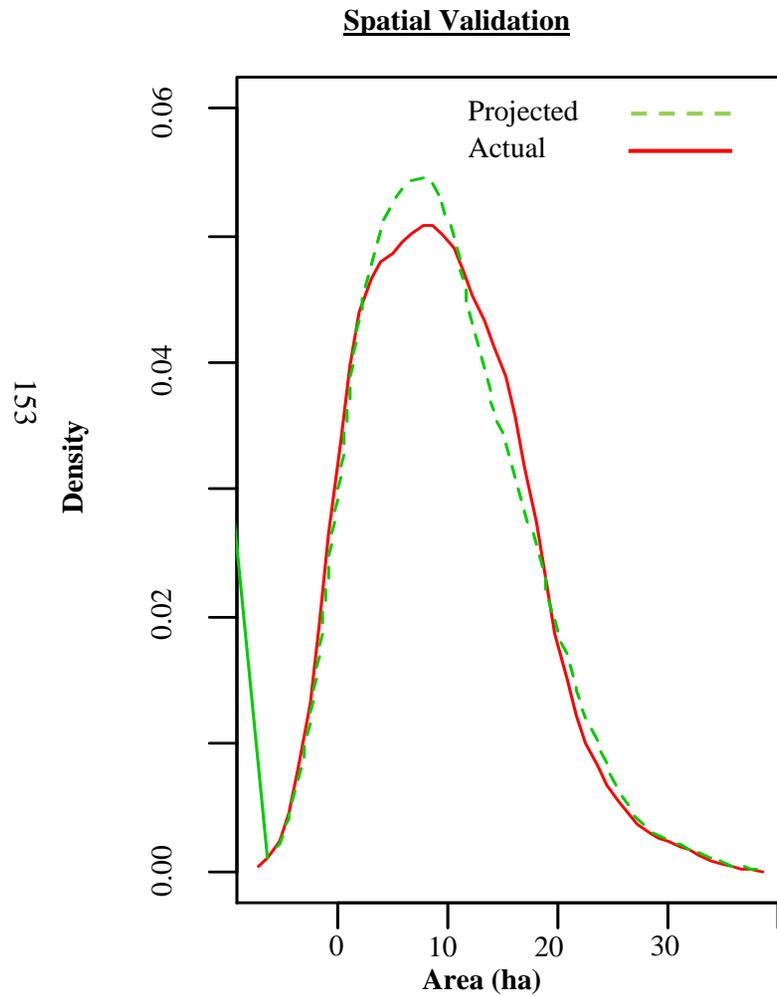
<i>Appendix D.1: continued</i>							
Kissinger	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	13.41	11.93	12%
Greenhead	3/2/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	15.29	16.89	10%
Greenhead	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	15.81	16.90	7%
Sandpiper	2/19/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	11.02	12.07	9%
Sandpiper	3/29/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	11.11	10.23	8%
Sora	2/25/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	13.74	19.69	36%
Sora	3/23/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	8.28	10.37	22%
Marsh Duck	3/30/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	22.89	20.51	11%
SpikeRush	2/23/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	11.43	15.99	33%
SpikeRush	3/23/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	4.56	4.24	7%
Tamora	2/21/2015	2/24/2015	Landsat: LC80290322015055LGN00	29/32	24.61	24.83	1%
Tamora	3/27/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	17.43	17.63	1%
Shypoke	3/6/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	8.35	8.51	2%
Shypoke	3/23/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	6.82	6.60	3%
DeepWell	3/12/2015	3/12/2015	Landsat: LC80290322015071LGN00	29/32	8.76	9.28	6%
DeepWell	3/27/2015	3/28/2015	Landsat: LC80290322015087LGN00	29/32	8.03	7.95	1%
Eckhardt	2/25/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	15.50	15.60	1%
Eckhardt	3/24/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	20.50	19.30	6%
BullRush	2/19/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	16.60	21.10	24%
BullRush	3/24/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	18.00	16.50	9%
Sand	2/19/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	9.69	11.08	13%
Sand	3/25/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	8.86	9.21	4%
Sora	2/25/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	5.75	5.04	13%
Sora	3/25/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	22.97	22.82	1%
WRP05	2/28/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	1.76	3.26	60%
WRP05	3/26/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	1.98	2.89	38%
WRP01	2/27/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	1.01	2.51	85%
WRP01	3/27/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	1.15	1.14	1%
Renquist	2/27/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	2.63	2.74	4%

<i>Appendix D.1: continued</i>							
Spike	2/27/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	0.69	0.96	33%
Spike	3/27/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	0.61	0.54	12%
Freeman	2/24/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	30.22	30.82	2%
Freeman	3/26/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	16.36	10.69	42%
Tamora	2/24/2014	2/21/2014	Landsat: LC80290322014052LGN00	29/32	1.26	1.90	41%
Tamora	3/26/2014	3/25/2014	Landsat: LC80290322014084LGN00	29/32	1.00	1.16	15%
Bergeron	2/26/2014	2/19/2014	Landsat: LC80310312014050LGN00	31/31	10.95	12.17	11%
Bergeron	4/14/2014	4/17/2014	Landsat: LC80300312014107LGN00	31/31	10.02	9.41	6%
RC02	2/26/2014	2/19/2014	Landsat: LC80310312014050LGN00	31/31	3.63	4.74	27%
RC02	4/14/2014	4/17/2014	Landsat: LC80300312014107LGN00	31/31	1.78	2.10	17%
RC03	2/26/2014	2/19/2014	Landsat: LC80310312014050LGN00	31/31	1.27	1.29	1%
RC03	4/14/2014	4/17/2014	Landsat: LC80300312014107LGN00	31/31	1.16	0.80	37%

Appendix D.2: Model Builder function used to manage, develop and edit spatial workflow.



Appendix D.3: Figure on left shows a Kernel density plot comparing the actual inundated area of a wetland obtained by walking the wetted edge with a handheld GPS, against projected area that was estimated from aerial and satellite imagery. Table on the right shows results of a paired t-test that compares actual wetland area against projected wetland area. There were no statistical differences between actual and projected areas (p value > 0.05).



t-Test: Paired Two Sample for Means

	<i>Actual</i>	<i>Projected</i>
Mean	9.49	9.60
Variance	44.25	45.33
Observations	77	77
Pearson Correlation	0.95	
Hypothesized Mean Difference	0	
df	76	
95% Confidence Level Upper	0.37	
95% Confidence Level Lower	-0.59	
t Stat	-0.45	
p-value	0.65	

Appendix E: Mean density (birds/ha) of waterfowl (Anseriformes) species observed during weekly surveys at 17 wetlands in Nebraska's Rainwater basin, 23 February-26 April 2014.

SPECIES		Week 1	Week 2	Week 3	Week 4	Week 5	Week 6	Week 7	Week 8	Week 9
Anatinae		17.10	45.98	121.64	140.29	162.18	60.64	79.45	27.69	21.75
American wigeon	<i>Anas americana</i>	0.08	0.24	0.86	19.34	13.49	3.56	9.62	1.43	0.65
Blue-winged teal	<i>Anas discors</i>	-	-	-	0.27	0.26	3.98	7.52	6.55	9.72
Gadwall	<i>Anas strepera</i>	-	-	0.07	6.38	2.36	1.66	5.83	1.34	0.56
Green-winged teal	<i>Anas carolinensis</i>	0.33	0.07	7.28	18.48	36.14	21.15	32.05	6.52	2.43
Mallard	<i>Anas platyrhynchos</i>	12.31	32.77	66.96	48.44	50.25	8.37	13.72	6.84	4.95
Northern pintail	<i>Anas acuta</i>	4.38	12.91	48.44	46.27	57.71	18.63	7.84	0.76	0.51
Northern shoveler	<i>Anas clypeata</i>	-	-	0.12	1.08	1.95	3.35	2.75	4.21	2.92
Wood duck	<i>Aix sponsa</i>	-	-	-	0.03	-	< 0.01	0.13	0.04	0.01
Aythiinae		-	-	1.49	0.42	0.44	0.71	0.20	0.10	0.12
Bufflehead	<i>Bucephala albeola</i>	-	-	0.07	0.13	0.04	0.08	-	-	0.05
Canvasback	<i>Aythya valisineria</i>	-	-	0.06	0.03	-	0.12	0.08	-	-
Lesser scaup	<i>Aythya affinis</i>	-	-	-	0.05	0.03	0.42	0.03	-	-
Redhead	<i>Aythya americana</i>	-	-	1.27	0.14	0.26	0.09	0.01	-	0.05
Ring-necked duck	<i>Aythya collaris</i>	-	-	0.16	0.07	0.13	< 0.01	0.04	0.01	-
Ruddy duck	<i>Oxyura jamaicensis</i>	-	-	-	-	-	-	0.04	0.09	0.02
Anserinae		-	2.80	53.31	1.15	0.11	0.61	0.03	-	-
Cackling goose	<i>Branta hutchinsii</i>	-	-	0.74	0.00	-	-	-	-	-
Canada goose	<i>Branta canadensis</i>	-	0.01	7.94	1.12	0.08	0.37	0.01	-	-
Snow goose	<i>Chen caerulescens</i>	-	2.42	43.16	0.02	0.01	0.24	0.02	-	-
White-fronted goose	<i>Anser albifrons</i>	-	0.16	1.28	0.01	0.02	-	-	-	-
Ross's goose	<i>Chen rossii</i>	-	0.21	0.22	-	-	-	-	-	-

Appendix F: Mean density (birds/ha) of waterfowl (Anseriformes) species observed during weekly surveys at 24 wetlands in Nebraska's Rainwater basin, 22 February-18 April 2015.

SPECIES		Week 1	Week 2	Week 3	Week 4	Week 5	Week 6	Week 7	Week 8
Anatinae		-	1.17	43.29	43.44	30.96	21.98	30.25	14.88
American wigeon	<i>Anas americana</i>	-	-	0.22	2.65	1.49	1.65	3.76	0.13
Blue-winged teal	<i>Anas discors</i>	-	-	0.01	0.02	0.04	3.09	4.56	4.46
Gadwall	<i>Anas strepera</i>	-	-	0.05	0.40	0.49	0.71	0.41	0.19
Green-winged teal	<i>Anas carolinensis</i>	-	0.06	1.57	4.12	9.53	7.75	13.89	5.16
Mallard	<i>Anas platyrhynchos</i>	-	0.81	21.94	20.52	13.28	5.90	4.52	2.32
Northern pintail	<i>Anas acuta</i>	-	0.30	19.45	15.52	5.17	1.63	0.92	0.22
Northern shoveler	<i>Anas clypeata</i>	-	-	0.05	0.21	0.95	1.25	2.20	2.40
Wood duck	<i>Aix sponsa</i>	-	-	-	0.01	-	0.02	-	0.01
Aythiinae		-	-	0.11	1.29	0.32	0.25	0.03	0.08
Bufflehead	<i>Bucephala albeola</i>	-	-	-	-	-	< 0.00	-	-
Canvasback	<i>Aythya valisineria</i>	-	-	0.06	0.12	-	0.16	-	-
Lesser scaup	<i>Aythya affinis</i>	-	-	-	0.15	0.01	0.00	-	-
Redhead	<i>Aythya americana</i>	-	-	0.03	0.68	0.12	0.01	0.02	0.08
Ring-necked duck	<i>Aythya collaris</i>	-	-	0.01	0.34	0.18	0.07	0.02	-
Ruddy duck	<i>Oxyura jamaicensis</i>	-	-	-	-	0.01	-	-	-
Anserinae		-	0.05	30.01	21.49	0.31	0.27	0.08	0.02
Canada goose	<i>Branta canadensis</i>	-	0.05	1.63	0.54	0.13	-	-	0.01
Snow goose	<i>Chen caerulescens</i>	-	-	27.73	20.20	0.16	0.27	0.08	0.01
White-fronted goose	<i>Anser albifrons</i>	-	-	0.64	0.75	0.03	-	-	-
Ross's goose	<i>Chen rossii</i>	-	-	-	0.01	-	-	-	-

Appendix G: Chapter I summary of local and landscape variables, and dabbling duck response (Density and Richness) among wetlands during springs 2014-2015 in Nebraska's Rainwater Basin. Data were collected from 17 wetlands in 2014 and 24 wetlands in 2015.

Year	Metric	Mean	Std Error	Std Dev	Median	Min	Max	CI (95%)
2014	Density	75.72	17.40	71.75	40.61	0.05	240	36.90
	Richness	4.44	0.24	2.53	5.00	0.00	8.00	0.47
	Invert (kg/ha)	10.26	2.09	8.60	7.69	0.27	33.20	4.42
	Seed (kg/ha)	442.57	87.23	359.64	441.46	20.70	1459.09	184.91
	TME (kcal/ha)	1793.49	92.00	379.30	1770.32	1152.82	2516.89	195.02
	Energy (kcal/ha)	747,586	153,161	631,498	7,52,521	40,502	2,681,801	324,686
	Area (ha)	17.18	2.28	28.23	9.60	0.30	131.80	4.51
	Depth (cm)	13.34	0.62	7.61	12.38	0.55	30.00	1.22
	Veg (%)	0.28	0.02	0.22	0.25	0.00	0.65	0.02
	5kmComp	19.93	3.10	33.66	3.61	0.00	155.33	6.14
	5kmDen	1.70	0.16	1.74	1.50	0.00	5.00	0.32
	2kmComp	4.19	0.70	7.63	0.00	0.00	45.75	1.39
	2kmDen	0.78	0.10	1.04	0.00	0.00	4.00	0.19
2015	Density	22.85	3.19	15.64	21.69	0.00	61.60	6.60
	Richness	3.88	0.17	1.93	4.00	0.00	8.00	0.33
	Invert (kg/ha)	16.17	1.99	9.73	14.39	1.34	44.34	4.65
	Seed (kg/ha)	614.23	59.79	292.90	615.93	34.07	1462.27	123.68
	TME (kcal/ha)	1679.36	60.52	296.48	1718.48	1129.17	2259.07	125.19
	Energy (kcal/ha)	1,046,096	130,549	639,555	997,357	62,125	3,303,381	270,060
	Area (ha)	10.20	0.50	6.88	9.33	0.20	35.32	0.98
	Depth (cm)	14.68	0.55	7.68	14.62	1.37	36.53	1.09
	Veg (%)	0.26	0.01	0.20	0.23	0.00	0.90	0.03
	5kmComp	46.49	4.98	58.98	25.64	0.00	278.99	9.86
	5kmDen	2.99	0.21	2.50	3.00	0.00	9.00	0.42
	2kmComp	21.26	2.68	31.74	10.52	0.00	167.10	5.30
	2kmDen	1.44	0.11	1.29	1.00	0.00	5.00	0.22

Appendix H: Mean dry density (kg/ha) of all nektonic aquatic invertebrate taxa observed in bimonthly net samples collected at wetlands in 2014 (n=17), and 2015 (n=24) in Nebraska's Rainwater Basin.

Order	Family	2014 Nektonic			2015 Nektonic			
		Mean (kg/ha)	SE	% Comp	Mean (kg/ha)	SE	% Comp	
Amphipoda	Gammaridae	0.01	0.00	0.00	0.04	0.01	0.01	
Anostraca	Artemiidae	0.38	0.06	0.21	0.26	0.17	0.04	
Cladocera	Daphniidae	0.04	0.01	0.02	0.12	0.05	0.02	
Coleoptera	Anthicidae	<0.01	<0.01	<0.01	-	-	-	
	Carabidae	0.01	0.01	0.00	-	-	-	
	Curculionidae	0.00	0.00	0.00	-	-	-	
	Dytiscidae	0.14	0.03	0.08	0.16	0.10	0.03	
	Gyrinidae	0.00	0.00	0.00	0.00	0.00	0.00	
	Haliplidae	-	-	-	0.01	0.00	0.00	
	Hydrophilidae	0.00	0.00	0.00	0.02	0.01	0.00	
	Copepoda	Copepoda	0.41	0.07	0.23	1.47	0.13	0.23
	Diptera	Cecidomyiidae	0.00	0.00	0.00	-	-	-
Ceratopogonidae		0.01	0.00	0.00	0.05	0.00	0.01	
Chaoboridae		-	-	-	0.21	0.08	0.03	
Chironomidae		0.05	0.00	0.03	0.78	0.12	0.12	
Culicidae		<0.01	<0.01	<0.01	-	-	-	
Empididae		<0.01	<0.01	<0.01	0.01	0.00	0.00	
Muscidae		0.00	0.00	0.00	-	-	-	
Stratiomyiidae		0.00	0.00	0.00	0.00	0.00	0.00	
Tabanidae		0.00	0.00	0.00	0.00	0.00	0.00	
Tipulidae		0.00	0.00	0.00	-	-	-	
Entomobryomorpha		Isotomidae	<0.01	<0.00	<0.01	0.00	0.00	0.00
Ephemeroptera	Ephemerellidae	-	-	-	0.00	0.00	0.00	
	Leptohephidae	-	-	-	0.03	0.01	0.00	
	Potamanthidae	-	-	-	0.00	0.00	0.00	
Gastropoda	Bithyniidae	0.03	0.02	0.01	0.08	0.03	0.01	

Appendix H Continued

Order	Family	2014 Nektonic			2015 Nektonic		
		Mean (kg/ha)	SE	% Comp	Mean (kg/ha)	SE	% Comp
	Physidae	0.19	0.14	0.10	0.23	0.10	0.04
	Planorbidae	0.09	0.05	0.05	0.30	0.08	0.05
Haplotaxida	Haplotaxida	0.09	0.01	0.05	0.24	0.01	0.04
Hemiptera	Corixidae	0.02	0.00	0.01	0.16	0.03	0.03
	Notonectidae	0.01	0.00	0.01	0.05	0.01	0.01
	Pentatomidae	-	-	-	0.00	0.00	0.00
Lepidoptera	Pyralidae	-	-	-	0.00	0.00	0.00
Odonata	Aeshnidae	0.07	0.06	0.04	0.35	0.20	0.06
	Coenagrionidae	0.03	0.01	0.02	0.93	0.10	0.15
	Gomphidae	0.01	0.01	0.01	0.10	0.03	0.02
	Libellulidae	0.00	0.00	0.00	0.04	0.02	0.01
	Lestidae	-	-	-	0.02	0.00	0.00
Trichoptera	Glossosomatidae	-	-	-	-	-	-
	Leptoceridae	-	-	-	0.00	0.00	0.00
Veneroida	Sphaeriidae	0.08	0.02	0.04	0.05	0.01	0.01
(Class)	Ostracoda	0.01	0.00	0.00	0.03	0.00	0.00
(Phylum)	Nematoda	0.08	0.01	0.04	0.28	0.06	0.05
(Sub Class)	Hirudinea	0.06	0.02	0.03	0.10	0.03	0.02

Appendix I: Mean dry density (kg/ha) of all benthic aquatic invertebrate taxa observed in bimonthly core samples collected at wetlands in 2014 (n=17), and 2015 (n=24) in Nebraska's Rainwater Basin

Order	Family	2014 Benthic			2015 Benthic		
		Mean (kg/ha)	SE	% Comp	Mean (kg/ha)	SE	% Comp
Amphipoda	Gammaridae	0.01	0.01	0.00	-	-	-
Anostraca	Artemiidae	0.14	0.05	0.02	-	-	-
Blattodea	Blattodea	<0.01	0.00	<0.01	<0.01	-	<0.01
Cladocera	Daphniidae	0.04	0.01	0.00	<0.01	-	<0.01
Coleoptera	Carabidae	0.09	0.05	0.01	-	-	-
Coleoptera	Curculionidae	<0.01	0.00	<0.01	-	-	-
Coleoptera	Hydrophilidae	0.68	0.52	0.08	-	-	-
Copepoda	Copepoda	1.81	0.68	0.21	0.07	0.02	0.01
Diptera	Cecidomyiidae	0.01	0.01	0.00	<0.01	-	<0.01
Diptera	Ceratopogonidae	0.07	0.06	0.01	0.05	0.02	0.01
Diptera	Chaoboridae	-	-	-	0.05	0.03	0.01
Diptera	Chironomidae	0.32	0.08	0.04	1.08	0.11	0.11
Diptera	Cyclorrhapha	<0.01	0.00	<0.01	-	-	-
Diptera	Muscidae	<0.01	0.00	<0.01	-	-	-
Diptera	Tabanidae	<0.01	0.01	<0.01	-	-	-
Gastropoda	Bithyniidae	0.44	0.22	0.05	0.03	0.05	0.00
Gastropoda	Physidae	0.36	0.31	0.04	0.10	0.01	0.01
Gastropoda	Planorbidae	0.74	0.18	0.09	1.02	0.57	0.10
Haplotaxida	Naididae	1.81	0.28	0.21	3.13	0.52	0.32
Hemiptera	Corixidae	0.02	0.01	0.00	-	-	-
Lepidoptera	Pyralidae	-	-	-	<0.01	-	<0.01
Odonata	Coenagrionidae	<0.01	0.01	<0.01	0.15	0.16	0.02
Odonata	Libellulidae	-	-	-	0.37	0.24	0.04
Veneroida	Sphaeriidae	0.17	0.08	0.02	<0.01	-	<0.01
(Class)	Ostracoda	0.06	0.03	0.01	<0.01	-	<0.01
(Phylum)	Nematoda	0.63	0.09	0.07	0.85	0.10	0.09
(Sub Class)	Hirudinea	1.05	0.44	0.12	2.95	0.28	0.30

Appendix J: True metabolizable energy (TME), Mean dry seed density (kg/ha), and energy (kcal/ha) of seeds collected at public wetlands in 2014 (n=12), and 2015 (n=12). Table accounts for >98% of the seeds observed in this study.

Seed Composition at Public Wetlands in Nebraska's Rainwater Basin

Food Item	2014 Public Seed Composition					2015 Public Seed Composition				
	TME	kg/ha	kcal	%kg/ha	%kcal/ha	TME	kg/ha	kcal	%kg/ha	%kcal/ha
<i>Amaranthus spp</i> ^a	2970	8.2	24313.0	1%	3%	2970	4.8	14165.4	1%	1%
<i>Echinochloa spp</i> ^a	2720	67.9	184789.5	12%	21%	2720	130.2	354091.9	19%	29%
<i>Helianthus spp</i> ^a	2500	5.3	13168.7	1%	1%	2500	2.1	5227.3	< 1%	< 1%
<i>Leptochloa fascicularis</i> ^a	2730	28.7	78357.2	5%	9%	2730	36.9	100646.3	5%	8%
<i>Panicum dichotomiflorum</i> ^a	2450	2.1	5186.3	< 1%	1%	2450	3.4	8402.7	< 1%	1%
<i>Polygonum spp</i> ^a	1590	146.2	232390.9	27%	26%	1590	189.5	301331.6	27%	25%
<i>Alisma plantago-aquatica</i> ^p	3060	1.5	4499.4	< 1%	1%	3060	8.1	24827.0	1%	2%
<i>Eleocharis spp</i> ^p	960	5.1	4853.1	1%	1%	960	7.9	7569.7	1%	1%
<i>Marsilea vestita</i> ^p	2050	5.8	11832.0	1%	1%	2050	31.0	63479.0	4%	5%
<i>Polygonum coccineum</i> ^p	1590	19.2	30607.4	4%	3%	1590	0.8	1235.5	0%	0%
<i>Potamogeton</i> ^p	1420	96.5	137083.9	18%	15%	1420	97.8	138868.6	14%	11%
<i>Sagittaria spp</i> ^p	3060	2.8	8673.0	1%	1%	3060	7.9	24053.7	1%	2%
<i>Scirpus spp</i> ^p	930	60.1	55939.2	11%	6%	930	94.2	87572.3	14%	7%
<i>Sparganium spp</i> ^p	960	86.1	82619.7	16%	9%	960	73.4	70456.4	11%	6%
<i>Glycine max</i> ^c	2650	2.6	6851.9	< 1%	1%	2650	-	-	-	-
Annual Communities	2082	260.5	542578.3	48%	60%	2136	370.2	790945.1	53%	65%
Perennial Communities	1227	281.2	345179.8	51%	38%	1312	324.2	425327.0	47%	35%
Cropped Communities	2650	2.6	6851.9	< 1%	1%	-	-	-	-	-
Total	1652	546.1	897520.4	100%	100%	1665	695.1	1217317.3	100%	100%

¹Detailed descriptions of estimates for true metabolizable energy (TME) can be reviewed in Appendix A.

^aAnnual plant species

^pPerennial plant species

^cCropped plant species

Appendix K: True metabolizable energy (TME), Mean dry seed density (kg/ha), and energy (kcal/ha) of seeds collected at WRP wetlands in 2014 (n=10), and 2015 (n=11). Table accounts for >98% of the seeds observed in this study.

Seed Composition at WRP Wetlands in Nebraska's Rainwater Basin

Food Item	2014 Seed Composition					2015 Seed Composition				
	TME ¹	kg/ha	kcal/ha	%kg/ha	%kcal/ha	TME ¹	kg/ha	kcal/ha	%kg/ha	%kcal/ha
<i>Abutilon theophrasti</i> ^a	2190	-	-	-	-	2190	4.4	9640.3	1%	1%
<i>Amaranthus spp</i> ^a	2970	3.2	9477.8	1%	1%	2970	3.6	10703.9	1%	1%
<i>Ambrosia spp</i> ^a	1260	9.2	11611.1	2%	1%	1260	2.5	3088.5	<1%	<1%
<i>Echinochloa spp</i> ^a	2720	52	141543.9	9%	14%	1590	148.2	235602.2	27%	25%
<i>Helianthus spp</i> ^a	2500	3.2	7877.7	1%	1%	2500	0.7	1629.8	<1%	<1%
<i>Iva spp</i> ^a	2500	1.3	3136.7	< 1%	< 1%	2500	7.6	19068.3	1%	2%
<i>Leptochloa fascicularis</i> ^a	2730	9	24599.9	2%	2%	2730	6.2	16823.1	1%	2%
<i>Panicum dichotomiflorum</i> ^a	2450	13.4	32907.6	2%	3%	2450	8.4	20584.8	2%	2%
<i>Polygonum spp</i> ^a	1590	141.9	225626.5	24%	22%	1590	166.6	264911.7	31%	29%
<i>Alisma plantago-aquatica</i> ^p	3060	0.1	185.6	< 1%	< 1%	3060	12.3	37769.2	2%	4%
<i>Cyperus spp</i> ^p	1690	-	-	-	-	1690	7.5	12711.0	1%	1%
<i>Eleocharis spp</i> ^p	960	59	56623.6	10%	6%	960	22.0	21151.5	4%	2%
<i>Leersia virginica</i> ^p	2910	0.7	2050.6	< 1%	< 1%	2910	1.8	5281.1	< 1%	1%
<i>Marsilea vestita</i> ^p	2050	104.6	214446.3	18%	21%	2050	90.8	186117.7	17%	19%
<i>Polygonum coccineum</i> ^p	1590	17.4	27742	3%	3%	1590	8.4	13381.7	2%	1%
<i>Potamogeton</i> ^p	1420	15.2	21646.3	3%	2%	1420	28.0	39820.9	5%	4%
<i>Rumex Crispus</i> ^p	2680	32.2	86325.9	5%	9%	2680	3.7	9942.8	< 1%	1%
<i>Sagittaria spp</i> ^p	3060	9	27600.8	2%	3%	3060	3.2	9715.5	< 1%	1%
<i>Scirpus spp</i> ^p	930	32.3	30024.9	5%	3%	930	30.1	27984.0	6%	3%
<i>Sparganium spp</i> ^p	960	87.7	84153	15%	8%	960	69.4	66641.8	13%	7%
Annual Communities	1956	234.3	458352.7	39%	45%	2041	260.5	531679.5	48%	56%
Perennial Communities	1541	359.9	554658.9	61%	55%	1549	279.1	432308.4	52%	44%
Total	1832	594.2	1013172	100%	100%	1741	539.7	956446.0	100%	100%

¹Detailed descriptions of estimates for true metabolizable energy (TME) can be reviewed in Appendix A.

^aAnnual plant species

^pPerennial plant species

Appendix L: True metabolizable energy (TME), Mean dry seed density (kg/ha), and energy (kcal/ha) of seeds collected at cropped wetlands in 2014 (n=10), and 2015 (n=10). Table accounts for >98% of the seeds observed in this study.

Seed Composition at Cropped Wetlands in Nebraska's Rainwater Basin										
Food Item	2014 Seed Composition					2015 Seed Composition				
	TME	kg/ha	kcal/ha	% kg/ha	% kcal/ha	TME	kg/ha	kcal/ha	% kg/ha	% kcal/ha
<i>Amaranthus spp</i> ^a	2970	6.5	19251.9	2%	3%	2970	8.4	24866.4	1%	1%
<i>Echinochloa spp</i> ^a	2720	73.9	200903.5	26%	33%	2720	398.2	1083018.2	43%	44%
<i>Helianthus spp</i> ^a	2500	2.7	6710.7	1%	1%	2500	0.1	252.4	< 1%	< 1%
<i>Leptochloa fascicularis</i> ^a	2730	22.1	60407.7	8%	10%	2730	3.1	8344.8	< 1%	< 1%
<i>Panicum dichotomiflorum</i> ^a	2450	9.4	23134.0	3%	4%	2450	4.9	12089.1	1%	< 1%
<i>Polygonum spp</i> ^a	1590	72.3	114910.8	26%	19%	1590	27.8	44134.8	3%	2%
<i>Andropogon spp</i> ^p	2730	-	-	-	-	2730	11.8	32203.6	1%	1%
<i>Eleocharis spp</i> ^p	960	3.0	2911.0	1%	< 1%	960	74.5	71545.9	8%	3%
<i>Marsilea vestita</i> ^p	2050	12.0	24675.5	4%	4%	2050	13.6	27792.1	1%	1%
<i>Phalaris spp</i> ^p	2730	-	-	-	-	2730	5.7	15508.0	1%	1%
<i>Polygonum coccineum</i> ^p	1590	5.5	8787.5	2%	1%	1590	0.5	803.3	< 1%	< 1%
<i>Potamogeton</i> ^p	1420	3.1	4365.1	1%	1%	1420	0.3	377.3	< 1%	< 1%
<i>Scirpus spp</i> ^p	930	30.2	28098.9	11%	5%	930	32.4	30095.2	3%	1%
<i>Sparganium spp</i> ^p	960	2.6	2451.2	1%	< 1%	960	0.8	754.0	< 1%	< 1%
<i>Glycine max</i> ^c	2650	20.9	55352.7	7%	9%	2650	191.3	507017.2	20%	21%
<i>Triticum aestivum</i> ^c	3900	-	-	-	-	3900	10.2	39749.4	1%	2%
<i>Zea Mays</i> ^c	3670	14.5	53344.8	5%	9%	3670	146.6	538003.6	16%	22%
Annual Communities	2275	188.0	427755.2	67%	70%	2650	442.8	1173426.5	47%	48%
Perennial Communities	1287	57.7	74193.9	20%	12%	1303	141.3	184122.2	15%	8%
Cropped Communities	3082	36.3	111892.5	13%	18%	3116	348.1	1084805.4	37%	44%
Total	2256	281.0	613841.6	100%	100%	2511	934.2	2447570.0	100%	100%

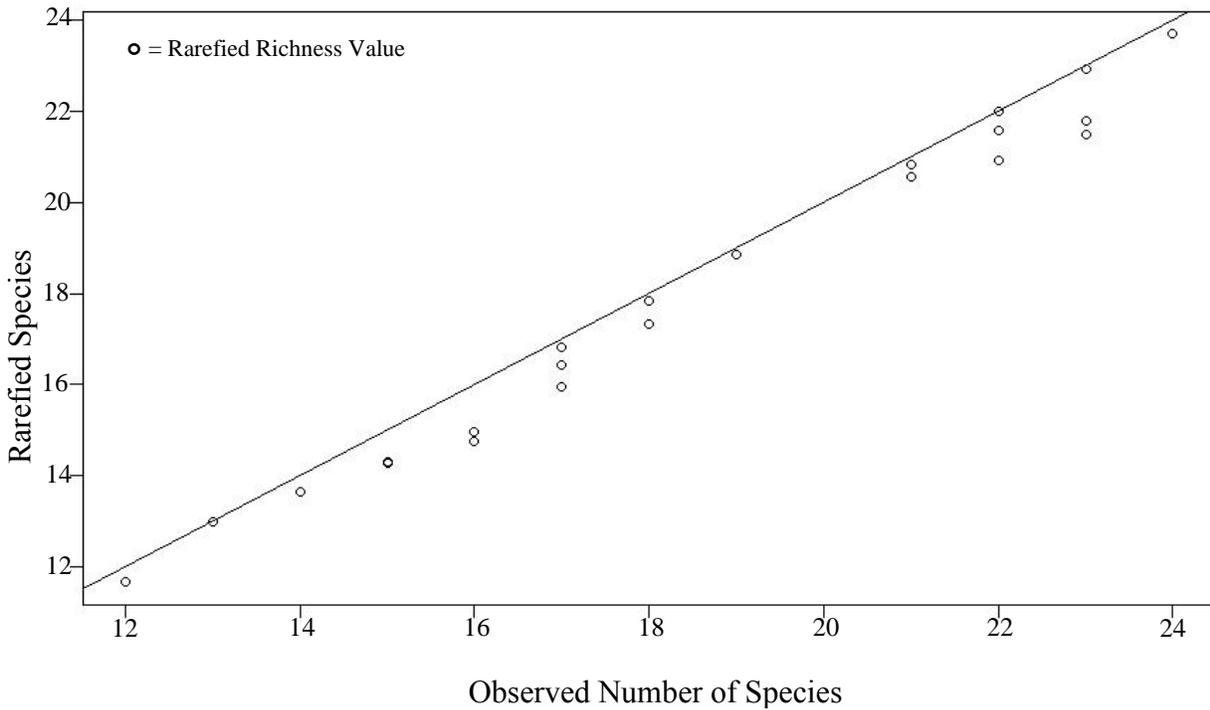
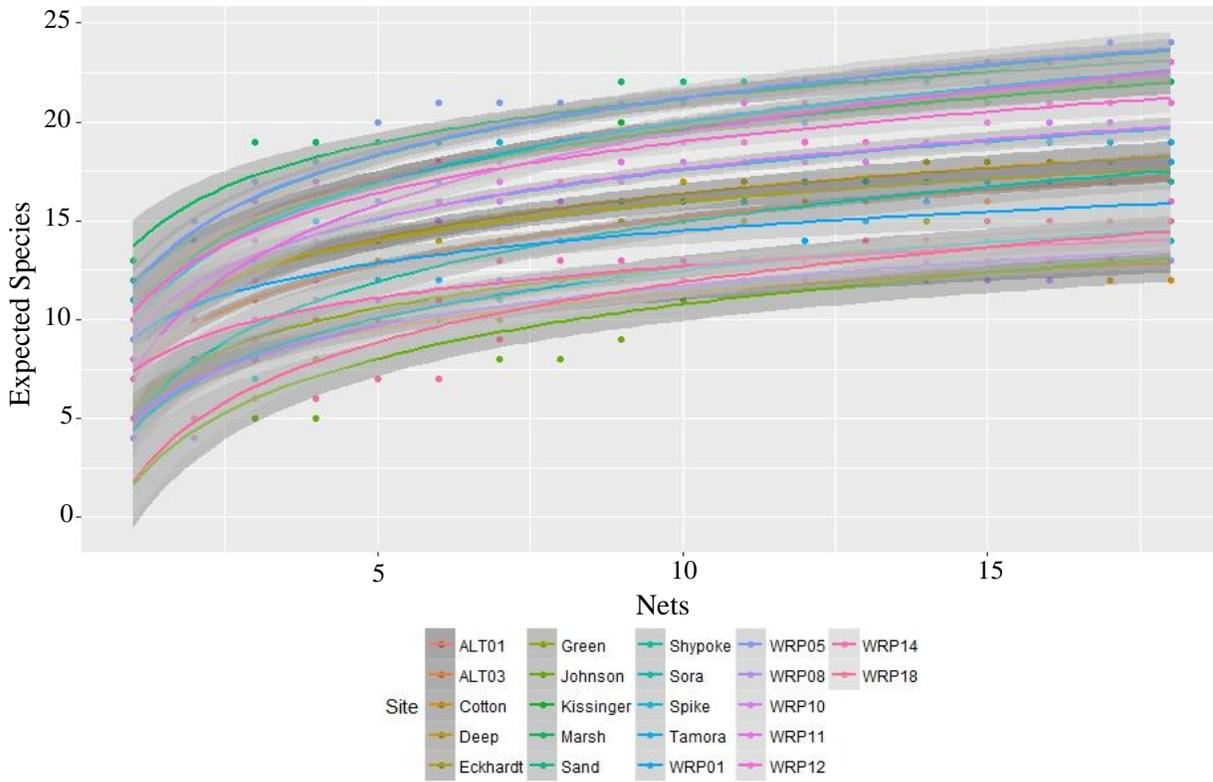
¹Detailed descriptions of estimates for true metabolizable energy (TME) can be reviewed in Appendix A.

^aAnnual plant species

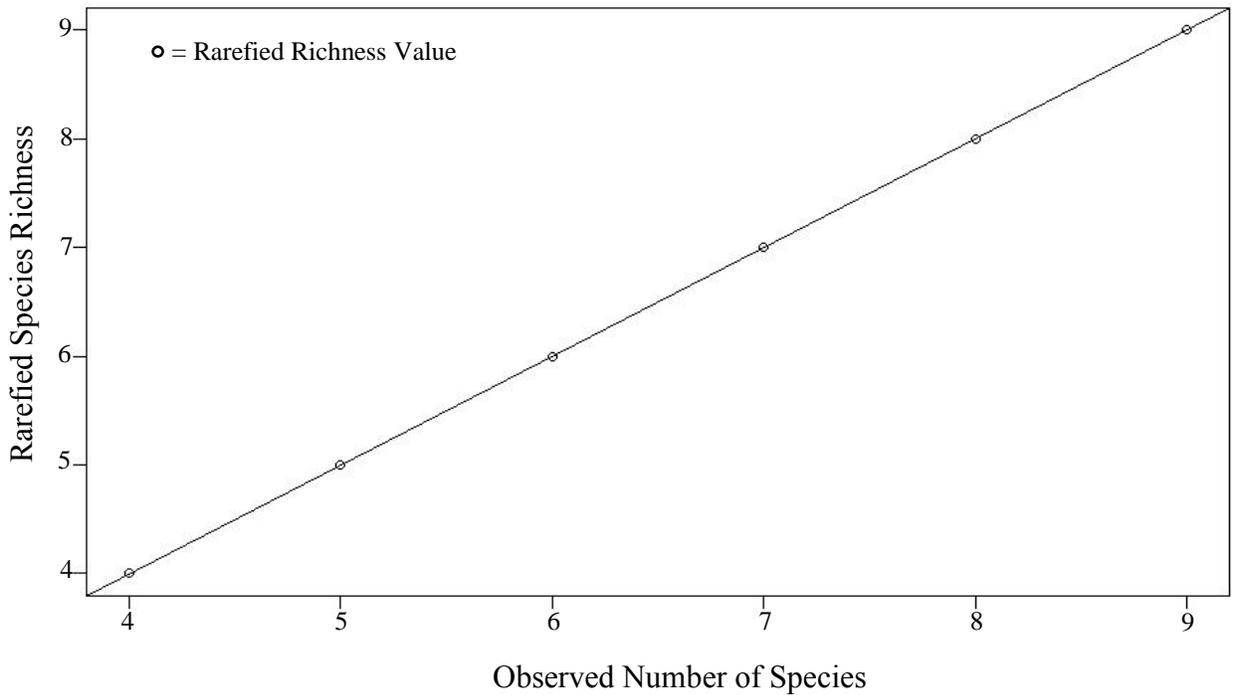
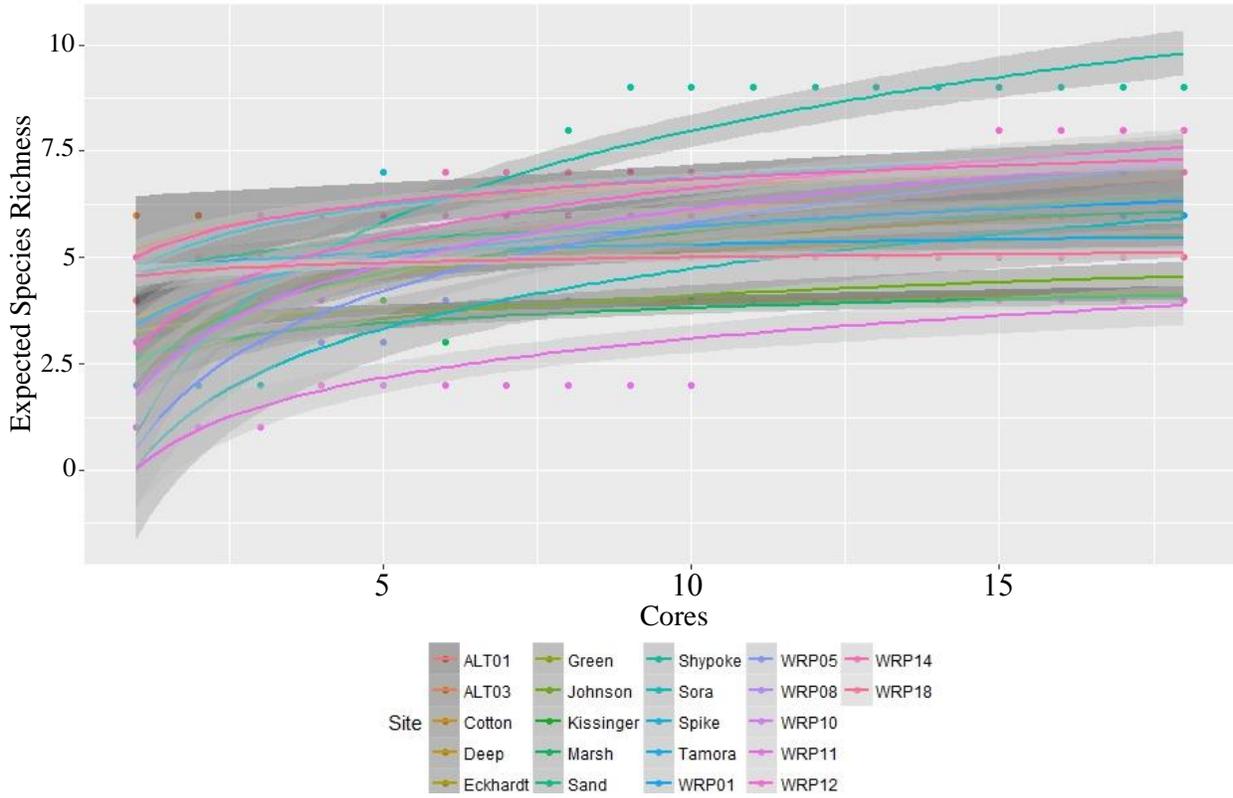
^pPerennial plant species

^cCropped plant species

Appendix M: (Top) Expected species richness curves (using rarefaction) for aquatic invertebrates as a function of number of nektonic samples collected at a site during spring 2015. Points represent actual cumulative richness. (Bottom) Nektonic rarefied richness plotted against the number of species observed.



Appendix N: (Top) Expected species richness curves (using rarefaction) for aquatic invertebrates as a function of number of benthic core samples collected at a site during spring 2015. Points represent actual cumulative richness. (Bottom) Benthic rarefied richness plotted against the number of species observed.



Appendix O: Chapter III metrics used to assess aquatic invertebrate communities.

Community	Metric	Mean	Std Error	Std Dev	Median	Min	Max	CI (95%)
Benthic	Rarefied Richness	6.27	0.34	1.61	6.00	4.00	9.00	0.71
	Diversity	0.81	0.05	0.25	0.82	0.25	1.18	0.11
	Density (g/m ²)	1.05	0.19	0.89	0.78	0.13	4.11	0.39
	Production (g/m ² /yr)	5.05	0.95	4.46	3.95	1.13	20.86	1.98
Nektonic	Rarefied Richness	17.98	0.77	3.63	17.58	11.65	23.69	1.61
	Diversity	1.41	0.07	0.31	1.39	0.68	1.87	0.14
	Density (g/m ²)	0.64	0.09	0.40	0.51	0.18	2.04	0.18
	Production (g/m ² /yr)	6.83	0.85	4.00	5.70	1.88	18.91	1.77

Appendix P: Neonicotinoid concentrations (ug/L) detected at wetlands sampled during March 2015 in Nebraska's Rainwater Basin.

Sample_ID	Collection Date	Clothianidin	Imidacloprid	Thiacloprid	Thiamethoxam
COTTON	3/11/2015	0.000	0.000	0.000	0.000
WRP 10	3/13/2015	0.000	0.000	0.000	0.000
WRP 8	3/11/2015	0.002	0.000	0.000	0.000
KISS	3/15/2015	0.000	0.000	0.000	0.000
SORA	3/14/2015	0.009	0.000	0.000	0.000
WRP1	3/11/2015	0.006	0.000	0.000	0.000
SHY	3/13/2015	0.003	0.000	0.000	0.000
WRP5	3/17/2015	0.006	0.000	0.000	0.000
WRP11	3/19/2015	0.002	0.000	0.000	0.000
WRP9	3/19/2015	0.003	0.000	0.000	0.000
WRP12	3/19/2015	0.005	0.000	0.000	0.000
JOHNSON	3/11/2015	0.000	0.000	0.000	0.000
WRP18	3/16/2015	0.009	0.000	0.000	0.000
TAM P1	3/13/2015	0.001	0.000	0.000	0.000
ALT 01	3/15/2015	0.003	0.000	0.000	0.000
WRP13	3/18/2015	0.009	0.000	0.000	0.000
SPIKE	3/18/2015	0.004	0.000	0.000	0.000
MARSH	3/18/2015	0.005	0.000	0.000	0.000
SAND	3/14/2015	0.000	0.000	0.000	0.000
DEEP	3/14/2015	0.003	0.007	0.000	0.000
GREEN	3/16/2015	0.009	0.000	0.000	0.000
ALT 03	3/17/2015	0.014	0.010	0.000	0.000
ALT 02	3/15/2015	0.004	0.000	0.000	0.000
WRP 14	3/15/2015	0.003	0.004	0.000	0.000
ECK	3/16/2015	0.001	0.006	0.000	0.000
WRP 15	3/17/2015	0.000	0.000	0.000	0.000
Method Detection Limits (ug/L) =>		0.002	0.002	0.002	0.002

Appendix Q: Neonicotinoid concentrations (ug/L) detected at wetlands sampled during April 2015 in Nebraska's Rainwater Basin

Sample_ID	Collection Date	Clothianidin	Imidacloprid	Thiacloprid	Thiamethoxam
MARSH	4/15/2015	0.009	0.000	0.000	0.000
WRP12	4/15/2015	0.006	0.000	0.000	0.000
SAND	4/12/2015	0.008	0.000	0.000	0.000
SPIKE	4/14/2015	0.007	0.000	0.000	0.000
WRP11	4/14/2015	0.005	0.000	0.000	0.000
WRP14	4/12/2015	0.011	0.000	0.000	0.000
GREEN	4/11/2015	0.000	0.000	0.000	0.000
SORA	4/12/2015	0.023	0.000	0.000	0.000
WRP5	4/12/2015	0.008	0.000	0.000	0.000
ALT3	4/12/2015	0.008	0.000	0.000	0.000
ECK	4/11/2015	0.000	0.000	0.000	0.000
WRP10	4/10/2015	0.000	0.000	0.000	0.000
ALT1	4/11/2015	0.000	0.000	0.000	0.000
KISS	4/11/2015	0.017	0.000	0.000	0.000
WRP1	4/10/2015	0.005	0.000	0.000	0.000
SHY	4/10/2015	0.004	0.000	0.000	0.000
JOHN	4/7/2015	0.000	0.000	0.000	0.000
TAM	4/10/2015	0.004	0.000	0.000	0.000
COT	4/7/2015	0.000	0.000	0.000	0.000
DEEP	4/7/2015	0.005	0.000	0.000	0.000
WRP18	4/11/2015	0.009	0.000	0.000	0.000
WRP8	4/10/2015	0.006	0.000	0.000	0.000
Method Detection Limits (ug/L) =>		0.002	0.002	0.002	0.002

