LINKING WATERFOWL DISTRIBUTION AND ABUNDANCE TO SPATIAL AND TEMPORAL DISTRIBUTION AND ABUNDANCE OF WETLAND HABITAT

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INTRODUCTION

Wetlands are dynamic ecosystems providing immense value to wildlife and humans. Wetlands store and purify water, recharge aquifers and reduce extreme flooding events (Acharya 2000). Wetlands also reduce soil erosion, decrease sediment loads, and sequester and transform nutrient runoff in agricultural areas and the surrounding landscape (Mitsch and James 2000, Woodward and Wui 2000). Wetlands provide essential resources for many biota and because of unique selective pressures, are home to numerous plant and animal species that rely on wetland ecosystems to persist (Brinson 1993, Gibbs 1993, Schweiger et al. 2002). For example, wetlands are important to many waterbird species for completing the annual cycle by providing foraging habitat in mid-latitude states to facilitate energy accumulation to over-winter or complete migration (Anderson and Smith 1991, Ma et al. 2009, Hagy and Kaminsky 2012). Specifically, dabbling ducks meet their autumn and winter food energy requirements through foraging in inundated wetlands and agricultural fields throughout their migratory range (Gruenhagen and Fredrickson 1990, Gibbs 2000, Checket et al. 2002, Brasher et al. 2006, Hagy and Kaminsky 2012).

Wetlands once covered an estimated 89 million hectares within the conterminous United States prior to European settlement in the early-1660s (Dahl 1990). By the mid-1980s more than half of U.S. wetlands had been degraded or destroyed, with >85% wetland loss in Missouri and five other states (Dahl and Allord 1996, Gibbs 2000, Dahl 2011). Agricultural practices, urbanization, dam construction and water diversions have been identified as the most destructive and common causes of wetland conversion (Tiner 1984). In northern Missouri, wetlands occurring within the Missouri River floodplain are largely hydrologically disconnected from the river due to impoundment, channelization, levee construction, and bank stabilization to improve navigation, reduce floods, and facilitate bottomland agriculture (Galat et al 1998, Jacobson et al. 2007). Remaining wetlands in the region have also been degraded or removed to support row crop agriculture (Dahl and Allord 1996, Nigh and Schroeder 2002, Nelson 2010, USGS Land Cover Trends 2012).

Considerable rates of wetland degradation and loss in conjunction with an increasing awareness of unique ecosystem functions provided by wetlands stimulated interest in restoring and protecting wetlands across North America, as well as species dependent upon wetland ecosystems (Williams et al. 1999, Mitsch and Gosselink 2000, Woodward and Wui 2001). The Clean Water Act of 1972 was a catalyst for government intervention of wetland degradation and loss by identifying and preventing point and non-point pollution sources and incentivizing the preservation of wetland integrity (Hough and Robertson 2008). The 1985 Farm Bill included the (Swampbuster) provision removing incentives for agricultural production on converted wetlands (Dahl and Allord 1996), while the Emergency Wetland Restoration Act of 1986 further reduced wetland losses using funds previously unavailable for wetland acquisition (Dahl and Allord 1996, Dahl 2011). The North American Waterfowl Management Plan (NAWMP) was implemented in 1986 with a goal of fostering international cooperation between the U.S., Canada and Mexico in protecting, restoring and enhancing waterfowl habitat in North America (NAWMP 2012). The North American Wetlands Conservation Act (NAWCA), a supplement to the NAWMP, has been essential to providing grants and matching funds aimed at restoring wetlands in Mexico, the U.S., and Canada since 1990. Since the implementation of NAWMP and NAWCA, more than 17 million wetland hectares have been restored in North America (USFWS 2013). In 1990 the U.S. Department of Agriculture implemented the Wetland Reserve Program (WRP) to help mitigate the loss and degradation of North American wetlands on private and tribal agricultural lands (King et al. 2006, NRCS 2013). The WRP provides landowners financial incentive to restore wetlands on private lands with marginal row crop production due to intermittent flooding and poor soil conditions (King et al. 2006, NRCS 2013). In Missouri, the first year of WRP implementation restored 691 hectares of wetlands on 19 conservation easements. As of 2016, over 59,000 hectares of wetland habitat are conserved on 1,068 WRP easements throughout Missouri (NRCS 2013; K. Dacey, NRCS, personal correspondence). Nationwide, the WRP program has restored approximately 1 million wetland hectares, and along with public lands, provides fall and winter habitat for millions of migratory waterfowl (Weibe et al. 1997, Gray and Teels 2006).

Although substantial resources have been devoted to wetland restoration, less effort has been directed toward documenting wetland distribution and the spatial and temporal occurrence of wetland inundation and identifying the potential factors influencing wetland inundation (Kuzilla et al. 1991, Houhoulis and Michener 2000, Kudray and Gale 2000, Klimas et al. 2009). For example, many National Wetland Inventory (NWI) state records, the most extensive spatial database for wetlands within the United States, predate the NAWMP, NAWCA, and WRP initiatives, indicating a potential deficiency in documenting current wetland distribution across the North American landscape (Kudray and Gale 2000, Houhoulis and Michener 2000). Compounding this issue, presence of surface water in many wetlands is ephemeral, meaning these sites can go extended periods without being inundated, creating the challenge of not only documenting spatial occurrence of wetlands but also characterizing temporal occurrence of surface water (Brinson 1996, Johnson et al. 2011, Dvorett et al. 2016).

The combined deficiencies in documenting wetland spatial and temporal occurrence limit the ability of waterfowl managers and researchers to effectively assess wetland conditions and habitat availability for migrating and wintering waterfowl. For example, Migratory Bird Joint Ventures focus conservation planning efforts on areas identified by the NAWMP based on available data of wetland spatial occurrence (Soulliere et al. 2007, Migratory Bird Joint Ventures 2016). Recognizing these deficiencies, researchers have worked to develop methods identifying wetland spatial locations, and factors influencing wetland inundation (Baker et al. 2006, Curie et al. 2007, Tsai et al. 2007, Maxa and Bolstad 2009, Johnson et al. 2011, Bartuszevige et al. 2012). Indeed, researchers have developed new methods of identifying spatial locations of wetlands worldwide using satellite imagery and Geographic Information Systems (GIS, Baker et al. 2006, Curie et al. 2007, Maxa and Bolstad 2009, Merot et al. 2013). Recent research has been directed towards characterizing spatial and temporal occurrence of wetlands and identifying factors influencing wetland inundation (Tsai et al. 2007, Johnson et al. 2011, Bartuszevige et al. 2012). For example, recent studies of playa wetlands indicated that wetland area, land cover and precipitation variables all influenced wetland inundation (Johnson et al. 2011, Bartuszevige et al. 2012). However, Missouri wetlands differ hydrologically from playa wetlands and factors influencing playa wetland inundation may not be significant determinants in Missouri. To my knowledge, little work has identified spatial and temporal occurrence of wetlands and the factors influencing wetland inundation in Missouri during the typical onset and peak of autumn dabbling duck migration.

Identifying the spatial and temporal occurrence of wetlands and the factors influencing inundation is necessary for effective natural resource management. By better understanding where wetlands occur and the factors that influence inundation, mangers can better meet habitat requirements and energetic demands of migrating waterfowl during autumn migration and winter. Thus, the objectives of this study were to 1) characterize autumn wetland inundation patterns for the GMRE, 2) evaluate the influence of land cover, soil, and precipitation variables on autumn wetland inundation and 3) assess the applicability of using NWI wetland basins as a measure of habitat availability in the Grand and Missouri River ecoregion, Missouri (GMRE). I assessed factors influencing autumn NWI wetland inundation in the GMRE using generalized linear mixed models and developed predictive models for NWI wetland inundation using top-ranked models and Receiver Operating Characteristics (ROC) curves. Because NWI and WRP data were the only readily available spatial data to estimate wetland habitat area in the GMRE, I also investigated the extent to which wetland area based on NWI and WRP data characterized observed wetland area within the GMRE using Landsat 5 (TM) satellite imagery.

STUDY AREA

The GMRE landscape is characterized by two distinct landforms; the Grand and Missouri River alluvial floodplain, and the Central Dissected Till Plains (Nigh and Schroeder 2002, USGS Land Cover Trends 2012, Missouri Ecological Classification System 2013). Alluvial floodplains associated with the Grand and Missouri Rivers have a high proportion of fine-textured, poorly drained soils that historically supported wetlands, wet prairies, and bottomland forest systems (Nigh and Schroeder 2002). The soils are deep alluvial sediments with sandy soil bordering the river and soils with higher clay content located in back swamp and slack waters further from the river channel. Historic Grand and Missouri River land cover was largely wet prairie and marsh with isolated stands of bottomland forest (Nigh and Schroeder 2002, Nelson 2005). The Central Dissected Till Plains are pre-Illinoisan glacial till with a thin layer of loess (Nigh and Schroeder 2002). The uplands are generally smooth with rolling hills and dissected tracts (Nigh and Schroeder 2002). Historical land cover was driven by patterns of ridges and valleys and was a complex mosaic of open prairie and timber (Nigh and Schroeder 2002).

In the early 1900's the Missouri and Grand Rivers were altered to stabilize and deepen the channel through installation of wing dikes and levees, as well as dredging the main channel (Nigh and Schroeder 2002). Since then, the Grand and Missouri River floodplains have been almost entirely converted to row crop agriculture with only small pockets of historical wetlands, wet prairie, and bottomland forest remaining (Nelson 2005). The Central Dissected Till Plains have been largely converted to row crop agriculture and cattle grazing. These alterations in land use are considered to be the primary source of woody invasive species in the region, due to poor farming and grazing practices (Nelson 2002). Remaining wetlands are often imbedded within an agricultural landscape and have experienced substantial degradation from siltation from eroding surrounding soils (Nelson 2002). Records indicate that the GMRE contains ~166,000 ha of NWI wetlands (NWI 2016), ~27,000 ha of WRP easements (personal correspondence, Kevin Dacey) and ~10,500 ha of pubic wetland areas managed by the Missouri Department of Conservation (MDC) and U.S. Fish and Wildlife Service (personal correspondence, public wetland area managers). Mean annual precipitation ranges from 89-99cm with May receiving the greatest average monthly precipitation (Nigh and Schroeder 2002). Migration and overwintering for most dabbling ducks in the region occurs between October and February (Schummer et al. 2010, personal correspondence, public wetland area managers), with October receiving the greatest mean monthly precipitation during the autumn and winter seasons (8-9cm; Missouri Climate Center 2016).

METHODS

Sampling Design and Data Collection

The GMRE landscape encompasses a diversity of floodplain and upland habitats with varying land cover, land use, hydrology, and geomorphology (Nigh and Schroeder 2002, Karstensen 2009, Nelson 2010, USGS Land Cover Trends 2012, EPA Ecoregions of Iowa and Missouri 2014). Therefore, stratification of the study area by floodplain and upland landscape was necessary because explanatory variables influencing wetland inundation likely differ between floodplain and upland wetlands (Quinn and Keough 2002). I identified three strata in the GMRE; two of the strata were the Grand and Missouri River floodplains (Hereafter Grand and Missouri) and the other stratum was the remaining area outside the Grand and Missouri Floodplain (hereafter Other; Figure 1.1).

Response Variable

To assess wetland distribution and inundation, I obtained spatial locations of NWI wetland polygons (n=8,474) from the USFWS wetlands mapper database (USFWS Wetlands Mapper 2016) in ArcGIS (GIS, Environmental Systems Research Institute ArcGIS 10.3, 2016). Polygons classified as deepwater habitats under the Cowardin classification (i.e., lakes, ponds, and rivers) were removed from consideration because their inundation status was expected to remain static over the study period (Cowardin 1979, Dvorett et al. 2011). I also excluded NWI wetland polygons <2.03 ha, generally offer low energetic value to migrating and wintering waterfowl (Johnson et al. 2011). Next, I randomly partitioned 80% of the NWI dataset to use as training data to develop predictive NWI wetland inundation models and 20% to evaluate model performance (Fielding and Bell 1995, Boyce 2002, Lozier 2009, Johnson et al. 2011).

I documented spatial occurrence of water within the GMRE in November which coincides with the typical onset of the waterfowl migration into Missouri (personal correspondence, public wetland area managers). I downloaded Landsat 5 Thematic Mapper (TM) scenes for the GMRE in November for 2004-2006 and 2008-2010. Images from 2007 were not included because all scenes for November 2007 were obscured by cloud cover. Further, no year provided complete spatial coverage of the GMRE because cloud cover obscured parts of the GMRE for every year of the study period (Figure 1.2). A total of 21,807 NWI and 2,350 WRP wetland inundation observations was identified over the six year study period. I mosaicked all Landsat 5 TM scenes for each year and stacked a 7, 5, 2 band combination to differentiate areas of inundation from agricultural fields, forested areas and urban centers (Lunetta and Balogh 1999, Pickins and King 2014, Dvorett et al. 2016). Band 7 (2.08-2.35 u m) separates water and land definitively and has both a strong water absorption region and strong soil and rock reflectance region (USFWS Wetland Mapper, 2016). Band 5 (1.55-1.75 u m) is sensitive to turgidity, or amount of water in plants, and is instrumental in separating inundated vegetation from surrounding cropland (USFWS Wetland Mapper, 2016). Band 2 (0.52-0.60 u m) is sensitive to water turbidity and sharply contrasts vegetation from bare soil (USFWS Wetland Mapper, 2016). I performed supervised classification in Earth Resources Data Analysis System (ERDAS) Imagine where I identified known land cover pixels and created specific land classes that could be converted to a raster dataset (ERDAS 2008, Pickens and King 2014). Because I was concerned with wetland inundation status, I developed only two classifications of land cover; inundated and non-inundated. I used MDC and USFWS wetlands known to be inundated as training sites for the water class and cropland, urban, and forested areas as training sites for the terrestrial class. The MDC and USFWS actively inundate wetlands on public lands in autumn, which provided a reliable source for consistently inundated training sites in November. Terrestrial training sites were easily identified on Landsat 5 TM imagery and I used a wide range of cropland, urban, and forested land cover for terrestrial classification (Appendix A). The resulting raster dataset was a binary classification of water and terrestrial land cover for the GMRE. The only misclassifications were cloud shadows and hillside shade, which were infrequent and easily identified by observer. I converted the binary raster dataset to a polygon dataset (hereafter Landsat wetlands) in ArcGIS and removed any polygons that were classified as water but determined to be cloud shadows. To remove misclassified hill shade, I excluded Landsat wetlands occurring on slopes greater than 3% using a 1990 60M Digital Elevation Model (DEM) collected from Missouri Spatial Data Information System (MSDIS) online. Finally, I performed a spatial join of the NWI and WRP polygon datasets and Landsat wetlands dataset in ArcGIS to determine inundation status of each NWI wetland for each year of the study period. I identified wetland inundation as the dependent variable with a binary distribution of 0 (not inundated) and 1 (inundated), regardless of area inundated.

Predictor Variables

I identified five predictor variables that potentially influenced inundation of NWI wetlands in autumn (Table 1.1). I obtained hydric soil data for the ecoregion from the Soil Survey Geographic Database (NRCS 2016). Hydric soils are characterized by evidence of anaerobic conditions developed due to sustained saturation, flooding or ponding, and are directly related to frequency and duration of wetland inundation (Wilen and Bates 1995, Richardson and Vepraskas 2001, Schaetzl and Anderson 2005). I determined the presence or absence of hydric soils within NWI wetland basins using the spatial join tool in ArcGIS. Wetland basins containing both hydric and non-hydric soils were classified as hydric. To evaluate the influence of precipitation on autumn wetland inundation, I collected annual (Nov. 1-Oct. 31) and October (Oct. 1-31) precipitation totals (cm) from United States Historical Climatology Network (USHCN) stations (n=17) located within the GMRE for years 2004-2006 and 2008-2010. Using precipitation data from USHCN weather station points, I created a continuous layer of interpolated precipitation values using the kriging tool in ArcGIS (Johnson et al. 2011). I then converted the kriging raster data to point data in ArcGIS and performed a spatial join function to associate a unique precipitation value to each NWI wetland basin. I evaluated the influence of percent land cover type in a HUC 12 watershed (i.e. row crop, hay and pasture, and deciduous forest; Figure 1.3) on autumn wetland inundation probability. I collected National Land Cover Database 2011 (NLCD) raster data and performed a zonal tabulation to calculate percent land cover type for each HUC 12 watershed within the GMRE. I used the zonal tabulation tool in ArcGIS to associate each NWI wetland with a specific watershed and calculated the percent land cover within the watershed. Finally, to evaluate the influence of NWI wetland area on autumn inundation, I used the ArcGIS calculate geometry function to calculate area of each NWI wetland polygon.

Statistical Analysis

I characterized autumn wetland inundation patterns for the GMRE by calculating proportion of sampled wetlands that were at least partially inundated (inundation frequency) for NWI and WRP wetlands among strata and year using PROC SURVEYMEANS in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA). I tested for differences in inundation frequency among strata using a one-way ANOVA in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA) to evaluate the null hypothesis that inundation frequencies did not differ among strata.

I used a Generalized Linear Mixed Model with a binary response distribution and logit link function to assess the influence of hydric soils, wetland area, annual precipitation, October precipitation, and watershed land cover on NWI and Landsat autumn wetland inundation probability using PROC MIXED in SAS 9.3 (Bolker et al. 2009, Johnson et al. 2011, Bartuszevige 2012). I included year as a random effect in all models to assess inter-annual variation in NWI and Landsat wetland inundation variation (Johnson et al. 2011, Bartuszevige et al. 2012). To account for multiple observations of the same wetlands over the 6 year study period, I included individual wetland site as a repeated measures variable (Quinn and Keough 2002). Landsat satellite images were not available for the entire study area for all years of the study period so I did not include observations of an individual wetland for years which data were not available. A primary assumption of regression analysis is that the predictor variables do not exhibit strong linear relationships known as multicollinearity (Wang 1996). Predictor variables exhibiting multicollinearity in regression modeling can lead to overestimates of variable parameters and erroneous selection of predictor variables (Wang 1996, Kroll and Song 2003). Thus, I used a Pearson correlation test to identify predictor variables that were highly correlated or exhibited multicollinearity (Johnson et al. 2011). Proportion of watershed in hay/pasture and deciduous forest land cover were highly negatively correlated with cropland land cover ($r=-0.90$, -0.95 respectively; Table 1.2), so I removed watershed land cover variables proportion deciduous and hay/pasture from the pool of potential predictor variables. Thirty one potential models were evaluated for each stratum using Akaike Information Criterion (AIC_c) for small sample sizes (Akaike 1998, Hurvich and Tsai 1989, Johnson et al. 2011, Bartuszevige et al. 2012) and I calculated AIC_c weights (w_i) to quantify strength of evidence for competing models (Burnham and Anderson 2002, Bartuszevige et al. 2012).

I applied the top ranked model for each strata to the remaining 20% of the original data to evaluate model performance (Fielding and Bell 1995, Chatfield 1995, Boyce 2002, Lozier 2009, Johnson et al. 2011). I developed a predictive range of NWI and Landsat inundation probabilities by calculating log odds ratios and developing a predictive threshold for top ranked models with a Receiver Operator Characteristic (ROC) curve (Fielding and Bell 1997). The ROC curve provides a graphical representation of continuous predictive thresholds of a binary response by plotting the rate of true positive (correct predictions) and false positive (incorrect predictions) wetland inundation predictions (Fielding and Bell 1995, Pearce and Ferrier 2000). An optimal predictive threshold can then be selected by identifying the predictive threshold with the lowest false positive rate (Pearce and Ferrier 2000). Once an optimal threshold was identified for each model using the ROC curve, I evaluated model performance by assessing the ratio of true positive and false positive predictions.

To investigate the extent that NWI and WRP wetlands characterized spatial occurrence of inundated wetlands in the GMRE, I compared the total area of NWI and WRP polygons to total area of Landsat wetlands (total area of observed wetland inundation on the GMRE landscape) for each year of the study period. I calculated the total area of NWI and Landsat wetlands in ArcGIS and compared those estimates to Landsat wetland area estimates using PROC SURVEYMEANS in SAS 9.3.

RESULTS

Patterns of Inundation

Mean NWI wetland inundation frequency differed among strata ($F_{2, 21806}$= 310.8, p=0.001). National Wetland Inventory wetlands had the greatest frequency of inundation in the Missouri stratum ($\bar{x}$=0.27) and the lowest inundation frequency in the Other stratum ($\bar{x}$=0.12). National Wetland Inventory wetlands had a lower mean inundation frequency than WRP wetlands among all strata and years ($\bar{x}$= 0.21 and 0.61 respectively; Table 1.3). Mean WRP wetland inundation frequency differed among strata ($F_{2, 2347}$= 12.8, p=0.0001), with greatest inundation frequency in the Grand stratum ($\bar{x}$=0.70) and the lowest inundation frequency in the Missouri stratum ($\bar{x}$=0.54).

NWI Inundation Modeling

Predictor variables in the top ranked models for NWI wetland inundation probability differed among strata (Table 1.4). The top ranked NWI model for the Grand stratum included annual precipitation, October precipitation, wetland area, and proportion of watershed in cultivation (w_i=0.93). The probability of NWI wetland inundation increased with increasing annual precipitation [$\beta_{(Ann)}$=0.019; SE=0.004], October precipitation [$\beta_{(Oct)}$=0.053; SE=0.013], wetland area [$\beta_{(ha)}$=0.0005; SE=0.00006], and proportion of watershed in cultivation [$\beta_{(Cult)}$=0.65; SE=0.05]. The top ranked model predicting NWI inundation probability for the Missouri stratum included wetland area, proportion of watershed in cultivation, and presence of hydric soils, (w_i=0.81). The probability of NWI wetland inundation increased with increasing wetland area [$\beta_{(ha)}$=0.0005; SE=0.00006], proportion of watershed in cultivation [$\beta_{(Cult)}$=0.73; SE=0.05], and presence of hydric soils [$\beta_{(Hyd)}$=0.012; SE=0.018]. The top ranked NWI model for the Other stratum included annual precipitation, October precipitation, wetland area, and hydric soils (w_i=0.92). The probability of NWI wetland inundation increased with an increase in annual precipitation [$\beta_{(Ann)}$=0.006; SE=0.0008], October precipitation [$\beta_{(Oct)}$=0.015; SE=0.003], wetland area [$\beta_{(ha)}$=0.002; SE=0.00008], and decreased with the presence of hydric soils [$\beta_{(Cult)}$=-0.04; SE=0.006].

Mean and range of NWI inundation probability values, and optimal predictive threshold and misclassification frequency of predictions differed among strata (Table 1.5). The Other stratum had the greatest range of NWI wetland inundation probability values ($\bar{x}$= 0.47, range= 0.49-0.77). Wetlands within the Missouri stratum had the greatest mean probability of inundation ($\bar{x}$= 0.57, range= 0.47-0.65), whereas wetlands within the Grand stratum had the lowest probability of inundation ($\bar{x}$ = 0.47, range= 0.40-0.52). Predicted inundation values for NWI wetlands in each stratum were assessed using a ROC to identify a predictive threshold in which an optimal "cutoff" would render the most correct predictions. A predictive threshold of 0.49 using ROC resulted in a misclassification frequency of 0.28 for the Grand stratum and predictive threshold of 0.61 had a misclassification frequency of 0.35 for the Missouri stratum (Figure 1.4). Finally, ROC indicated a predictive threshold of 0.53 resulted in a misclassification frequency of 0.28 for the Other stratum (Figure 1.4).

Landsat 5 TM water polygon Inundation Modeling

Predictor variables in the top ranked models for Landsat wetland inundation probability differed among strata (Table 1.6). The top ranked model for Landsat wetlands for the Grand stratum included proportion of watershed in cultivation (w_i=0.49). The probability of Landsat wetland inundation in the Grand stratum increased with an increase in proportion of watershed in cultivation [β(Cult)=0.1164; SE=0.33]. The top ranked Landsat wetland model for the Missouri stratum included presence of hydric soils, (w_i=0.34). The probability of Landsat wetland inundation in the Missouri stratum decreased with presence of hydric soils [β(Hyd)=-0.0909; SE=0.0545]. The top ranked Landsat model for the Other stratum included annual precipitation and hydric soils (w_i =0.24). The probability of Landsat wetland inundation in the Other stratum decreased with an increase in annual precipitation [β(Ann)=-0.0651; SE=0.03], and with the presence of hydric soils [β(Cult)=-0.0263; SE=0.001].

Mean and range of Landsat inundation probability values, and optimal predictive threshold and misclassification frequency of predictions differed among strata (Table 1.7). The Other stratum had the greatest range of Landsat wetland inundation probability values (0.54-0.75; Table 1.7). Wetlands within the Grand and Other strata had similar mean inundation probabilities ($\bar{x}$= 0.64 and 0.63, respectively; Table 1.7). A predictive threshold of 0.634 using ROC resulted in a misclassification frequency of 0.49 for the Grand stratum (Table 1.7). The Missouri stratum contained only two predictive values (0.64 and 0.66) as the predictor variable was binary (either 0 or 1) and could not produce a range of predictive values to be analyzed using ROC (Table 1.7). Misclassification frequency for the Missouri stratum was 0.44 (Table 1.7). Finally, ROC indicated a predictive threshold of 0.58 resulted in a misclassification frequency of 0.42 for the Other stratum (Table 1.7).

I was unable to estimate total area of inundation for the GMRE based on Landsat wetlands for a given year because spatial coverage of Landsat 5 TM scenes varied by year and no year had complete spatial coverage (Figure 1.2). Maximum possible area of autumn wetland inundation based on Landsat polygons pooled for the entire GMRE was 4,794 ha. The total area of observed wetland inundation using Landsat polygons was less than area of NWI and WRP (6% and 18% respectively) in the GMRE (Table 1.8).

DISCUSSION

Few studies have sought to assess representation of inundated wetland conditions in autumn using NWI spatial datasets (Stolt and Baker 1995, Kudray and Gale 2000), and none to my knowledge have assessed NWI accuracy in Missouri. Further, few studies have attempted to quantify NWI or WRP availability based on inundation and the factors influencing availability during the autumn waterfowl migration in Northern Missouri. My results demonstrates that Missouri NWI and WRP data may not represent actual inundated wetland conditions during the fall waterfowl migration and would likely represent an overestimate of waterfowl habitat availability in the GMRE.

I found that NWI wetlands had a low inundation frequency in the GMRE in autumn over the six year study period. For example, overall mean probability of NWI inundation was only 21%, which is similar to a study of playa wetlands in the Texas High Plains in January (Johnson et al. 2011). It is possible that many NWI wetlands that were not inundated during the study period were inundated at other times of the year or during the 16 day period between Landsat satellite pass over. It is also likely that many NWI wetlands are hydrologically altered from ditching and drain tiles and are only briefly inundated or no longer become inundated at all. My results indicate, similar to other studies (Stolt and Baker 1995), that many NWI wetland records in the GMRE are possibly outdated or do not provide an accurate spatial representation of inundated wetlands during the autumn waterfowl migration. WRP wetlands had a greater inundation frequency than NWI wetlands and were likely more consistently inundated when migrating waterfowl arrived in autumn. Further, many landowners manipulate WRP wetland hydroperiod with water control structures and pumping river or ground water to provide migratory bird habitat and waterfowl hunting opportunities (Kaminski et al. 2006, King et al. 2006). Many WRP wetlands in the GMRE are intentionally managed for migratory waterfowl hunting and were likely inundated during the study period regardless of environmental conditions. Because of this, it is plausible that WRP wetlands provide reliably inundated wetland habitat for migrating dabbling ducks in the GMRE during autumn.

Factors influencing NWI wetland inundation differed among strata. Indeed wetland area was the only predictor variable present in top models among all strata for NWI wetlands and was positively associated with inundation probability. Although increased water surface area facilitates evaporation (Tsai et al. 2007), wetland area can increase the amount of time a wetland basin holds water and likely influenced the positive association with wetland inundation probability (Hayashi and Van der Kamp 2000, Brooks and Hayashi 2001). Annual and October precipitation were predictor variables present in top models for two of the stratum and were positively associated with NWI wetland inundation. These results are not surprising as precipitation can influence overbank river flooding and surface inflow and is a primary means for filling wetland basins (Wigham 1999, Hunter et al. 2007, Dadson et al. 2010). Also, hydric soil was a predictor variable present in top models for two of the three stratum and was positively associated with NWI wetland inundation. Hydric soils form under conditions of prolonged soil saturation and are used as indicators of wetland conditions (Cowardin et al. 1979, Mitsch and Gosselink 2007). NWI models performed well when predicting inundation for all stratum using ROC. Two of the strata had a misclassification rate of 28% and one stratum had a misclassification rate of 35% indicating most inundation predictions would have been correct. However, it should be noted that NWI wetlands in the GRME had a low inundation frequency over the study period and predicting that most wetlands would not be inundated in autumn would inherently yield accurate results.

Factors influencing Landsat wetland inundation differed among strata and contained different and fewer variables than models predicting NWI wetland inundation. Top Landsat inundation models for two strata indicated the probability of wetland inundation decreased with the presence of hydric soils. Because >50% of wetlands in the USA have been drained for agriculture (Dahl 1990), it is possible that early land conversion practices focused on ditching and drain tiling areas that flooded more frequently in the GMRE. These results suggest that areas in the GMRE with historic hydric soils are likely now effectively drained to prevent the presence of standing water and inundated wetland conditions. Further, croplands are often ditched and tiled to facilitate removal of floodwaters. It is possible that cropland precipitation runoff accumulated in the few remaining areas not in agricultural use. Indeed, it is likely that areas not in agricultural use within the Grand River floodplain act as reservoirs for cropland water runoff.

Research aimed at assessing NWI wetland inundation and its ability to represent accurate wetland conditions is somewhat conflicting (Stolt and Baker, Kudray and Gale 2000, Bartuszevige et al. 2012). My results indicate that NWI and WRP data for the GMRE likely do not provide a comprehensive estimate of autumn wetland habitat availability for dabbling ducks. Mean inundation frequency among years for NWI and WRP was 15% and 61% respectively. Combined Landsat wetlands data from November 2004-2006 and 2008-2010 suggest that maximum possible area of inundation for the GMRE was only 6% of NWI and 18% of WRP area during the study period. Further, actual area of inundation based on Landsat 5 TM satellite images in most years was less than the maximum possible area, although I was unable to verify this trend for the entire GMRE because no year of the study period had complete Landsat satellite coverage (Figure 1.2). Consequently, wetland availability in autumn for migrating waterfowl is perhaps far less than assumptions based on NWI and WRP wetland coverage would suggest. When managing a landscape for waterfowl conservation, the implications of overestimating autumn wetland habitat availability can be problematic. For example, autumn can be energetically taxing for migrating waterfowl and is considered to be a potentially limiting period of the annual cycle (Fretwell 1972, Dubouy 1988, Heitmeyer 1988). Wetland managers and researches may need to adjust management strategies to compensate for the limited available wetland habitat in autumn on the Missouri landscape.

Recent land use and environmental changes such as conversion of agricultural land to WRP, and more frequent and intense flood events have likely impacted spatial and temporal wetland occurrence in the GMRE. Because NWI data likely do not represent inundated wetland conditions in the GMRE in autumn, it would benefit researchers to amend NWI records for the region to better assess waterfowl habitat conditions. It is possible that many of the NWI wetlands that were not inundated during the study period were inundated at other times of the annual cycle and provided migratory waterfowl habitat. My study demonstrates that Landsat 5 TM or similar satellite imagery can be instrumental in identifying occurrence of inundated wetlands in the GMRE and used to further investigate factors influencing wetland inundation. Further, these methods could be used to document spatial occurrence of wetlands during other times of the year to assess the efficacy of using NWI and WRP datasets outside the autumn dabbling duck migration. Given the uncertainty of NWI accuracy and using WRP data to quantify area of migratory waterfowl habitat, I recommend further investigation into the factors influencing the distribution of wetland habitat availability in autumn for the GMRE.

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Table 2.1. Descriptions of variables used to model wetland inundation probability in the Grand and Missouri River ecoregion during autumn 2004-2006, and 2008-2010

Variable Identifier	Variable Description	Mean or levels (Range)
Response Variable		
Inundated	Inundation status of wetland basin	2 levels (0,1)
Predictor Variable		
Hyd	Hydric soil present within wetland	2 levels (0,1)
Area	Wetland area (ha)	14.4 ha (2.01 - 2314)
Annual	Annual precipitation value (cm) for wetland	44.7 cm (25.5 – 65.1)
October	October precipitation value (cm) for wetland	3.9 cm (0.37 – 12.2)
Cult	Percent cropland of the associated watershed	0.46 (0.003 – 0.99)
Random Effect Variable		6 levels (1 - 21,807)
Year	Year of inundation observation	6 levels (2004-2010)
Repeated Measures Variable		
Wetland ID	Wetland identification number	N/A (1 – 5,996)

Table 1.2. Pearson correlation test identifying highly correlated predictor variables. Data were pooled among strata to assess overall correlation of variables for the entire Grand and Missouri River Ecosystem (GMRE)[a]

	Hyd	Annual	October	Ha	Decid[b]	Hay[c]	Cult
Hyd	1.00000	0.06710	0.04124	-0.02934	-0.20774	-0.24632	0.24807
Annual	0.06710	1.00000	0.18456	0.04872	-0.11579	-0.16466	0.15556
October	0.04124	0.18456	1.00000	-0.01099	-0.04620	-0.05283	0.05394
Ha	-0.02934	0.04872	-0.01099	1.00000	-0.09658	-0.04853	0.07365
Decid[b]	-0.20774	-0.11579	-0.04620	-0.09658	1.00000	0.71627	-0.89759
Hay[c]	-0.24632	-0.16466	-0.05283	-0.04853	0.71627	1.00000	-0.95054
Cult	0.24807	0.15556	0.05394	0.07365	-0.89759	-0.95054	1.00000

[a] Refer to table 1.1 for variable identified codes
[b] This variable describes the proportion of deciduous forest within a given watershed
[c] This variable describes the proportion of hay and pasture within a given watershed

Table 1.3. Annual percent National Wetland Inventory (NWI) and Wetland Reserve Program (WRP) wetlands inundated in the Grand and Missouri River ecoregion, Missouri in November 2004-2006 and 2008-2010. Inundation status was determined using Landsat 5 Thematic Mapper (TM) imagery.

Data source	2004	2005	2006	2008	2009	2010	Mean
NWI (overall)	36%	15%	16%	11%	28%	17%	21%
Grand	51%	4%	12%	9%	36%	18%	22%
Missouri	32%	32%	24%	20%	33%	23%	27%
Other	24%	9%	11%	3%	15%	7%	12%
WRP (overall)	71%	49%	41%	78%	66%	61%	61%
Grand	88%	N/A[1]	51%	46%	89%	75%	68%
Missouri	54%	N/A	48%	41%	66%	60%	54%
Other	71%	N/A	49%	36%	80%	65%	60%

[1]Analysis of WRP in 2005 was excluded because Landsat 5 TM coverage was limited due to cloud cover (figure 1.2) and only 12 of 543 WRP wetlands were visible during that year.

Table 1.4. Top three ranked models explaining National Wetland Inventory (NWI) wetland inundation for three strata in the Grand and Missouri River Ecoregion. Models were ranked using Akaike Information Criterion for small sample size (AIC_c) for the effects of variables on inundation of NWI wetlands (model terms: Annual = annual precipitation, October = October precipitation, Ha = hectares, Cult = proportion land cover cultivated in associated watershed, Hyd = hydric soils).

Stratum	Model	K	log(*L*)	AIC_c	ΔAIC_c	w_i
Grand	Annual, October, Ha, Cult	5	-1257.5	1261.5	0.0	0.93
	Annual, October, Ha, Cult, Hyd	6	-1263.5	1267.5	6.0	0.04
	Annual, Ha, Cult	4	-1266.5	1270.5	9.0	0.01
Missouri	Ha, Cult, Hyd	4	-4260.9	4264.9	0.0	0.81
	October, Ha, Cult, Hyd	5	-4264.8	4268.8	3.9	0.12
	Ha, Cult	3	-4266.3	4270.3	5.4	0.50
Other	Annual, October, Ha, Hyd	5	-7255.8	7259.8	0.0	0.92
	Annual, October, Ha, Cult, Hyd	6	-7260.9	7264.9	5.1	0.07
	Annual, Ha, Hyd	3	-7265.8	7269.8	10.0	0.01

Table 1.5. Mean and range of inundation probability, optimal predictive threshold and misclassification frequency for National Wetland Inventory wetland inundation predictions in November for the Grand and Missouri River Ecoregion, Missouri 2004-2006 and 2008-2010. Inundation predictions were developed using generalized linear mixed models and ranked using AIC_c. A receiver operator characteristics (ROC) curve was used to identify an optimal cutoff value for wetland inundation predictions.

Stratum	Mean	Range	Optimal Cutoff Value	Misclassification frequency
Grand	0.47	0.40-0.52	0.50	0.28
Missouri	0.57	0.47-0.65	0.61	0.39
Other	0.53	0.49-0.77	0.53	0.28

Table 1.6. Top three ranked Landsat wetland inundation models for each strata. Models were ranked using Akaike Information Criterion for small sample size (AIC_c) for the effects of variables on inundation of NWI wetlands (model terms: Cult = proportion land cover cultivated in associated watershed, Hyd = hydric soils).

Stratum	Model	K	AIC_c	ΔAIC_c	w_i
Grand	Cult	2	202.8	0	0.44
	Cult + Hyd	3	204.9	2.1	0.15
	Hyd	2	205.2	2.4	0.13
Missouri	Cult	2	339.9	0.0	0.34
	Hyd	2	340.6	0.7	0.24
	Cult + Hyd	3	342.1	0.3	0.11
Other	Cult	2	1,311.5	0.0	0.94
	Hyd + Cult	3	1,312.0	0.5	0.05
	Hyd	2	1,312.1	0.6	0.01

Table 1.7. Mean and range of inundation probability, optimal predictive threshold and misclassification frequency for wetland inundation predictions for wetland basins developed using Landsat 5 Thematic Mapper scenes in November for the Grand and Missouri River Ecoregion, Missouri 2004-2006 and 2008-2010. Inundation predictions were developed using generalized linear mixed models and ranked using AIC_c. A receiver operator characteristics (ROC) curve was used to identify an optimal cutoff value for wetland inundation predictions.

Stratum	Mean	Range	Optimal Cutoff Value	Misclassification frequency
Grand	0.64	0.633-0.642	0.638	0.49
Missouri	0.69	0.641-0.662	N/A[1]	0.44
Other	0.63	0.536-0.750	0.58	0.42

[1]The top ranked model for the Missouri stratum contained only one variable and had a binary distribution.

Table 1.8. Percent area of National Wetland Inventory (NWI) wetland, Wetland Reserve Program (WRP) wetland and NWI, WRP area combined in the Grand and Missouri River Ecoregion, Missouri identified as inundated based on Landsat 5 TM aerial imagery. Landsat 5 (TM) aerial imagery was collected for years 2004-2010, although years 2005 and 2007 were not included because spatial coverage of the study area was obscured by cloud cover.

	2004	2006	2008	2009	2010	$\bar{x}$
NWI	7%	5%	7%	6%	6%	6%
WRP	16%	18%	18%	20%	20%	18%
NWI+WRP	5%	4%	5%	5%	5%	5%

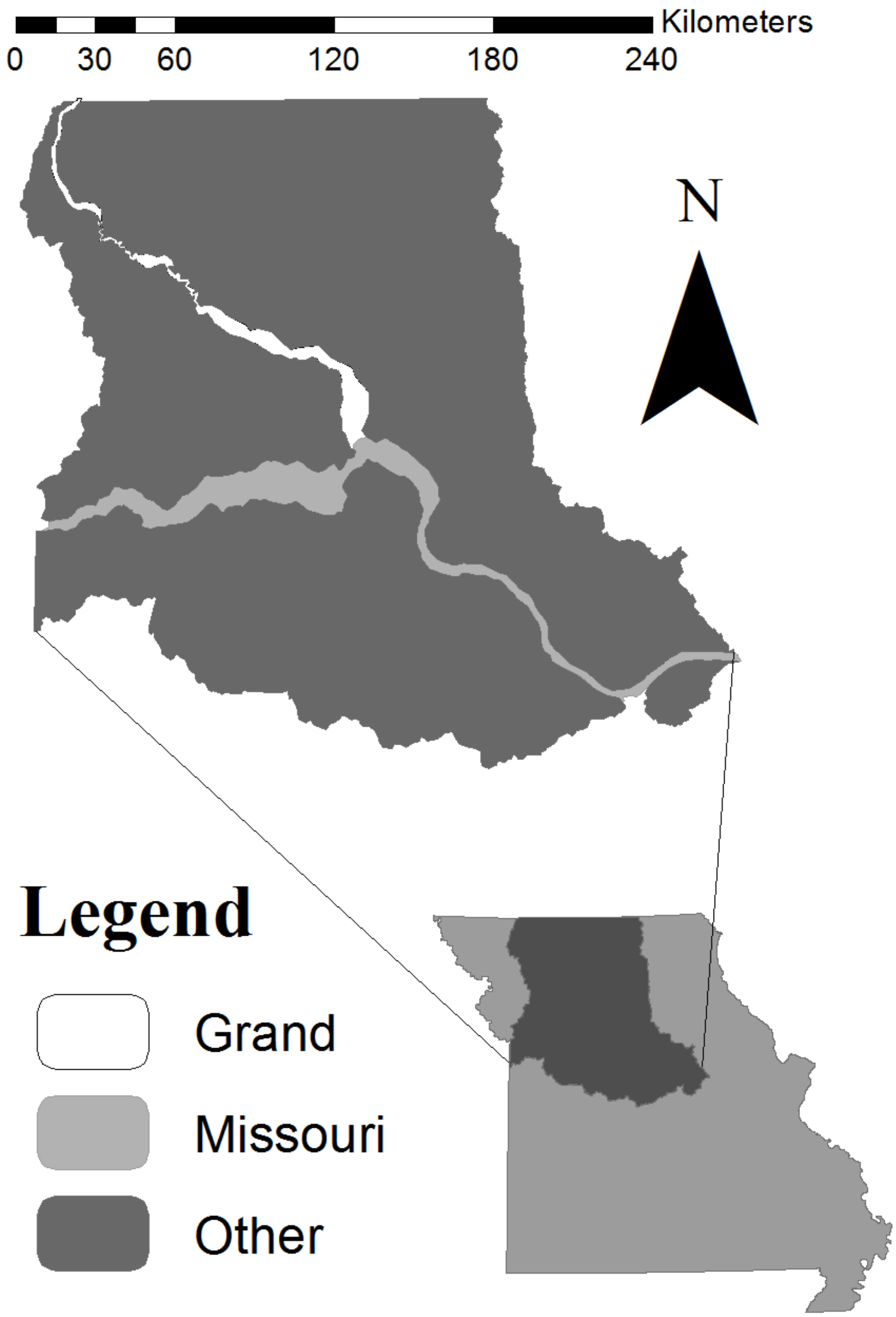

Figure 1.1. Stratification of the Grand and Missouri River Ecoregion (GMRE) was necessary to ensure accurate modeling of potential factors influencing inundation because the landscape encompassed a diversity of land cover, hydrology, and geomorphology. The GRME was partitioned into three strata based on the geographic extent of two major floodplains (Grand and Missouri) and all other land types (Other).

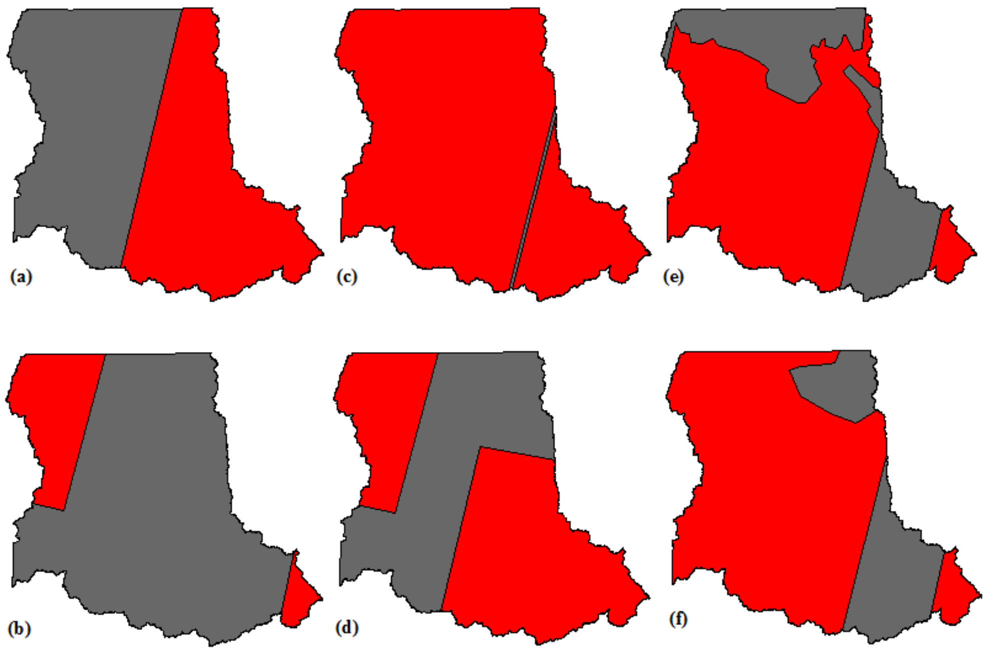

Figure 1.2. Area in red depicts spatial coverage of Landsat 5 Thematic Mapper scenes for years (a) 2004, (b) 2005, (c) 2006, (d) 2008, (e) 2009, (f) 2009. Spatial coverage varied by year and no year had complete spatial coverage because of partial cloud cover during satellite pass over.

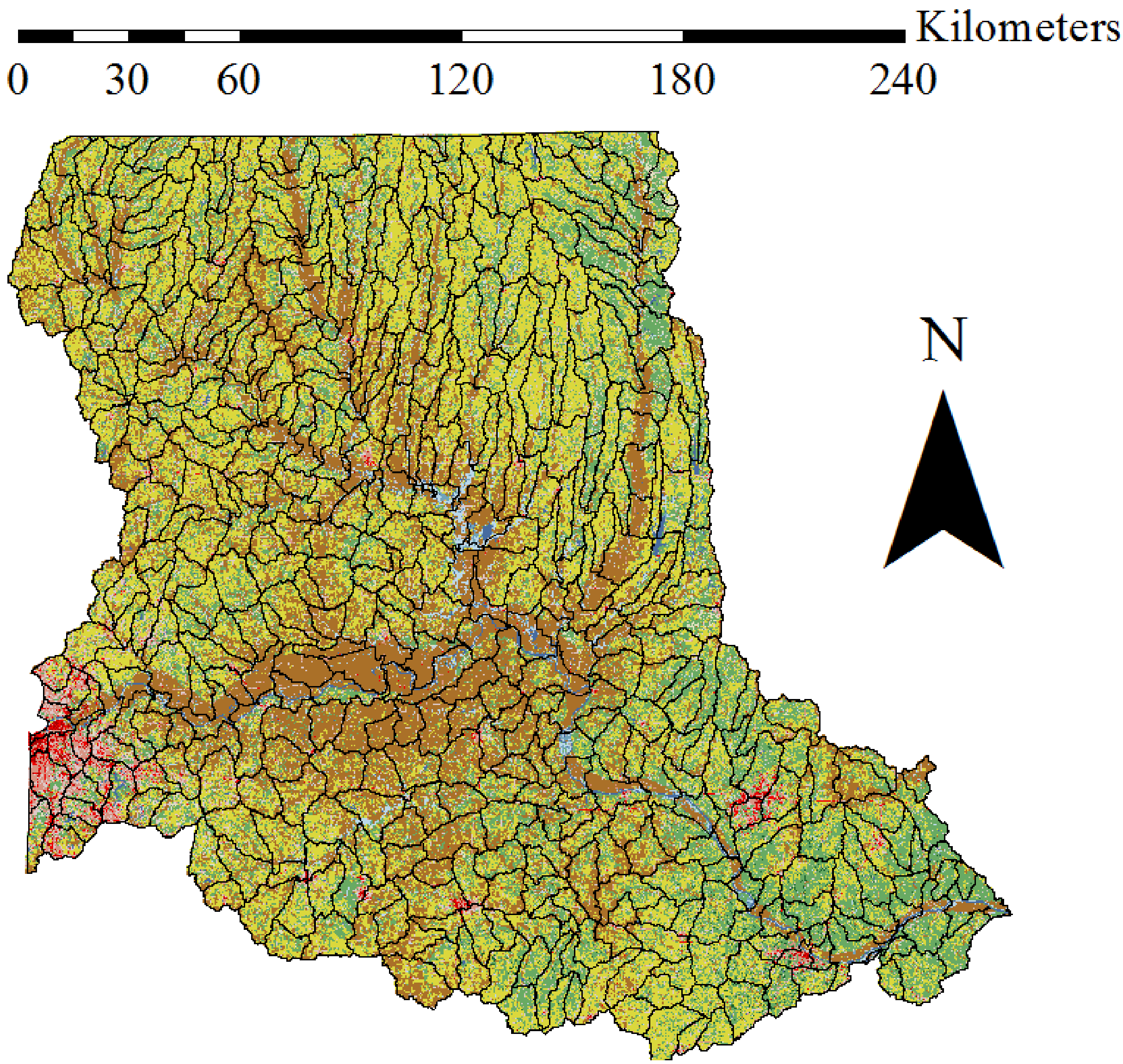

Figure 1.3. National Land Cover Dataset for the Grand and Missouri River Ecoregion, Missouri. Dominant land cover included cultivated (represented by brown), hay and pasture (represented by yellow), and deciduous forest (represented by green). Black delineations are hydrologic unit code 12 watersheds. Proportion of land cover in a watershed were predictor variables used to model probability of National Wetland Inventory wetland inundation in November years 2004-2006 and 2008-2010.

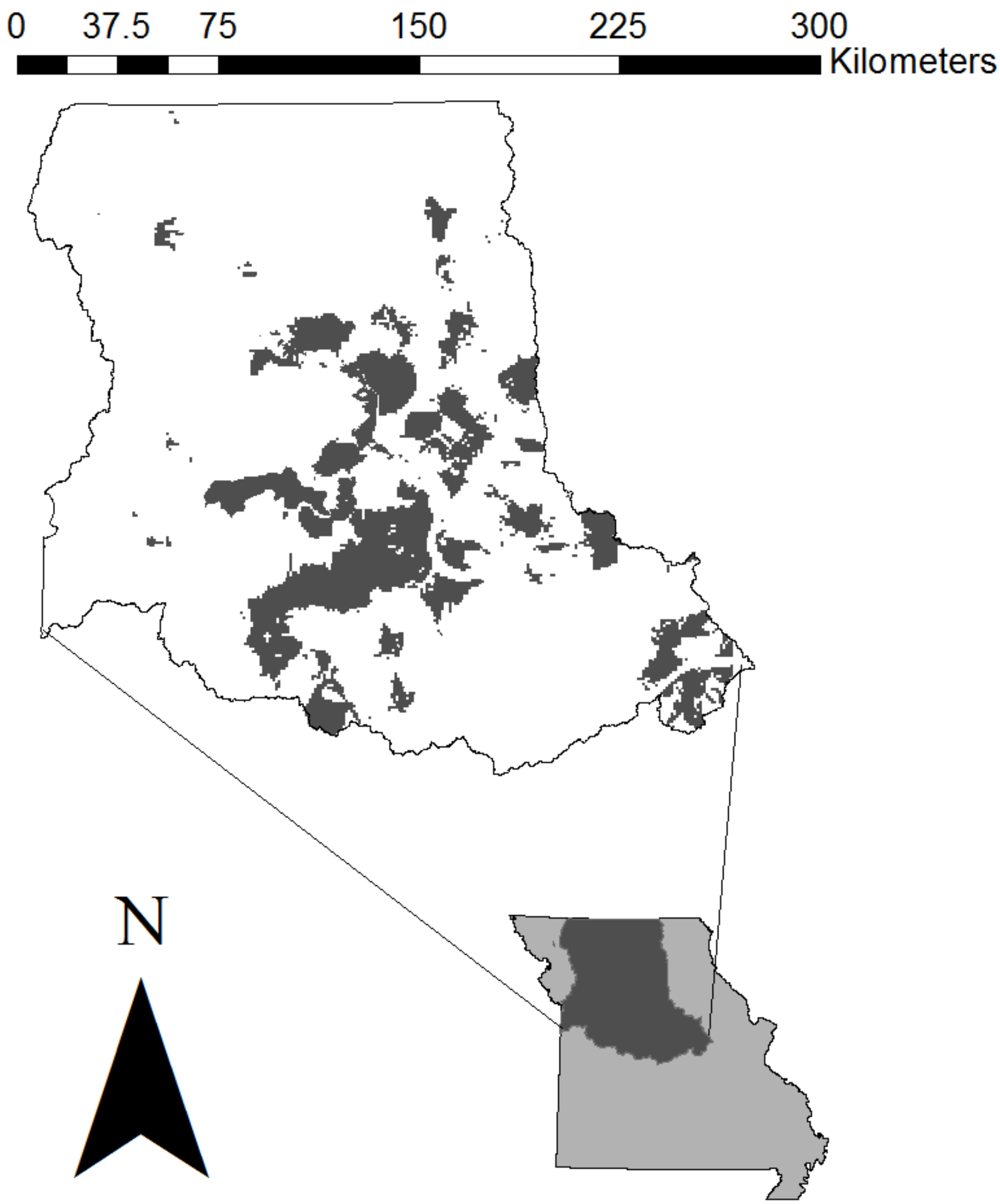

Figure 1.4. Predicted National Wetland Inventory wetland inundation (grey shaded areas) in November 2004-2006 and 2008-2010 in the Grand and Missouri River Ecoregion, Missouri. Inundation predictions were developed using generalized linear mixed models and ranked using AIC_c. A receiver operator characteristics (ROC) curve was used to identify an optimal cutoff value for wetland inundation predictions.

Chapter 2

CHAPTER II: DESIGN AND EVALUATION OF AN AERIAL STRIP-TRANSECT SURVEY TO ASSESS WATERFOWL ABUNDANCE AND DISTRIBUTION IN THE GRAND AND MISSOURI RIVER ECOREGION, MISSOURI

INTRODUCTION

At least 45 waterfowl species breed in North America and are distributed from the Canadian arctic to South America throughout the annual cycle (Nichols et al. 1995, Baldassarre 2014). Waterfowl are considered the most economically valuable assemblage of migratory bird species in North America and have received substantial attention in terms of conservation and management efforts over the last century (Brown and Hammack 1972, Nichols et al. 1995, Williams and Johnson 1995, Grado et al. 2001). Despite their relative importance, numerous migratory waterfowl species experienced drastic population declines influenced by drought, overharvest, and habitat conversion or degradation beginning in the early 20th century (Jordan and Bellrose 1951, Bellrose 1959, Williams et al. 1999). Because many North American waterfowl species are migratory and widely distributed throughout the annual cycle, regulatory collaboration among countries and states throughout their range is essential to stymie further declines of waterfowl populations (Williams et al. 1999, Nichols et al. 2007). In response to population declines, international regulatory and conservation efforts such as the Migratory Bird Treaty Act of 1918 and the North American Waterfowl Management Plan (NAWMP) explicitly outline protections and harvest regulations for migratory waterfowl across their North American range (Williams et al. 1999, Nichols et al. 2007, North American Waterfowl Management Plan Committee 2012). It is therefore important that scientists and wildlife managers develop and use robust methods to estimate regional waterfowl populations as a means for establishing and monitoring conservation efforts and monitoring populations over time (Reinecke 1992, Nichols et al. 1995, Williams et al. 1999, Nichols et al. 2007).

Monitoring migratory waterfowl populations is critical to developing harvest regulations, safeguarding current and future waterfowl populations, and developing landscape conservation strategies based on estimated waterfowl energetic demands (Reinecke et al. 1992, Brasher et al. 2002, Pearse et al. 2007, Flyways 2016). Population surveys on the breeding ground and wintering areas are used at the flyway scale to modify harvest regulations based on changes in estimated waterfowl populations (Eggeman and Johnson 1989, Brasher et al. 2002). For example, waterfowl population estimates obtained with aerial surveys are used in part to reduce uncertainty of harvest regulations in North America (Brasher et al. 1996). For example, the Midwinter Waterfowl Survey provides an index of waterfowl abundance and is used to inform habitat management and harvest regulation decisions in the Central Flyway and throughout the United States (Sharp et al. 2002). Waterfowl survey data are used to develop population based habitat conservation plans within Joint Ventures to assess regional landscape capability in meeting energetic demands of migrating waterfowl (Soulliere et al. 2007, Williams et al. 2014, Migratory Bird Joint Ventures 2016). Finally, state agencies use aerial waterfowl survey data to develop strategies for habitat acquisition, restoration, and maintenance (Raedeke et al. 2013).

Waterfowl surveys are an informative tool allowing researchers and managers to estimate and monitor waterfowl populations, habitat use and conditions, and develop population and habitat conservation goals (Smith 1995, Williams et al. 1996, Williams 1997, Brasher 2002, Pearse et al. 2007, Zimmerman et al. 2012). Notable examples of the utility of waterfowl surveys are the Waterfowl Breeding Population and Habitat (BPOP) survey and Mid-Winter Waterfowl Index (MWI) survey used to assess population changes, inform habitat conservation agendas, develop harvest regulations, and evaluate waterfowl response to changing land use (Nichols 1991, Nichols et al 1995, Butler et al. 1995, Smith 1995, Heusmann 1999, Smith 2002). Although aerial surveys are a primary tool for estimating waterfowl populations, many state and federal agencies use methods to estimate waterfowl abundance and habitat use that do not incorporate probability based sampling designs and thus do not include estimates of variance or precision (Eggeman and Johnson 1989, Reinecke et al. 1992, Heusmann 1999, Pearse et al. 2007). Further complicating estimation of regional waterfowl populations, migrating and wintering waterfowl are often highly aggregated and spatially clumped, making comprehensive survey design challenging (Nichols et al. 1983, Reinecke et al. 1992).

Many researchers assert that aerial survey designs can potentially lack statistical robustness due to inconsistencies in standardized sampling and reporting procedures (Eggeman and Johnson 1989, Heusmann 1999, Sharp et al 2002). Questions on the efficacy of survey methods used to monitor North American waterfowl populations have led to discussion on the necessity to adopt statistically rigorous design based survey methods to estimate waterfowl abundance (Reinecke et al. 1992, Pearse et al. 2007). Recognizing potential deficiencies in aerial survey methods, researchers have refined methodology for reducing variability and increasing confidence in winter waterfowl estimates (Conroy et al. 1988, Reinecke 1992, Pearse et al. 2007). Indeed, literature suggests that carefully designed aerial waterfowl surveys can provide precise estimates of waterfowl populations on their wintering range (Conroy et al. 1988, Reinecke 1992, Pearse et al. 2007). Precise estimates of wintering waterfowl abundance have been collected using surveys on wintering grounds known to support large waterfowl populations. For example, Conroy et al. (1988) designed an aerial survey for American black ducks (*Anas rubripes*) along the Atlantic coast and concluded their survey methods were acceptable for migratory waterfowl wintering grounds. Further, precise waterfowl estimates using aerial surveys have been generated for the lower Mississippi Alluvial Valley (MAV), an area known to contain high densities of migrating and wintering waterfowl on wetland habitats including open water, flooded agricultural fields, open emergent wetlands, and forested wetlands (Reinecke et al. 1992, Pearse et al. 2007). While aerial surveys have been demonstrated to be an effective tool for estimating waterfowl abundance in areas of known high waterfowl densities using specific wetland habitats, it is unclear if precise estimates of waterfowl abundance can be obtained through aerial surveys in regions with lower waterfowl densities and sparsely distributed wetland habitats.

The Grand and Missouri River floodplains in north-central Missouri were historically important to migratory waterfowl in autumn and winter as they provided stopover and wintering habitats waterfowl needed to rest and replenish energy reserves (Nigh and Schroeder 2002, USGS Land Cover Trends 2012). The Missouri Department of Conservation (MDC) has conducted autumn and winter aerial waterfowl surveys in north-central Missouri since 1952. The MDC aerial survey was implemented during a time when approximately 85% of all wetlands in the state had been converted for agricultural purposes, thus limiting waterfowl habitats primarily to publicly owned wetland areas (Nichols and Hines 1987, Nigh and Schroeder 2002, USGS Land Cover Trends 2012). Because of limited wetland habitat in the region, the MDC waterfowl survey focused on four intensively managed public wetland areas since the mid-1990's. Missouri Department of Conservation aerial surveys are typically flown on Monday mornings with the assumption that most waterfowl in the region are located on public wetland area refuges to escape weekend hunting pressure. For this reason, MDC waterfowl estimates of public wetland areas are considered to be a census of regional waterfowl populations based on the assumption that all waterfowl relocate to public wetland areas in response to weekend hunting pressure. However, due to recent wetland restoration efforts in Missouri, it is possible that many migrating waterfowl now occur beyond areas sampled by the traditional MDC aerial survey. For example, approximately 27,000 hectares of Wetland Reserve Program (WRP) wetlands have been established in the region since 1995 (personal correspondence, K. Dacey, Natural Resource Conservation Service) and may provide important habitat for migrating waterfowl. The current MDC aerial waterfowl survey is not design based because the survey area is relatively small and can be surveyed in its entirety. Thus, the data are expressed as raw counts with no estimates of variance or statistical confidence. Because of this, current methods for estimating waterfowl abundance by the MDC may not be adequate for sampling waterfowl populations within the entire region. Recognizing the possibility that substantial numbers of migratory waterfowl now use wetland habitat beyond traditional survey areas, the MDC identified the need to reassess current aerial waterfowl survey methods and explore the use of design based survey approaches to estimate waterfowl abundance and distribution in north-central Missouri.

I designed an aerial strip-transect survey for the Grand and Missouri River Ecoregion (GMRE; Figure 2.1) in north-central Missouri to assess waterfowl abundance and distribution in autumn and winter 2014 and 2015-2016. I compared waterfowl abundance estimates among strata and between MDC public area counts and the aerial strip-transect survey. I was unable to compare estimates of precision between MDC public area counts and the aerial strip-transect survey because MDC estimates were raw counts and thus did not include estimates of precision. I describe waterfowl abundance, and distribution in north-central Missouri and outline considerations to improve precision of survey methodology to increase confidence in regional waterfowl abundance estimates for areas of low wetland density.

STUDY AREA

I restricted the study area to a ~2 million ha subset of the GMRE (Figure 2.2) because survey time was limited by funding constraints and daylight hours. The GMRE is an ecologically important stopover and wintering area for migrating waterfowl in the Mississippi Flyway (Nichols and Hines 1987, North American Waterfowl Management Plan Committee 2012). The GMRE encompasses over 4 million hectares of floodplain and upland habitat that is currently dominated by agricultural land use (Nigh and Schroeder 2002, Nelson 2010, USGS Land Cover Trends 2012). Historically the large floodplains associated with the Grand and Missouri Rivers included wetlands, wet meadows, and bottomland forests (Nigh and Schroeder 2002). Upland habitat was historically driven by patterns of ridges and valleys creating a complex mosaic of open prairie and forest (Nigh and Schroeder 2002, Missouri Ecological Classification System 2013).

Although >85% of historic wetlands in the region have been converted to agricultural land use, wetlands restored through the WRP (~27,000 ha) have ensured that wetland habitats persist in the GMRE on private lands (personal correspondence, Kevin Dacey). In fact, the GMRE spans only 24% of the total land area of Missouri, but contains 55% of the WRP area in Missouri (personal correspondence, K. Dacey). Further adding to wetland area in the region, 4 intensively managed public wetland areas (Eagle Bluffs Conservation Area [CA; 1,859 ha], Grand Pass CA [2,040 ha], Fountain Grove CA (3,030 ha), and Swan Lake National Wildlife Refuge [NWR; 3,664 ha]) are scattered throughout the GMRE within the Grand and Missouri River floodplains (Figure 2.2). The 4 public wetland areas provide consistently available wetland habitat during hunting season through restricted hunting hours and refuge wetlands entirely closed to hunting. Migration and overwintering for most dabbling ducks in the region occurs between October and February, with peak numbers reaching >300,000 dabbling ducks (personal correspondence, public wetland area managers).

METHODS

Survey design

I used a stratified random strip transect survey design to assess waterfowl abundance and distribution in the GMRE (Conroy et al. 1988, Reinecke et al. 1992, Pearse et al. 2007). I stratified the study area based on expected differences in waterfowl densities and distribution (Conroy et al. 1988, Reinecke et al. 1992, Pearse et al. 2007). Previous waterfowl surveys conducted by the MDC in the GMRE indicate autumn and wintering waterfowl aggregate on the four intensively managed public wetland areas in the GMRE (Raedeke 2016, unpublished data). Global positioning satellite transmitters attached to mallards (*Anas platyrhynchos*) in the region indicate that daily foraging flights rarely exceeded 30km (Beatty et al. 2013), suggesting waterfowl densities are likely greater within 30km of an intensively managed wetland area. Further, the Grand and Missouri River floodplains have greater densities of WRP easements than the surrounding upland landscape, likely accumulating greater waterfowl densities than surrounding upland landscape (Chapter 1). Thus, I delineated the study area into four strata based on a 30 km distance from intensively managed wetland areas and floodplain/upland landscape (Figure 2.3). The first stratum included areas outside the Grand and Missouri River floodplain and beyond 30 km of a public wetland area (O-FP, >30km). The second stratum encompassed areas outside the Grand and Missouri River floodplain and within 30 km of a public wetland area (O-FP, <30km). The third stratum included areas within the Grand and Missouri River floodplain and beyond 30 km of a public wetland area (FP, >30km). The fourth stratum was located within the Grand and Missouri River floodplain and within 30 km of a public wetland area (FP,

<30km). If little a priori information is available for a species of interest in the study area, strata should be sampled using proportional allocation (i.e., in proportion to its area; Buckland et al. 2005). Because waterfowl abundance estimates were only available for the intensively managed wetland areas and a few WRP complexes in the GMRE, I used proportional allocation to designate sampling effort among survey strata.

I developed a Geographic Information System (GIS) database of 250m wide strip transects in ArcGIS for each stratum and oriented transects to avoid directly following landforms such as rivers and ridges (Buckland et al. 2005). Transect length varied among and within strata because strata varied by area and were not of uniform shape (Figure 2.3). Estimates of wetland distribution in the GMRE indicated that less than 4% of the area was classified as wetland habitat (USFWS 2016). Within each stratum I positively weighted transect selection based on wetland density (wetland area/ha) within a transect in order to increase the probability a selected transect would encounter potential waterfowl habitat (i.e., wetlands). Specifically, I calculated density of National Wetland Inventory (NWI) and WRP wetland habitat within each transect and subdivided the distribution of wetland density of transects into quartiles. Transect selection probability was weighted by quartile group, with transects in the upper quartile distribution of wetland area represented four times in the sampling universe, while transects in the lowest quartile were represented only once in the sampling universe. This sampling framework ensured that transects with greater wetland density had a greater chance of being selected for sampling. Transects were selected randomly with replacement for each survey, and I did not allow adjacent transects to be sampled during the same survey to minimize potential for double counting birds, (Reinecke et al. 1992, Pearse et al. 2007).

Sampling Methods

I flew nine aerial strip-transect surveys during autumn and winter 2014-2016 in a Cessna model 210 fixed wing aircraft. To maintain a consistent transect width, I used window markers calibrated for 250-m transect width at 150-m height on the passenger (starboard) side of the plane (Conroy et al. 1988). The aircraft flew at approximately 150-km/hour to ensure consistency in detection probability and detection rate (Conroy et al. 1988, Pearse et al. 2007). To estimate waterfowl abundance along transects, I used a double observer Peterson method, which requires two observers to simultaneously record the number of waterfowl within the transect borders while keeping counts and locations separate (Seber 1982, Seber 1986, Pollock and Kendall 1987). Observers recorded flock observation times and locations to identify which groups of waterfowl were observed by both observers. I flew three surveys in 2014 (11 November, 9 December, 23 December) and six surveys in 2015-2016 (2 November 2015, 20 November 2015, 4 December 2015, 30 December 2015, 11 January 2016, and 26 January 2016). Dabbling ducks (Genus *Anas*) are the most abundant assemblage of duck species in the region during autumn and winter, and mallards can account for 78% of dabbling ducks during fall migration (Raedeke et al. 2003). Unfortunately, duck species were often difficult to identify and many species can occur within a single group of waterfowl. For this reason, I combined all duck observations into a single category for statistical analysis. Canada goose (*Branta canadensis*) accounted for a large portion of detected waterfowl and were thus included in analysis. Because of their distinguishing characteristics and large size relative to dabbling ducks I was able to confidently differentiate Canada geese from other waterfowl species. Trumpeter swans (*Cygnus buccinator*) were detected during several surveys but low detection rate and sample size prohibited analysis (Appendix B).

I obtained aerial waterfowl estimates from four intensively managed public wetland areas (Eagle Bluffs CA, Grand Pass CA, Fountain Grove CA, and Swan Lake NWR) to compare to aerial survey estimates. Public wetland area aerial waterfowl surveys were flown biweekly and I used the closest date of public wetland area aerial survey estimates to compare to aerial strip-transect survey estimates. Public wetland area aerial waterfowl surveys were flown at an altitude >400m to collect overall waterfowl estimates and identify locations of flocks without flushing waterfowl from wetlands. Once flock locations were identified, altitude was decreased to collect species composition estimates of larger concentrations of waterfowl. When ducks were more heavily concentrated, the pilot would circle the concentration of ducks several times to ensure complete coverage from several angles (Personal correspondence, A. Raedeke, MDC). Further, estimates were pooled for each public wetland area and assumed to be a census of waterfowl in the region. Negligible numbers of Canada geese were detected on the public wetland area and were therefor excluded from comparison to aerial survey estimates.

Estimation and Analysis

I estimated population indices ($\hat{I}$; abundance estimate not corrected for observer bias) for all ducks (hereafter, ducks), Canada geese and all waterfowl species (all ducks and Canada Geese; hereafter total waterfowl) for each survey using the two sample capture-recapture Petersen estimator (Grier et al. 1981, Caughley and Grice 1982, Seber 1982, Pollock and Kendall 1987). The Petersen method requires two observers to collect observations simultaneously while keeping observations and locations of observations independent (Grier et al. 1981, Caughley and Grice 1982, Seber 1982, Pollock and Kendall 1987). Upon completion of a survey, observers compared individual observation times to identify groups of waterfowl detected by both observers or groups of waterfowl detected by one observer and missed by the other. Waterfowl groups detected by both observers within 5 seconds of each other were considered the same group. The Petersen Model states:

$$\hat{I} = \frac{(n_1+1)(n_2+1)}{m+1} - 1$$

Where

$\hat{I}$ = Estimated population indices for a transect

n_1 = number detected by the first observer

n_2 = number detected by the second observer

m = number detected by both observers

I calculated $\hat{I}$, standard errors (SE), and coefficients of variation (CV) for summed counts of each waterfowl group using the SURVEYMEANS procedure in SAS (SAS Institute, Cary, NC).

I estimated population abundance ($\hat{N}$) for a transect using a correction factor calculated using probability of detection for each observer (Grier et al. 1981, Caughley and Grice 1982, Pollock and Kendall 1987). I calculated probability of detection for the first observer as:

$$\hat{P}_1 = \frac{m}{m + n_2}$$

and probability of detection for the second observer as:

$$\hat{P}_2 = \frac{m}{m + n_1}$$

Where

$\hat{P}_1$= probability of a bird being detected by observer 1

$\hat{P}_2$= probability of a bird being detected by observer 2

I then calculated a correction factor ($\hat{C}$) from estimates of $\hat{P}$ which I multiplied by $\hat{I}$ to estimate $\hat{N}$ (Grier et al. 1981, Caughley and Grice 1982, Pollock and Kendall 1987):

$$\hat{C} = \frac{1}{\hat{P}}$$

When

$$\hat{P} = (\hat{P}_1 + \hat{P}_2)/2$$

I calculated standard errors of abundance estimates using 1000 bootstrapped samples because an explicit variance estimator was not available (Smith 1993, Cogen and Deifenbach 1998, Pearse et al. 2007). To estimate ecoregion waterfowl abundance, I first calculated waterfowl density as birds/ha for entire area sampled from each stratum by pooling density estimates and area sampled of each transect (Caughley and Grice 1981, Reinecke et al. 1992). I then multiplied the waterfowl density estimate by the total area of each stratum and pooled abundance estimates of each stratum to estimate ecoregion abundance (Caughley and Grice 1981, Reinecke et al. 1992). Finally, I calculated total waterfowl abundance for the GMRE by combining abundance estimates from aerial strip transect surveys and estimates from intensively managed public wetland areas.

I tested for differences of duck and Canada goose abundance among all strata by comparing waterfowl transect estimates using a one-way ANOVA to evaluate the null hypothesis that waterfowl abundance estimates were similar among strata in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA). To account for heterogeneity of variance among strata abundance estimates, I used the Welch's *t*-test for pairwise comparison of strata because other common comparison tests such as the student *t*-test and Mann-Whitney *U* test perform poorly in terms of both Type I and Type II errors when sample variance is unequal (Zar 1999, Ruxton 2006). Using the Welch's *t*-test, I evaluated whether abundance estimates differed between combined floodplain and non-floodplain stratum, and between >30km of a public wetland area and <30km of a public wetland area stratum in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA).

RESULTS

I completed 9 surveys during autumn and winter 2014-2016. I conducted 3 aerial surveys in autumn 2014 (11 November, 9 December, and 23 December) and 6 aerial surveys in 2015-2016 (2 November, 20 November, 4 December, 30 December, 11 January, and 26 January). A survey scheduled for late November 2014 was cancelled due to inclement weather and two January 2015 surveys were cancelled due to airplane malfunctions. I flew 407 transects, totaling 4,847 km over both years of the study period. I detected at least one waterfowl species on 29% (n=117) of transects and mean flock size was 140 ± 23 birds. I detected ducks on 19% (n=76) of transects and mean flock size was 169 ± 40, whereas Canada geese were detected on 10% (n=41) of transects and mean flock size was 67 ± 18.

Variance of abundance estimates was relatively high among all surveys and years (CV ≥0.36). Average CV of abundance estimates was 0.76 ± 3% for total waterfowl, 0.83 ± 4%) for ducks, and 0.88 ± 3%) for Canada geese (Table 2.1). Bias correction increased CV by 1% for total waterfowl and ducks, and did not increase CV for Canada geese thus producing a negligible decrease in precision of abundance estimates. Bias correction increased estimates of abundance for total waterfowl and ducks by an average of 11% ± 2%, and Canada geese by 6% ± 1%.

I compared duck and Canada goose abundance estimates among stratum to evaluate waterfowl distributions across the landscape (floodplain vs. non-floodplain) and in proximity to public wetland areas (Table 2.2). Duck abundance estimates varied among all stratum ($F_{3, 405}$= 2.89, p=0.04), as well as between combined floodplain and non-floodplain stratum. ($F_{1, 405}$= 8.65, p=0.004) and was greatest in the floodplain stratum ($\bar{x} = 0.19 \pm 0.06$ ducks/ha). Duck abundance estimates did not differ between FP, >30km and FP, <30km stratum ($F_{1, 210}$= 0.03, p=0.86), between the OFP, >30km and OFP, <30km stratum $F_{1, 192}$=0.74, p=0.39, or between combined >30km and <30km stratum ($F_{1, 405}$= 0.12, p=0.73). Canada goose abundance estimates differed between combined floodplain and non-floodplain stratum ($F_{1, 405}$= 4.71, p=0.03) and were greatest in the non-floodplain stratum ($\bar{x} = 0.03 \pm 0.01$ Canada geese/ha). Canada goose abundance did not vary among all strata ($F_{3, 405}$= 2.13, p=0.1), between OFP, >30km and OFP, <30km stratum ($F_{1, 194}$= 0.85, p=0.36), between FP, >30km and FP, <30km stratum ($F_{1, 210}$= 0.46, p=0.50), or between combined >30km and <30km stratum ($F_{1, 405}$= 0.67, p=0.41).

Duck density was greatest in the FP>30km stratum (0.15 birds/ ha) and lowest in the O-FP <30 km stratum (0.01 birds/ha) in 2014, whereas in 2015-2016 duck density was greatest in the FP <30km stratum (0.26 birds/ha) and lowest in the O-FP <30 km stratum (0.01 birds/ha; Table 2.3). Canada goose density was greatest in the O-FP <30 km stratum (0.05 birds/ha) and lowest in the FP >30km stratum (0.001 birds/ha) in 2014, while Canada goose density was greatest in the O-FP >30 km stratum (0.05 birds/ha) and lowest in the FP <30km stratum (0.1 birds/ha) in 2015-2016 (Table 2.3). Duck densities were greater within the Grand and Missouri River floodplains in both survey years (0.13 and 0.5 birds/ha, respectively) compared to outside the Grand and Missouri River floodplains (0.02 and 0.01 birds/ha, respectively; Table 2.3). Canada goose density was greater outside the Grand and Missouri River floodplain for both survey years (0.04 and 0.01 birds/ha, respectively) compared to within the Grand and Missouri River floodplain (0.003 and 0.001 birds/ha, respectively; Table 2.3).

Estimates of duck abundance during the 2014 study period peaked on 11 November 2014 with an estimated abundance of 86,209 (SE=2%; Figure 2.4). It is likely the number of mallards increased over the three survey sampling periods, although total number of dabbling duck numbers declined. Canada goose abundance estimates increased over the three survey sampling periods with a peak abundance estimate of 166,105 (SE=3%; Figure 2.4) on 23 December 2014. Estimates of duck abundance peaked on 11 January 2016 during the 2015-2016 study period with an estimated abundance of 198,499 (SE=8%; Figure 2.4). Estimates of Canada goose abundance estimates also peaked during the 2015-2016 study period on 11 January 2016 with an estimated abundance of 224,225 (SE=3%; Figure 2.4).

Aerial strip transect survey abundance estimates differed from intensively managed public wetland area abundance estimates although I could not evaluate statistical significance because public wetland area estimates consisted of a single sample with no measure of error (Table 2.4). Percentage ducks in the GMRE located outside intensively managed public wetland areas varied during the 2014 sampling period and peaked on 23 December with 15% of estimated ducks in the GMRE located outside intensively managed public wetland areas (Table 2.4). On average, 21% of dabbling ducks occurred outside intensively managed public wetland areas in 2014. Percentage of ducks in the GMRE located outside intensively managed public wetland areas also varied during the 2015-2016 sampling period and peaked 11 January with 54% of estimated ducks in the GMRE occurring outside intensively managed public wetland areas (Table 2.3). On average, 27% of dabbling ducks occurred outside intensively managed public wetland areas in 2015

DISCUSSION

Precision of aerial waterfowl survey estimates can be measured and expressed as the CV, and an a *priori* goal is often set at ≤0.25 to improve confidence in estimates (Conroy 1988, Reinecke et al. 1992, Pearse et al. 2008). Precision was relatively poor for all abundance estimates of ducks, Canada geese and all waterfowl species combined (CV ≥0.36) when compared to similar aerial strip transect waterfowl surveys (Conroy et al. 1988, Reinecke et al. 1992, Pearse 2008). In fact, an a priori goal of precision is often established at CV <0.15 when estimating waterfowl abundance during autumn and winter (Conroy et al. 1988, Pearse et al. 2008). Coefficient of variation is a measure of dispersion in a series of data relative to the mean and was likely elevated in my study due to pronounced variation of waterfowl distribution and flock size in the study area. For example, only a small proportion of transects sampled over the study period contained duck and Canada goose detections (19% and 10% respectively), although when waterfowl were detected, flock size was often large but varied greatly $\bar{x}$ = 140 ± 23), thereby inflating variance estimates. Further, low estimated density of waterfowl in the study area likely contributed to poor precision of abundance estimates. Indeed, estimated waterfowl densities in the study area were significantly lower than similar aerial waterfowl surveys designed for other regions (Conroy et al. 1988, Reinecke et al. 1992, Pearse 2008). For example, an aerial strip transect survey using similar methods in the Mississippi Alluvial Valley considered <0.111 ducks/ha low density, 0.111-0.410 ducks/ha medium density, and >0.410 ducks/ha high density (Pearse et al. 2008). By those standards, my survey estimates indicate three of the four strata in 2014 and two of the four strata in 2015-2016 contain low duck densities, while no stratum contained high densities of ducks. Further, estimates indicate that all four strata contained only low densities of Canada geese during the entire study period ($\leq$0.05 Canada geese/ha).

It is widely accepted that observer bias contributes negatively to the estimation of apparent wildlife abundance (Caughley 1974, Caughley and Grice 1982, Pollock and Kendall 1987). To account for observer bias, researchers have worked to develop methods of establishing a correction factor to adjust estimates to absolute abundance (Kendall and Pollock 1987). I corrected abundance estimates using probability of detection calculated for both observers to account for individual observer bias in detecting waterfowl. There was no tradeoff between precision of abundance estimates and correction for observer bias because CV increased only slightly for both ducks and Canada geese (1% and <1% respectively). Further, bias correction increased estimates of ducks $\bar{x}$ =10% (SE=2%) and Canada geese $\bar{x}$ = 11% (SE=1%) among all surveys for both years yielding estimates likely closer to reality. My results complement a body of investigation recommending correcting observer bias to increase precision of survey estimates (Caughley 1974, Caughley 1976, Cook 1979, Caughley and Grice 1982, Reinecke 1992, Pearse et al. 2008).

It is recommended that prior information about detection rate, density, and distribution be used to design an aerial waterfowl survey and determine required sampling effort for a desired level of precision (Conroy 1988, Reinecke 1992, Pearse et al. 2007). Unfortunately, a priori knowledge of waterfowl distribution and abundance was only available for the Missouri River stratum prior to the design of the aerial survey. Consequently, I was limited to stratifying the study area based on factors likely influencing duck distribution in the study area. For example, I hypothesized that waterfowl densities were greater within the Grand and Missouri River floodplain where wetland restoration and availability is most prominent (Chapter 1). My results indicate that duck abundance is much greater in the Grand and Missouri River floodplains when compared to duck abundance in the two stratum outside the floodplain. Indeed, 98% of ducks were detected in the floodplain strata although the floodplain strata comprised only 10% of the study area. I also hypothesized that waterfowl densities were greater within 30km of a public wetland area because Beatty et al. (2013) found that daily foraging flights for mallards rarely exceeded 30km and the only documented concentrations of waterfowl in the study area were collected during public wetland area waterfowl surveys. My results indicate that public wetland areas likely do not influence the distribution of a large proportion of dabbling ducks in the study area because I was unable to detect differences in abundance estimates between the FP, >30km and FP, <30km strata.

Although I did not stratify the aerial survey by expected Canada goose density, I was able to make inferences on their abundance and distribution in the study area. Unlike ducks, Canada goose abundance was much greater outside the floodplain strata. Consequently, 99% of Canada geese in the study area were detected outside the Grand and Missouri River floodplain. Canada goose populations have increase substantially since the last half of the 20th century and appear to take advantage of human altered landscapes such as parks, golf courses, country clubs, farm ponds, and urban areas (Conover and Chasco 1985, Maccarone and Cope 2004, Unkless and Makarewicz 2007). Although I did not quantify Canada goose habitat availability, I suspect that human altered landscapes utilized by Canada geese are more prevalent outside the Grand and Missouri River floodplain. For example, nearly all Canada geese detections occurred on farm ponds surrounded by mowed lawns and pasture, a habitat type rarely observed within the Grand and Missouri River floodplain (B. Hidden, personal observation). Also, I did not detect differences in Canada goose abundances between the FP, >30km and FP, <30km strata. It is not surprising that public wetland areas did not appear to influence Canada goose distribution because nearly all Canada geese in the study area occurred outside public wetland areas.

MANAGEMENT IMPLICATION

Sampling to quantify abundance and distribution of waterfowl in the GMRE has been limited to public wetland areas and a three year study within the Missouri River floodplain (Raedeke et al. 2003). To my knowledge, autumn and winter aerial waterfowl surveys in the GMRE are limited to public wetland areas leaving much uncertainty in regional waterfowl abundance estimates. Although precision of my aerial survey estimates are not optimal, my results suggest that 9-54% of waterfowl, particularly ducks, in the region are not accounted for in autumn and winter by exclusively sampling public wetland areas. My study demonstrates that though the Grand and Missouri River floodplains are a small proportion of the GMRE, a majority of ducks in the GMRE occur in this area of the region. This knowledge can direct future sampling designs aimed at quantifying duck abundance in the region. Indeed, researchers can increase precision of estimates by directing future sampling efforts to the Grand and Missouri River floodplain as well as public wetland areas.

Wetland habitat has declined by >85% in the region suggesting that remaining wetlands are exceedingly important to migrating waterfowl (Dahl and Allord 1996, Gibbs 2000, Dahl 2011). My data indicate that conserving wetland habitat outside public wetland areas is important to providing autumn and winter habitat to migrating waterfowl as several surveys indicate a large proportion of waterfowl detected in the region occur outside public wetland areas. Continuing wetland conservation efforts in floodplain areas known to attract waterfowl will likely complement existing wetland habitat restoration and management actions within the region. I recommend researchers and managers assess changes in waterfowl abundance in the region in response to conservation efforts and management using a design based aerial waterfowl survey.

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Table 2.1. Indices, standard error, and mean coefficient of variation (CV) of abundance estimates after bootstrapping estimates for an aerial waterfowl survey autumn/winter 2014-2016 in the Grand and Missouri River Ecoregion, Missouri. I analyzed abundance estimates by Canada Goose (CAGO), all waterfowl species combined (Waterfowl Combined), and all duck species combined (Duck).

	Combined[1]			Duck			CAGO		
Date	$\hat{I}$	SE	CV	$\hat{I}$	SE	CV	$\hat{I}$	SE	CV
11-Nov-2014	93,950	10,335	0.68	86,289	7,766	0.80	7,661	460	0.85
9-Dec-2014	97,351	6,815	0.71	60,383	3,019	0.94	36,969	3,697	0.78
23-Dec-2014	226,051	18,084	0.67	59,965	5,397	0.81	166,105	16,611	0.75
2-Nov-2015	61,949	6,195	0.70	45,508	5,006	0.71	16,441	822	0.94
20-Nov-2015	68,639	10,296	0.69	57,026	8,554	0.81	11,612	1,974	0.84
4-Dec-2015	202,753	20,275	0.65	127,265	16,544	0.68	75,489	6,794	0.90
30-Dec-2015	43,541	34,833	0.88	9,823	589	0.94	33,719	337	0.99
11-Jan-2016	422,724	46,500	0.91	198,499	31,760	0.90	224,225	13,454	0.96
26-Jan-2016	67,395	33,698	0.92	67,395	3,370	0.94	0	0	0.94

[1]Pooled Duck and Canada Goose estimates from each survey date

Table 2.2. Mean and standard error of duck and Canada Goose (CAGO; birds/ha) density estimates for an aerial strip transect survey performed autumn 2014 and autumn and winter 2015-2016 in the Grand and Missouri River Ecoregion, Missouri.

	Stratum					
	O-FP[a] (>30km)	O-FP (<30km)	FP (>30km)	FP (<30km)	Floodplain	Outside floodplain
Ducks[b]	0.02 ±0.006 A[c]	0.01±0.005 AB	0.20±0.06 AB	0.18±0.11 B	0.19±0.06 C	0.02±0.004 D
CAGO	0.04±0.02 A	0.02±0.01 AB	0.01±0.01 AB	0.01±0.004 B	0.005±0.004 C	0.03±0.01 D

[a]Refer to figure 2.3 for descriptions of stratum

[b]Means and standard errors calculated per transect for both years of the aerial surveys

[c]Means sharing the same letter within rows do not differ ($p < 0.05$) based on Welch's t-test pairwise comparison

Table 2.3. Comparison of mean duck and Canada Goose (CAGO; birds/ha) density over three sample periods in 2014 and six sample periods in 2015-2016 for an aerial waterfowl survey in the Grand and Missouri River Ecoregion, Missouri. Aerial survey sampling effort was proportionally allocated to strata for all waterfowl surveys.

	Waterfowl Combined		Duck		CAGO	
Strata	2014	2015	2014	2015	2014	2015
O-FP (>30 km)[a]	0.06	0.07	0.03	0.02	0.03	0.05
O-FP (<30 km)[b]	0.06	0.02	0.01	0.01	0.05	0.02
FP (>30km)[c]	0.11	0.25	0.11	0.25	0.01	0.01
FP (<30km)[d]	0.17	0.25	0.15	0.26	0.01	0.01
Outside Floodplain[e]	0.06	0.06	0.02	0.01	0.04	0.03
Floodplain[f]	0.14	0.25	0.13	0.25	0.003	0.01

[a]O-FP (>30 km) indicates the strata outside a floodplain beyond 30 km of an intensively managed public wetland area.

[b]O-FP (<30 km) indicates the strata outside a floodplain and within 30km of an intensively managed public wetland area.

[c]FP (>30 km) indicates the strata within the Grand or Missouri River floodplain beyond 30 km of an intensively managed public wetland area.

[d]FP (<30 km) indicates the strata within the Grand or Missouri River floodplain within 30 km of an intensively managed public wetland area.

[e]Outside Floodplain indicates the combined strata O-FP (>30 km) and O-FP (<30 km).

[f]Floodplain indicates the combined strata FP (>30 km) and FP (<30 km).

Table 2.4. Comparison of public wetland area waterfowl ground estimates to aerial waterfowl survey estimates in the Grand and Missouri River Ecoregion, Missouri. Public wetland area estimates and aerial survey estimates were conducted on the same day or within two days of each other.

Date	Public wetland area estimate	Aerial survey estimate	Percent ducks outside public wetland areas
11 November 2014	234,949	82,209	26%
9 December 2014	247,898	60,383	20%
23 December 2014	340,220	59,965	15%
Mean			**20%**
2 November 2015	84,035	45,508	35%
20 November 2015	247,133	57,026	19%
4 December 2015	450,218	127,264	22%
30 December 2015	102,541	9,822	9%
11 January 2016	166,626	198,499	54%
26 January 2016	59,062	67,394	53%
Mean			**32%**

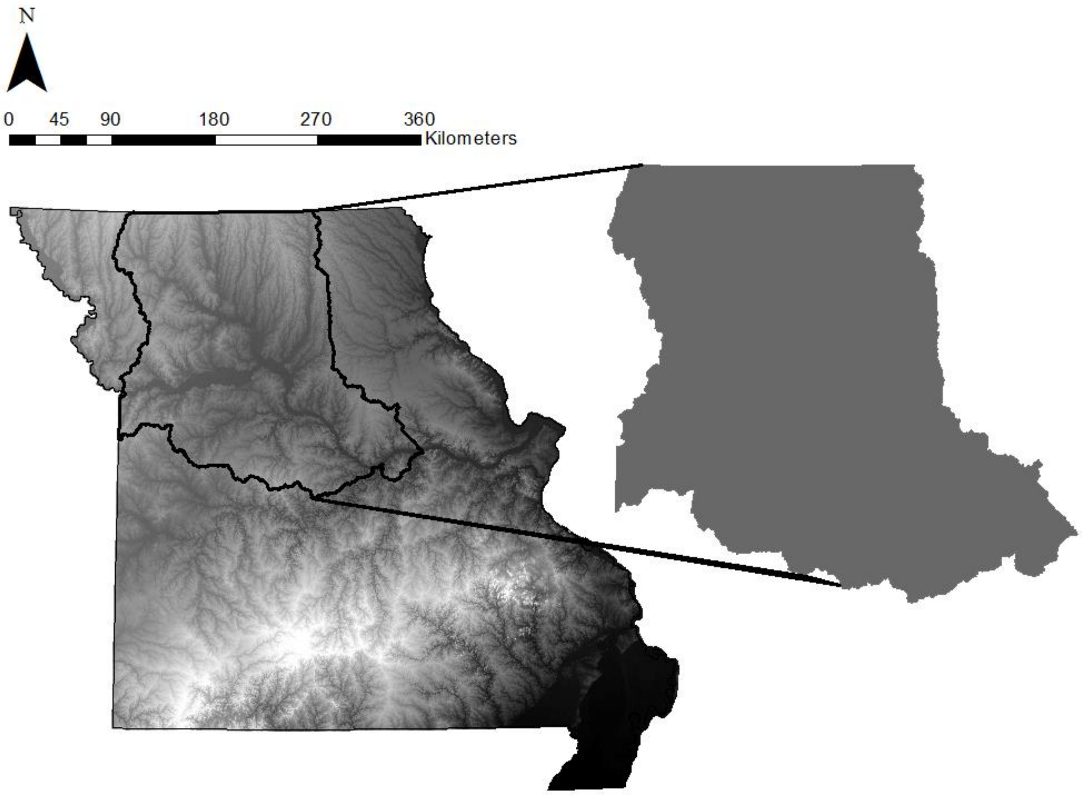

Figure 2.1. The Grand and Missouri River Ecoregion in North-central Missouri where aerial strip-transect surveys were flown to assess waterfowl abundance and distribution during autumn and winter 2014-2016. The shading gradient is a digital elevation model with highest elevations represented as white and lowest elevations represented as black.

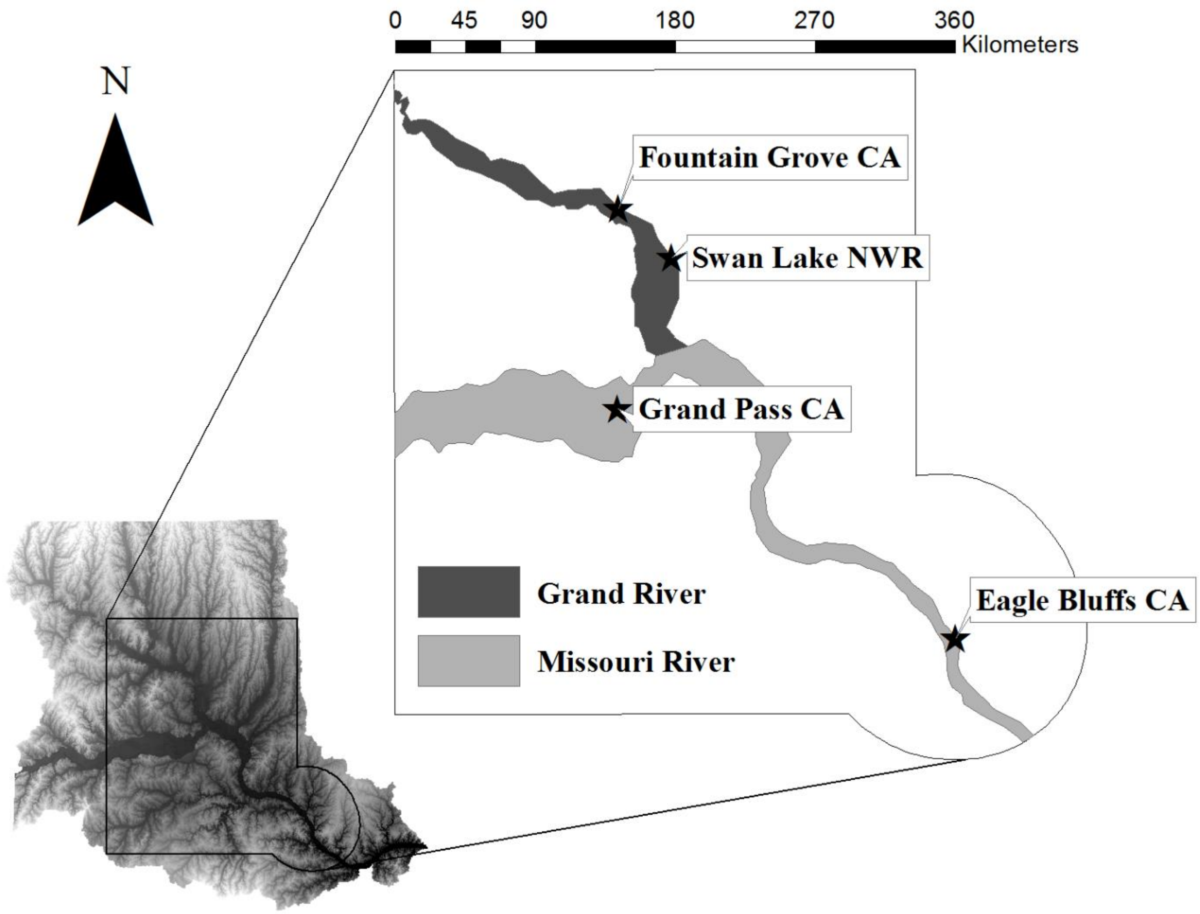

Figure 2.2. The study area located within the Grand and Missouri River Ecoregion (GMRE) in North-central Missouri. I constricted the study area because survey time was limited by funding constraints and daylight hours. The study area contains the Grand and Missouri River floodplains, three intensively managed wetland Conservation Areas (CA), and one intensively managed National Wildlife Refuge (NWR).

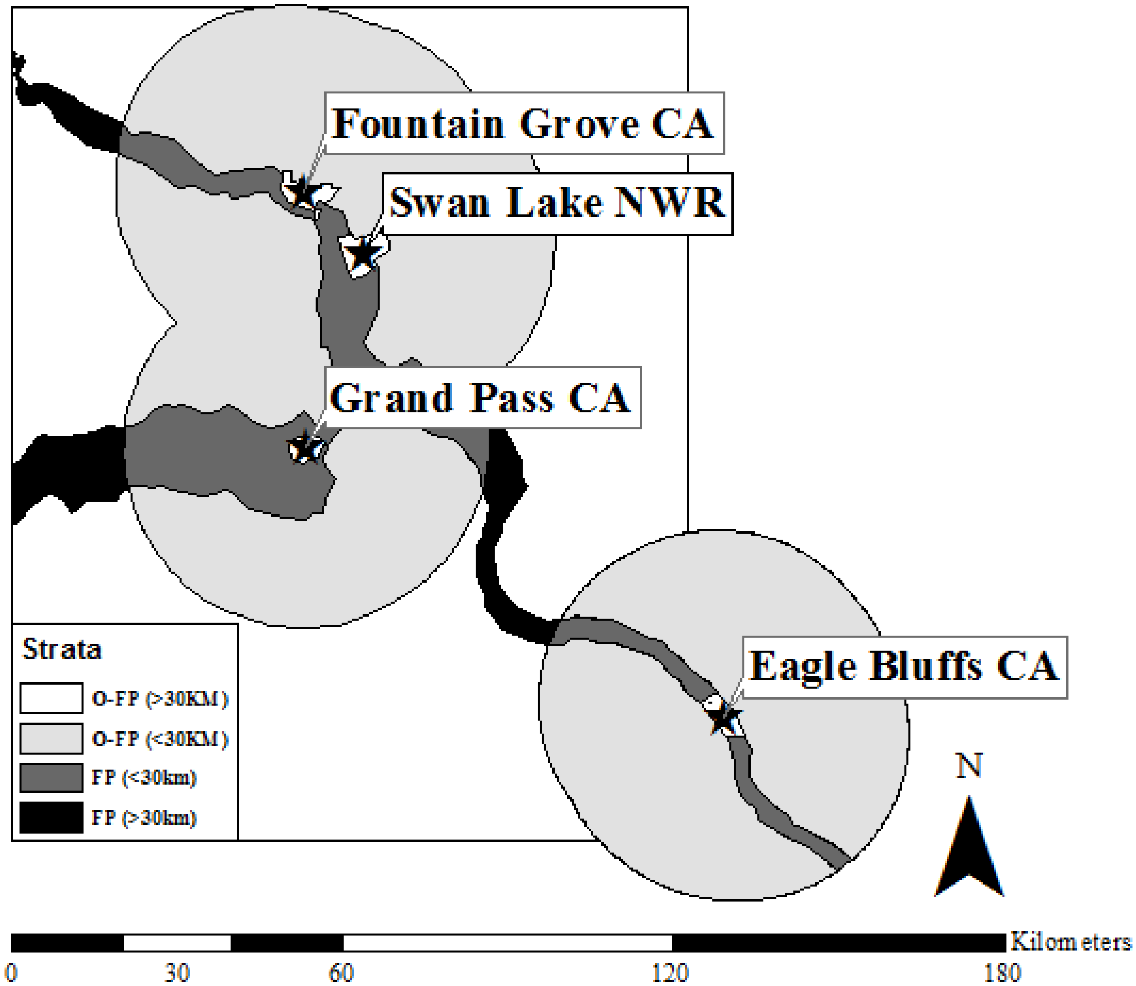

Figure 2.3. I expected waterfowl densities to be greater within 30km of an intensively managed wetland area and within the Grand and Missouri River floodplains. I delineated the study area into 4 strata; O-FP indicates a stratum outside a floodplain and either beyond or within 30km of an intensively managed wetland area. FP indicates a stratum within the Grand or Missouri River floodplain and either beyond or within 30km of an intensively managed wetland area. Aerial survey sampling effort was proportionally allocated to strata for 2014 and 2015-2016 waterfowl surveys.

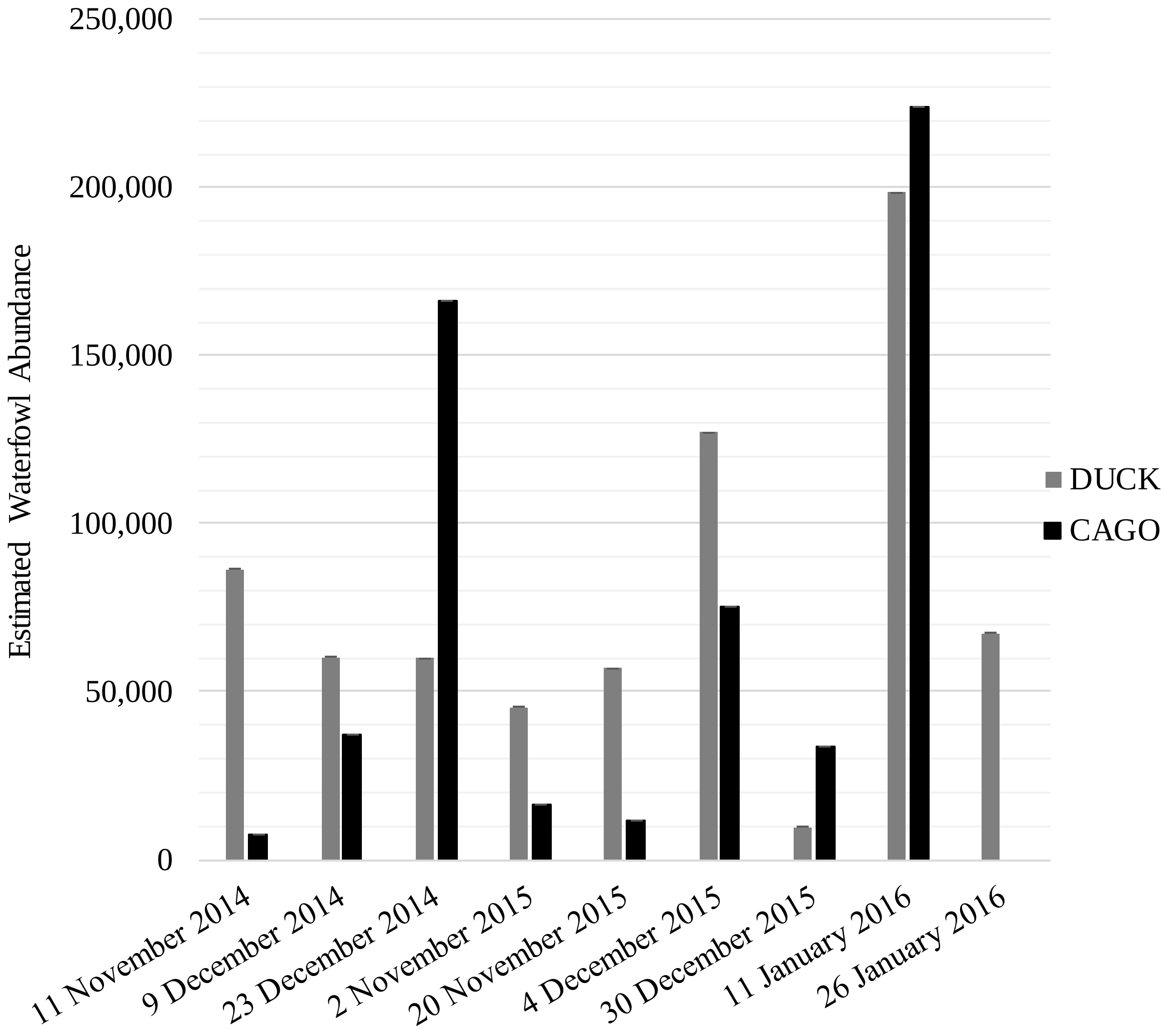

Figure 2.4. Dabbling duck (DUCK) and Canada goose (CAGO) abundance estimates from aerial strip transect surveys in north-central Missouri in autumn and winter 2014-2016.

Chapter 3

EFFECTS OF WETLAND HABITAT AVAILABILITY ON AUTUMN DABBLING DUCK ABUNDANCE ON PUBLIC WETLAND AREAS IN THE GRAND AND MISSOURI RIVER ECOREGION, MISSOURI

INTRODUCTION

Migratory waterfowl use a range of habitats throughout the annual cycle to meet specific seasonal life functions such as reproduction, migration, and overwintering (Flake 1979, Baldassarre and Bolen 1984, DuBowy 1988, Kaminski and Elmberg 2014). Further, waterfowl may need to adjust to within season environmental pressures such as food depletion, predation, competition, and hunting presence (Schranck 1972, Flake 1978, Kaminski and Prince 1984, Evans and Day 2002, Webb et al. 2010). Seasonal habitat conditions can vary greatly throughout the migratory waterfowl range and lack of wetland habitat may limit resources critical to completing important life functions (Dubowy 1988, Webb et al. 2010, Emery et al. 2013, Davis et al. 2014). Specifically, habitat availability and food resources may be most limited during autumn migration stopover and winter when competition for food resources among individual ducks is most intense (Fretwell 1972, Dubouy 1988, Heitmeyer 1988). Hence, migration stopover and winter habitat availability is exceedingly important to individual fitness because winter survival is necessary to complete spring migration and reproduction (Heitmeyer and Fredrickson 1981, Kaminski and Gluesing 1987, Davis et al. 2014, Kaminski and Elmberg 2014). For this reason it is critical that wetland managers understand autumn and winter waterfowl habitat requirements to better meet physiological demands of migration and overwintering waterfowl.

Despite significant ecological values to waterfowl, wetlands have been converted or altered at unprecedented rates in recent times. In fact, the U.S. has lost 53% of its wetlands since the 1780's, with remaining wetlands undergoing some form of alteration leading to reduced function (Dahl and Allord 1996, Gibbs 2000). The conservation of waterfowl populations is fundamentally linked to the habitats waterfowl use throughout the annual cycle (Porter and Kooten 1993, Brown et al. 2002, King et al. 2006, Soulliere et al. 2007, U.S. Fish and Wildlife Service 2009, North American Waterfowl Management Plan Committee 2012, Migratory Bird Joint Ventures 2016). Thus, the concern for waterfowl habitat loss has in part led to regulations and programs aimed at decreasing overall rates of wetland loss in the U.S. (Brown et al. 2002, King et al. 2006, Dahl 2011, EPA 2013). For example, wetland conservation on public and private grounds have been instrumental in conserving migratory waterfowl populations since the 1980's (Williams et al. 1999, Musacchio and Coulson 2001, King et al. 2006). Indeed, programs such as the North American Waterfowl Management Plan (NAWMP), North American Waterfowl Conservation Act (NAWCA), and Wetland Reserve Program (WRP) have decreased rates of wetland loss ensuring waterfowl have access to wetland habitat across their migratory range (Williams et al. 1999, King et al. 2006, Dahl 2011, Patterson 2008).

Wetland inundation and food availability are important factors influencing dabbling duck distribution and habitat selections during migration (Pearse et al. 2012, Williams et al. 2014). Dabbling ducks meet their food energy requirements by foraging in a variety of wetland and agricultural habitats, with wetland plant seeds and waste agricultural grains comprising up to 97% of a mallards (*Anas platyrhynchos*) diet in autumn and winter (Fredrickson and Taylor 1982, Baldassarre and Bolen 1984, Gruenhagen and Fredrickson 1990, Hagy et al. 2012, Kaminski and Elmberg 2014). Consequently, a management practice known as moist-soil management focuses on promoting plant species whose seeds are frequently consumed by dabbling ducks during nonbreeding portions of the annual cycle, and has become a primary method of migratory waterfowl habitat management with over 80% of federal refuges in the Mississippi Alluvial Valley (MAV) using moist-soil management techniques (Fredrickson and Taylor 1982, Gruenhagen 1987, Gruenhagen and Fredrickson 1990, Checkett et al. 2002, Hagy and Kaminski 2012). Thus, when managed wetlands become inundated they have potential to provide an abundance of food energy required by migrating dabbling ducks in autumn.

Waterfowl migration between breeding and non-breeding grounds includes extensive movements and selection of mid-migration and winter stopover sites is often critical to survival (Kaminski and Elmberg 2012, Davis et al. 2014). Mounting evidence suggests that food availability on individual wetlands does not exclusively determine initial habitat selection by migrating waterfowl. In fact, landscape scale availability of wetland resources likely influences habitat selection during migration events (Ma et al. 2009, Beatty et al. 2014, Davis et al. 2014, Kaminski and Elmberg 2014). Subsequently, finer scale factors such as local wetland availability, food abundance and complexity of habitat structure may influence wetland selection and interval of time spent in a local area (Brown and Dinsmore 1986, Soulé and Simberloff 1986, Ma et al. 2009, Beatty et al. 2014). Though many researchers agree that habitat quality and availability influences habitat selection, the lack of research aimed at quantifying within season estimates of habitat availability and how it influences habitat selection may be the largest gap in understanding how waterfowl use landscapes to fulfill life requirements (Davis et al. 2014).

Missouri, along with four other Midwestern states has experienced >85% wetland conversion from agricultural practices, urbanization, dam construction and water diversion (Tiner 1984, Dahl and Allord 1996, Gibbs 2000, Dahl 2011). Despite the recent loss of wetlands in Missouri, the WRP has restored over 59,000 hectares of wetlands on private lands in Missouri since the 1990's (NRCS 2013, K. Dacey, personal correspondence). The increase of wetland areas in Missouri raises important questions about wetland habitat dynamics throughout the state, their role in providing food energy required by migrating and wintering dabbling ducks, and their potential to influence dabbling duck (Anatini) distribution on public wetland areas. It is likely that availability of remaining wetlands and restored WRP wetlands influence dabbling duck abundance in the region (King et al. 2006). To my knowledge, no work has assessed how the combination of food abundance and wetland availability influences dabbling duck distribution in North Missouri.

Understanding how local food abundance and habitat availability influences dabbling duck use of public wetland areas is paramount to managing landscapes to meet energetic requirements of migrating waterfowl in autumn and winter. For example, wetland researchers and managers can better meet the waterfowl nutritional demands during autumn migration and winter by understanding how and why migrating ducks choose wetland habitat along migratory routes. Therefore, my objectives were to assess moist-soil seed abundance on four public wetland areas and WRP units before the onset of the autumn dabbling duck migration in the Grand and Missouri River Ecoregion (GMRE). Further, I assessed the relationship between wetland habitat availability outside the four public wetland areas and waterfowl abundance on public wetland areas during autumn and winter.

STUDY AREA

The GMRE (Figure 3.1) is an ecologically important migration stopover and wintering area for waterfowl in the Mississippi Flyway (Nichols and Hines 1987, North American Waterfowl Management Plan Committee 2012). In fact, waterfowl surveys reveal that >300,000 waterfowl use the GMRE at the peak of autumn migration and winter (Personal correspondence, public wetland area managers). The GMRE is over four million ha characterized by upland rolling hills and large floodplains associated with the Grand and Missouri Rivers (Nigh and Schroeder 2002, Missouri Ecological Classification System 2013). The GMRE landscape is now dominated by agricultural land use which has lead to conversion of >85% of historic wetlands in the region. Further, 4 public wetland areas (Eagle Bluffs Conservation Area [CA; 1,859 ha], Grand Pass CA [2,040 ha], Fountain Grove CA (3,030 ha), and Swan Lake National Wildlife Refuge [NWR; 3,664 ha]) are located within the Grand and Missouri River floodplains and provide consistent autumn and winter wetland habitat to migrating waterfowl (USGS Land Cover Trends 2012, EPA Ecoregions of Iowa and Missouri 2014). Recently, wetlands have been restored on private lands through the WRP with 543 easement spanning >27,000 ha and comprising 54% of the WRP area in the state (personal correspondence, K. Dacey). The 4 public wetland areas provide consistently inundated wetland habitat throughout the autumn and winter seasons. Further, public wetland areas provide sanctuary from hunting pressure to migrating waterfowl through restricted hunting hours and refuge wetland areas closed to hunting. Migration and overwintering for most dabbling ducks in the region occurs between October and February, and peak numbers can reach >300,000 dabbling ducks (personal correspondence, public wetland area managers).

METHODS

Seed sampling

I collected seed biomass estimates from four public wetland areas and 24 private wetlands enrolled in the WRP in the GMRE October 2014 and 2015 (Figure 2.3). I randomly selected three independent units within each public wetland (n=12) and 3 WRP units (n=12) within 30km of each public wetland area. Wetland Reserve Program sites were selected within a 30 km buffer of public areas because Beatty et al. (2013) found that mallard (*Anas platyrhynchos*) daily foraging flights in the region rarely exceeded 30km (Figure 3.2). By sampling within 30km of a public wetland area I could compare food abundance between public wetland areas and WRP wetlands that were hypothetically equally available to the same dabbling ducks. I randomly selected eligible WRP sites that met the following criteria: 1) ≥ 20 hectares, 2) date of restoration was ≥ 3 years from initial sampling date, 3) and access granted by land owner (Evans-Peters et al. 2012).

I collected seed head clippings and soil core samples from each wetland after seed heads appeared to have matured (Reinecke and Hartke 2005, Evans-Peters et al. 2012, Olmstead et al. 2013). Some units were naturally inundated before sampling began due to rain and flooding events and sampling was performed in inundated wetland conditions limiting the identification of individual plant species. I collected seed head clippings, and soil core samples at ten random points and intervals along transects oriented diagonally between the furthest corners of each wetland unit (Olmstead et al. 2013). At each sampling location, I clipped all seed heads connected to plant stalks emerging from soil within a 25-cm^2 quadrat (Greer et al. 2007, Evans-Peters et al. 2012) and collected a soil core sample within each 25-cm^2 quadrat using a 10cm wide x 5cm deep (392.7cm^3) soil core sampler (Reinecke and Hartke 2005, Kross et al. 2007, Olmstead et al. 2013). Seed head clippings were stored in paper bags in a dry location until they reached constant mass and soil core samples were frozen to prevent decomposition until the samples could be processed (Kross et al. 2008, Olmstead et al. 2013, Evans-Peters et al. 2012).

I processed seed head clippings by separating seeds from vegetative matter and any other non-seed material using a series of graduated sieves, air separation, manual gravity, and by hand

(Harmond et al. 1968, Evans-Peters et al. 2012). I selected specific sieve aperture size and shape (2-mm, 1-mm, 500-µm, 355-µm, and 250-µm; round, triangle, and square holes) based on plant species composition and seed size. Soil core samples were thawed and soaked in a 3% hydrogen peroxide solution and rinsed through graduated sieves (2.0-mm aperture and 300-µm) separating coarse and fine vegetative matter (Reinecke and Hartke 2005, Olmstead et al. 2013). I dried vegetative matter at 87C for ≥24 hours to a constant mass. I separated all seeds and tubers from the coarse matter using forceps and recorded mass to the nearest 0.1 mg (Evans-Peters et al. 2012, Olmstead et al. 2013). I divided fine vegetative matter equally by mass into four equal subsamples and selected a random subsample for processing (Kross 2008, Olmstead et al 2013). I used forceps and an illuminated 1.25X magnification desk mounted lens to separate seeds from fine vegetative matter and recorded mass to the nearest 0.1 mg. (Olmstead et al. 2013). I removed seeds of species unlikely to be consumed by dabbling ducks like Cocklebur (*Xanthium strumarium*) because their contribution to biomass estimates would inflate beneficial seed biomass estimates (Fredrickson and Taylor 1982, Olmstead et al. 2013). Identification of individual plant species in the field was challenging in 2014 because 22 of the 24 wetland units sampled were inundated during the sampling period. Therefore, I limited reporting several plant species observations to genera.

Statistical analysis

I combined seed head clipping and soil core seed biomass estimates for each sampling point and standardized estimates to kg/ha to generate a single seed biomass value for each sampling point (Evans-Peters et al. 2012). I evaluated total mean (±SE) seed mass (kg/ha) separately to evaluate differences between years (2014 versus 2015) because weather conditions could have varied and management regimes may have changed. I compared wetland type (public wetland area versus WRP) because potential differences in management intensity and ability to manipulate wetland conditions may influence seed production (Kross et al 2008, Hagy et al. 2011, Olmstead et al. 2013). I rank transformed seed biomass and used a one-way analysis of variance (ANOVA; PROC NPAR1WAY, SAS Institute, Inc., Cary, NC, USA) to test for differences between wetland types for both years of the study period.

Influence of habitat availability on public wetland area duck abundance

I was interested in assessing the relationship between dabbling duck (hereafter duck) abundance on public wetlands and wetland habitat availability in the surrounding landscape during and post-hunting season in the GMRE. I used modeling procedures and satellite imagery developed in chapter 1 to quantify National Wetland Inventory (NWI) and WRP inundated wetland availability within 30 km of the four public wetland areas in the GMRE. The study period included autumn and winter 2004-2006 and 2008-2010, because variables used to predict NWI wetland availability and determine WRP inundation frequency were available for these time periods. Unpublished public area duck abundance estimates were available for all public wetland areas through the Missouri Department of Conservation (MDC) and United States Fish and Wildlife Service (USFWS). The MDC and USFWS conducts aerial and ground waterfowl surveys on the four public wetland areas in the GMRE from October to January 31st. Surveys were performed at least every two weeks and reported as guilds (i.e., dabbling ducks) and by species.

Dependent Variable

I designated peak duck abundance estimates on four public wetland areas in the GMRE during hunting seasons and post-hunting season for all years as the dependent variable (Table 3.1). Presence of waterfowl hunting may influences duck abundance on public wetland areas, thus I analyzed peak abundance estimates for hunting and post-hunting season separately (Bregnballe et al, 2004, Dooley et al. 2010, St. James et al. 2013, Beatty et al. 2014). Public wetland area managers conducted waterfowl surveys during and post-hunting season to assess autumn and winter waterfowl abundance fluctuations and habitat use within and among years. Survey methods were consistent within public wetland areas but can vary among wetland areas. Methods involve estimating clusters of ducks on wetland units, audible estimation of visually obscured birds, and estimates of ducks leaving and entering roost sites (Personal correspondence, C. Freeman).

Independent variables

I hypothesized that inundated wetland availability within 30 km of a public wetland area influenced duck abundance because mallard (*Anas platyrhynchos*) daily foraging flights in the region rarely exceed 30km (Beatty et al. 2013). I identified six potential independent variables within 30 km of a public wetland area influencing duck abundance; total area of NWI wetlands, number of NWI wetlands, total area of WRP wetlands, number of WRP wetlands, area of public wetland area, and ordinal date (Table 3.1). I only included NWI and WRP wetlands within the Grand and Missouri River floodplain because aerial survey estimates from chapter 2 indicate 98% of dabbling ducks occurred on wetlands within the Grand and Missouri River floodplains in the GMRE. I also included total area of each public wetland area as an independent variable because wetland area can influence dabbling duck abundance (Warnock et al. 2002, Paracuellos and Telleria 2006, Ma et al. 2009). I used models predicting NWI wetland inundation from chapter 1 to quantify area and number of NWI wetlands available to waterfowl. I applied the top ranked model to all NWI wetlands for each year of the study period and estimated inundation probability for individual wetlands. In chapter one I identified a predictive threshold of inundation probability using Receiver Operating Characteristics (ROC) curves (Fielding and Bell 1995, Pearce and Ferrier 2000). Any wetland with an inundation probability value below the predictive threshold was excluded from analysis because wetlands below this threshold were not expected to be available to dabbling ducks. I then calculated total area and number of NWI wetlands for each year of the study period. Results from Chapter 1 indicated that mean percent area and percent number of inundated WRP wetlands in the GMRE was 71% and 59% respectively. I calculated 71% of total WRP wetland area and 59% of the number of WRP wetland units within 30 km of each public wetland area and within the GMR floodplain. Finally, I hypothesized that ordinal date of peak duck abundance would be positively associated with duck abundance. I converted peak duck abundance calendar date to ordinal date beginning November 1st and ending January 31st. For example, if peak duck abundance during hunting season occurred on December 1st, the ordinal date would be 31.

Statistical analysis

I postulated that multicollinearity was present among some independent variables because number of wetlands and wetland area are likely correlated. Independent variables exhibiting multicollinearity in regression modeling can lead to highly variable parameter estimates and erroneous selection of predictor variables (Wang 1996, Kroll and Song 2003). I tested multicollinearity among independent variables using a Pearson correlation test using PROC CORR in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA). I restricted any highly correlated variables ($r > 0.5$) from being present in the same model. Finally, I removed the Public wetland area variable from candidate models because Public wetland area was highly correlated ($r \geq 0.65$) with area of NWI, number of WRP and area of WRP (Table 3.2).

I used Generalized Linear Mixed Models (GLMMs) to assess the influence of wetland habitat availability on dabbling duck abundance estimates at public wetland areas using PROC GLIMMIX in SAS 9.3 (SAS Institute, Inc., Cary, NC, USA). Generalized Linear Mixed Models are useful tools for analyzing ecological count data that is nonnormal and include random effects (Bolker et al. 2009). I included year as a random effect variable in all models to account for inter-annual variation in estimates of dabbling ducks on public areas. I used an information-theoretic model selection approach using Akaike Information Criterion adjusted for sample size (AIC_c) to compare and rank competing models (Akaike 1998, Burnham and Anderson 1998). I then calculated AIC_c weights (w_i) to quantify strength of evidence for competing models (Burnham and Anderson 2002, Bartuszevige et al. 2012).

RESULTS

Seed biomass estimates

I collected a total of 41 seed species, 25 seed species in 2014 from 240 seed head and soil core samples, and 41 seed species in 2015 from 240 seed head and soil core samples. The most frequently collected seeds included Smartweed (*Polygonum pensylvanicum* and *glabrum*; 61% of samples), Millet (*Echinochloa crus-galli*, *E. colona*, and *E. muricata*; 38% of samples), Blunt Spikerush (*Eleocharis obtusa*; 25% of samples), Panic Grass (*Panicum spp.*; 25% of samples), and Redroot Flatsedge (*Cyperus erythrorhizos*; 12% of samples; Table 3.3). The most commonly collected seed species for public wetland areas overall were Millet (63% of samples), Smartweed (56% of samples), and Blunt Spikerush (27% of samples; Table 3.3). The most commonly collected seed species for WRP wetlands overall were Smartweed (65% of samples), Blunt Spikerush (29% of samples), and Millet (18% of samples; Table 3.3). Mean number of species collected per wetland unit over both years of the study period was 2.8 (±0.07) kg/ha. Mean number of species collected during both years of the study period was 2.9 (±0.08) for public wetland areas and 2.7 (±0.12) for WRP wetland units. Cocklebur (*Xanthium strumarium*) was a frequently observed species collected in both public wetland areas and WRP units (10.8% of samples) and is not known as a beneficial food source for dabbling ducks (Fredrickson and Taylor 1982). Therefore, I excluded cocklebur seed biomass from further analysis.

Overall (public wetland area and WRP seed biomass combined over both years) mean seed mass was 6,452.1 kg/ha (± 410.8; Table 3.4). Overall seed biomass was greater in 2014 (8,137±717 kg/ha) than 2015 (4,767±371 kg/ha; F_{238}=16.02, p=0.0001; Table 3.4). Public wetland seed biomass was greater in 2014 (11,114±1195 kg/ha) than 2015 (5,755±620 kg/ha; F_{238}=16.02, p=0.0001) but I did not detect a difference between 2014 and 2015 WRP seed biomass (F_{238}=2.85, p=0.09; Table 3.4). Finally I tested for differences of seed biomass between public wetland areas and WRP wetlands for both years combined, 2014, and 2015. Combined year seed biomass was greater on public wetland areas (8,449±649 kg/ha) than WRP (4,450±402 kg/ha; F_{478}=24.81, p=0.0001; Table 3.4). In 2014, seed mass was greater on public wetland areas (11,114±1,195 kg/ha) than WRP (5,131±699 kg/ha; F_{238}=18.86, p=0.0001; Table 3.4). Finally, in 2015, seed biomass was greater on public wetland areas (5,755±620 kg/ha) than WRP (3,779±389 kg/ha; F_{238}=7.28, p=0.009; Table 3.4).

Duck abundance

Mean peak duck abundance on public wetland areas was greater during hunting season (77,870 ± 16,163 ducks) than post-hunting season (25,886 ± 16,163; Table 3.5; Figure 3.3). Grand Pass CA had the greatest duck abundance both during and post-hunting season (186,277 ± 29,951 and 77,976 ± 31205 respectively; Table 3.5; Figure 3.3). Eagle Bluffs CA had the lowest mean peak duck abundance during hunting season (16,699 ± 2,464), while Fountain Grove CA had the lowest mean peak abundance post-hunting season (7,232 ± 3,983; Table 3.5; Figure 3.3). I calculated overall mean number (n=210 ± 46) and area (7,255 ± 894 ha) of NWI wetlands (Table 3.6). Grand Pass CA had the greatest mean number of NWI wetlands (n=840) while Eagle Bluffs CA had zero predicted NWI wetlands among all years of the study period (Table 3.6). Though Swan Lake NWR had the second lowest mean number of NWI wetlands (n=120), it had the greatest mean area of NWI wetland (10,445 ± 133 ha; Table 3.6). I calculated overall mean number (90 ± 6) and area (3,705 ± 239 ha) of WRP wetlands (Table 3.6). Swan Lake NWR had the greatest mean number (n=118) and area (4,776 ha) of WRP wetlands, while Eagle Bluffs CA had the least mean number (n=48) and area (1,807 ha) of WRP wetlands (Table 3.6).

The top ranked model for predicting peak dabbling duck abundance during hunting season included number of NWI wetlands, number of WRP wetlands and ordinal date (w_i=0.99; Table 3.7). Probability of peak duck abundance during hunting season increased with an increase in number of NWI wetlands [$\beta_{(N\_NWI)}$=258.42; SE=55.5], number of WRP wetlands [$\beta_{(W\_WRP)}$=417.59; SE=420.47], and ordinal date [$\beta_{(O\_Date)}$=167.52; SE=862.47; Figure 3.4]. The top ranked model for predicting peak dabbling duck abundance post-hunting season included number of NWI wetlands, number of WRP wetlands and ordinal date (w_i=0.99; Table 3.7). Probability of peak duck abundance post-hunting season increased with an increase in number of NWI wetlands [$\beta_{(N\_NWI)}$=150.76; SE=38.23], and decreased with number of WRP wetlands [$\beta_{(W\_WRP)}$=-299.72; SE=327.38], and ordinal date [$\beta_{(O\_Date)}$=-2117.32; SE=1851.41; Figure 3.5].

DISCUSSION

A principal objective of this study was to quantify dabbling duck food abundance on public wetland areas and WRP wetlands within foraging distance of public wetland areas. Overall public wetland area seed mass estimates were unexpectedly greater than estimates reported in other studies of public wetland areas ([2,719 kg/ha] Greer 2004, [603 kg/ha] Reinecke and Hartke 2005, [496 kg/ha] Kross et al. 2008, [545 kg/ha] Evans-Peters et al. 2012) as well as WRP ([441 kg/ha] Evans-Peters et al. 2012, [528 kg/ha] Olmstead et al. 2013) and Willamette and Lower Columbia River Valleys ([545 kg/ha] Evans-Peters et al. 2012). Methods for collecting biomass estimates were similar to Reinecke and Hartke (2005) and Evans-Peters et al. (2012) in that I collected both seed head clippings and soil core samples at each sampling point. Dissimilarities between the Reinecke and Hartke (2005) and Evans-Peters et al. (2012) seed biomass estimates can likely be attributed to differences of environmental variables influencing vegetative growth and plant community composition. For example, precipitation is greatest during winter months in the Willamette and Lower Columbia River Valleys while summer is characterized as warm and dry (Evans-Peters et al. 2012). Oppositely, Missouri receives the most rainfall in spring and summer during the growth phase of wetland vegetation (Nigh and Schroeder 2002, Missouri Climate Center 2016). My methods of collecting seed biomass differed from other studies reporting substantially lower seed biomass. For example, Kross et al. (2008) and Olmstead et al. (2013) collected soil core samples but did not collect seed heads though seed heads appearing to have attached seeds were threshed. Further, Naylor (2005) used ocular methods for estimating seed biomass and did not directly measure seed mass for individual sampling points.

Overall seed biomass estimates were greater on public wetland areas than WRP wetlands (8,449 kg/ha vs. 4,450 kg/ha) as well as in 2014 (11,114 kg/ha vs. 5,131 kg/ha) and 2015 (5,755 kg/ha vs. 3,779 kg/ha). All four public wetland areas in the GMRE practice moist-soil management, a technique used to encourage early successional plant communities producing an abundance of seeds and tubers used by waterfowl in autumn and winter (Fredrickson and Taylor

1982; Fredrickson 1996; Kross 2008). The four public wetland areas enjoy a full time staff dedicated to annual maintenance of wetland units, including timed water drawdowns and soil manipulation (V. Bogosian and C. Freeman [MDC], and S. Whitson [USFWS], personal correspondence). WRP landowner interviews revealed that though they attempt annual moist-soil management, several factors inhibit continuous management such as lack of time, funds, and manpower. Further, many of the WRP units were rarely managed due to landowners living out of the area, or intense flooding during periods that moist-soil management is normally performed (i.e., spring and early summer). Although seed biomass estimates were lower on WRP wetlands, results indicate that when flooded they provide substantial amounts of food to dabbling ducks. For example, the threshold of seed mass at which dabbling ducks "give-up" and explore alternate feeding locations is documented to be 50-180 kg/ha (Reinecke 1989, Kross et al. 2008, Greer et al. 2009, Hagy and Kaminski 2015), though Hagy and Kaminski (2015) assert that other factors such as anthropogenic disturbance, predation risk, and plant species composition may be more influential to habitat selection. Still, given a WRP wetland becomes inundated in the GMRE, food availability will likely be great enough to attract waterfowl using nearby public wetland areas when other factors such as anthropogenic disturbance are not present.

Substantial research shows that hunting pressure can limit waterfowl use of wetlands in the absence of refuge areas (Davidson and Rothwell 1993, Evans and Day 2002, Madsen 2008, St. James et al. 2013). Public area peak duck abundance was greater during hunting season than post-hunting season except in 2004. It is probable that this pattern is influenced by both hunting pressure and seasonal weather trends. Though many waterfowl species utilize human dominated landscapes, waterfowl often seek out protected areas free from the presence of anthropogenic pressures (Beatty et al. 2014). Many WRP wetlands in the area were used for waterfowl hunting, which likely precluded access to waterfowl during diurnal periods. To my knowledge, none of the WRP units I sampled restricted hunting pressure, while the four public wetland areas each provided consistent refuge areas throughout the hunting season. It is possible that many of the ducks using public wetland area refuges during hunting season relocate to private wetlands after shooting hours or the conclusion of hunting season (Beatty et al 214). Moreover, it is likely that many ducks migrated out of the region post-hunting season due to decreasing ambient temperatures and likelihood of snow cover and wetland icing (Schummer 2010).

The same variables (number of NWI wetlands, number of WRP wetlands, and ordinal date) appeared in the top ranked model for both hunting season and post-hunting season abundance estimates, though parameter estimates differed. For example, all hunting season parameter estimates indicate a positive association with peak duck abundance, while number of WRP wetlands and ordinal date were negatively associated with peak duck abundance post-hunting season. It is not surprising that number of NWI and WRP wetlands is positively associated with peak duck abundance on public wetland areas during hunting season. Indeed, a plethora of research indicates that availability of wetlands in the surrounding landscape can positively influence duck abundance and distribution (Brown and Dinsmore 1986, Soulé and Simberloff 1986). Further, many investigations into waterbird use of wetland complexes indicate that number of wetlands can be equally or more influential than total area of a wetland (Brown and Dinsmore 1986, Soulé and Simberloff 1986, Ma et al. 2009). For example, a group of small wetlands are more likely to exhibit habitat heterogeneity supporting a greater range of life functions than a single large wetland (Soulé and Simberloff 1986). Interestingly, little research has evaluated the importance of wetland complexes using the estimated number and area of inundated wetlands based on seasonal environmental factors (Davis et al 2014). My results highlight the utility of using probability of wetland inundation and estimated number and area of wetlands in modeling waterfowl habitat use and selection.

It is likely that many migrating ducks pass over the GMRE entirely when wetland availability is low because migratory waterbirds may select migratory habitat based on habitat availability (Skagen and Knopf 1993, Ma et al. 2010, Davis et al. 2014). Ordinal date was positively associated with peak duck abundance during hunting season and this relationship can likely be attributed to wetland habitat availability outside hunting areas. Years with fewer NWI and WRP wetlands tend to have earlier peak duck abundance ordinal dates and lower peak abundance on public wetlands during hunting season. Many migrating ducks likely leave the GMRE earlier when wetland resources are restricted due to decreased food availability and refuge from hunting pressure. A later ordinal date of peak duck abundance would indicate that autumn migrating ducks stopover in the GMRE, likely because of favorable habitat conditions, and continue to increase in abundance as the waterfowl migration continues. Conversely, ordinal date was negatively associated with peak duck abundance post-hunting season, with mean peak duck abundance substantially lower post-hunting season than during hunting season (77,870 ± 16,163 vs. 25,886 ± 9,761). In Missouri, January is on average the coldest month of the annual cycle and many wetland habitats become unavailable to dabbling ducks because of icing over (Schummer 2010, Missouri Climate Center 2016). Indeed, increasing ordinal date post-hunting season increases probability of energy expenditure and wetland icing over due to seasonally low ambient temperatures (Schummer 2010). Consequentially, peak duck abundance post-hunting season is likely not a peak in fall and winter migration, but simply a symptom of either ducks migrating further south due to decreasing wetland habitat availability, or moving onto nearby private wetlands in the absence of hunting pressure (Beatty et al. 2014).

MANAGEMENT IMPLICATIONS

My results indicate that wetland availability on private lands (i.e., NWI and WRP wetlands), likely influences dabbling duck stopover habitat selection during the fall migration (Ma et al. 2009, Beatty et al. 2014, Davis et al 2014, Kaminski and Elmberg 2014). For example, it appears that dabbling duck abundance on public wetlands is positively influenced by wetland habitat availability on the local landscape outside public wetland areas. Certainly, duck abundance and ordinal date of peak abundance on public wetland areas was greater during years with more available wetland habitat outside public wetland areas. In other words, migratory ducks are more likely to use public wetland areas and stay at those areas longer when additional wetland habitat is available within a 30 km foraging distance (Beatty et al. 2013). My results highlight the importance for both wetland conservation on public and private lands, especially where currently present wetlands are highly isolated such as Eagle Bluffs CA. I recommend further investigation into reestablishing and increasing factors influencing NWI wetland inundation. Further I recommend focusing WRP restoration efforts in areas likely to complement public wetland areas and other wetlands on private lands.

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Table 3.1. Variables used to develop models explaining waterfowl abundance on public managed wetland areas in the Grand and Missouri River Ecoregion, Missouri during autumn and winter 2004-2006, and 2008-2010.

Variable Identifier	Variable Description	Mean
Dependent Variable		
HS	Duck abundance on public wetland area during hunting season	77,870 ± 16,163
AS	Duck abundance on public wetland area post-hunting season	25,886 ± 9,761
Independent Variable		
NWI_Area	Total area (ha) of NWI wetlands within 30km of a public wetland area	210 ± 46
NWI_Number	Total number of NWI wetlands within 30km of a public wetland area	7,225 ± 894
WRP_Area	Total area (ha) of Wetland Reserve Program (WRP) wetlands within 30km of a public wetland area	90 ± 6
NWI_Number	Total number of WRP wetlands within 30km of a public wetland area	3,705 ± 239
Public Area	Total area (ha) of each public wetland area	10,593
HS_ODate	Ordinal date of peak duck abundance during hunting season	30 ± 2.7[1]
AS_ODate	Ordinal date of peak duck abundance after hunting season	67 ± 0.86[2]

[1]Ordinal date of peak duck abundance on a public wetland area during hunting season

[2]Ordinal date of peak duck abundance on a public wetland area post-hunting season

Table 3.2. Pearson correlation test identifying highly correlated predictor variables used to develop models explaining waterfowl abundance on public managed wetland areas in the Grand and Missouri River Ecoregion, Missouri during autumn and winter 2004-2006, and 2008-2010.

	HS_Odate	AS_ODate	N_NWI	A_NWI	N_WRP	A_WRP	A_Public
HS_Odate	1.00000	0.19655	0.46826	0.02318	0.12565	0.15483	-0.26162
AS_ODate	0.19655	1.00000	0.13392	0.13105	0.17673	0.17364	0.06563
N_NWI	0.46826	0.13392	1.00000	0.36351	0.47007	0.53854	-0.27349
A_NWI	0.02318	0.13105	0.36351	1.00000	0.92656	0.93181	0.78349
N_WRP	0.12565	0.17673	0.47007	0.92656	1.00000	0.99581	0.69734
A_WRP	0.15483	0.17364	0.53854	0.93181	0.99581	1.00000	0.64962
A_Public	-0.26162	0.06563	-0.27349	0.78349	0.69734	0.64962	1.00000

[a] Refer to table 3.1 for variable identified codes

Table 3.3. Frequency (percent samples a species was detected) of plant species collected from seed head clippings and soil core samples after plants had gone dormant in October. Samples were collected in the Grand and Missouri River Ecoregion, Missouri to evaluate potential waterfowl food abundance.

Species	Overall[1]	Public Wetland Area		WRP	
		2014	2015	2014	2015
Polygonum spp.	60.8%	88.3%	55.8%	34.2%	65%
Echinochloa spp.	38.1%	47.5%	63.3%	24.2%	17.5%
Eleocharis obtusa	24.8%	26.7%	26.7%	17.5%	28.3%
Panicum spp.	24.6%	45.8%	19.2%	19.2%	14.2%
Cyperus erythrorhizos	11.9%	11.7%	17.5%	4.2%	14.2%
Xanthium strumarium	10.8%	11.7%	16.7%	≤1.0%	14.2%
Leersia oryzoides	10.0%	8.3%	10.8%	6.7%	14.2%
Setaria spp.	9.2%	13.3%	≤1.0%	10.0%	13.2%
Amaranthus spp.	9.0%	13.3%	9.2%	≤1.0%	12.5%
Bidens aristosa	6.7%	≤1.0%	16.7%	≤1.0%	10.0%
Iva annua	5.6%	≤1.0%	≤1.0%	6.7%	15.0%
Spartina pectinate	2.3%	≤1.0%	7.5%	≤1.0%	≤1.0%
Ammannia auriculata	1.7%	3.3%	≤1.0%	2.5%	≤1.0%
Leptochloa fascicularis	1.7%	3.3%	≤1.0%	≤1.0%	2.5%

[1]Mean percent species collected from all samples for public wetland areas and WRP units 2014-2015.

Table 3.4. Number of sites sampled, number of samples collected, and mean (± SE) total seed biomass (kg/ha) collected from public wetland area and Wetland Reserve Program (WRP) wetlands in the Grand and Missouri River Ecoregion, Missouri. Combined seed head clippings and soil core samples were collected in October 2014 and 2015 to estimate total seed biomass.

Variable	*n* sites	*n* samples	Total seed biomass[1]	
			$\bar{x}$	SE
Year				
2014	24	240	8,137.2	717.7
2015	24	240	4,767.0	371.0
Combined[2]	48	480	6,452.1	410.8
Public wetland area				
2014	12	120	11,114	1,195.2
2015	12	120	5,754.7	620.4
Combined[3]	24	240	8,449.2	694.1
WRP				
2014	12	120	5,130.9	699.1
2015	12	120	3,779.2	389.4
Combined[4]	24	240	4,450	401.6

[1] Total seed biomass does not include cocklebur (*Xanthium strumarium*) because they are unlikely to be consumed by dabbling ducks and would greatly inflate biomass estimates.

[2] All samples (public wetland area units and WRP units) combined over both years of the study period (2014 and 2015)

[3] Public wetland area samples combined over both years of the study period (2014 and 2015)

[4] WRP samples combined over both years of the study period (2014 and 2015)

Table 3.5. Mean and standard error (SE) peak public wetland area dabbling duck abundance estimates in the Grand and Missouri River Ecoregion, Missouri during hunting season (HS) and post-hunting season (PS). Weekly ground count estimates were collected by public wetland area managers from the onset of autumn migration in October to January 31st 2004-2016 and 2008-2010.

Public Wetland Area	HS duck abundance		PS duck abundance	
	$\bar{x}$	SE	$\bar{x}$	SE
Eagle Bluffs	16,699	2,464	11,916	4,072
Fountain Grove	25,089	4,120	7,232	3,983
Grand Pass	186,277	29,951	77,976	31,205
Swan Lake	83,415	14,986	6,420	4,458
Combined[1]	77,870	16,163	25,886	9,761

[1]Peak public wetland area dabbling duck abundance estimates were pooled among public wetland areas

Table 3.6. Mean and Standard error estimates of number of National Wetland Inventory (NWI), NWI area, number of Wetland Reserve Program (WRP), and WRP area within 30 km of a public wetland area in the Grand and Missouri River Ecoregion, Missouri. All NWI and WRP data was collected from the Grand and Missouri River floodplains because regional aerial survey estimates indicate >90% of dabbling ducks occur within the floodplains in autumn and winter. NWI estimates were based on probability of inundation from modeling in chapter 1 and WRP estimates were collected from the Natural Resource Conservation Service.

Public Wetland Area	# NWI	Area NWI	# WRP	Area WRP
Eagle Bluffs	0	0	48	1,807
Fountain Grove	134 ± 8	10,325 ± 223	92	3,845
Grand Pass	840 ± 0	8,251 ± 0	103	4,395
Swan Lake	120 ± 6	10,445 ± 133	118	4,776

Table 3.7. Top three ranked models explaining peak dabbling duck abundance estimates on public wetland areas during hunting season (DS) and post-hunting season (PS) in the Grand and Missouri River Ecoregion, Missouri. Models were ranked using Akaike Information Criterion for small sample size (AIC_c) for the effects of variables on probability of peak duck abundance (model terms: N_NWI = estimated number of available National Wetland Inventory (NWI) wetlands, A_NWI = estimated area of NWI wetlands available, N_WRP = estimated number of Wetland Reserve Program wetlands available, and O_Date = ordinal date of peak waterfowl abundance on public wetland areas.

Season	Model	K	AIC_c	ΔAIC_c	w_i
DS	N_NWI, N_WRP, O_Date	4	524.14	0	0.99
	N-NWI, A_NWI, O_Date	4	535.39	11.25	0.01
	N_NWI, O_Date	3	539.04	14.9	0.00
PS	N_NWI, N_WRP, O_Date	4	515.53	0	0.99
	N-NWI, A_NWI, O_Date	4	525.45	9.92	0.01
	N_NWI, O_Date	3	529.75	14.22	0.00

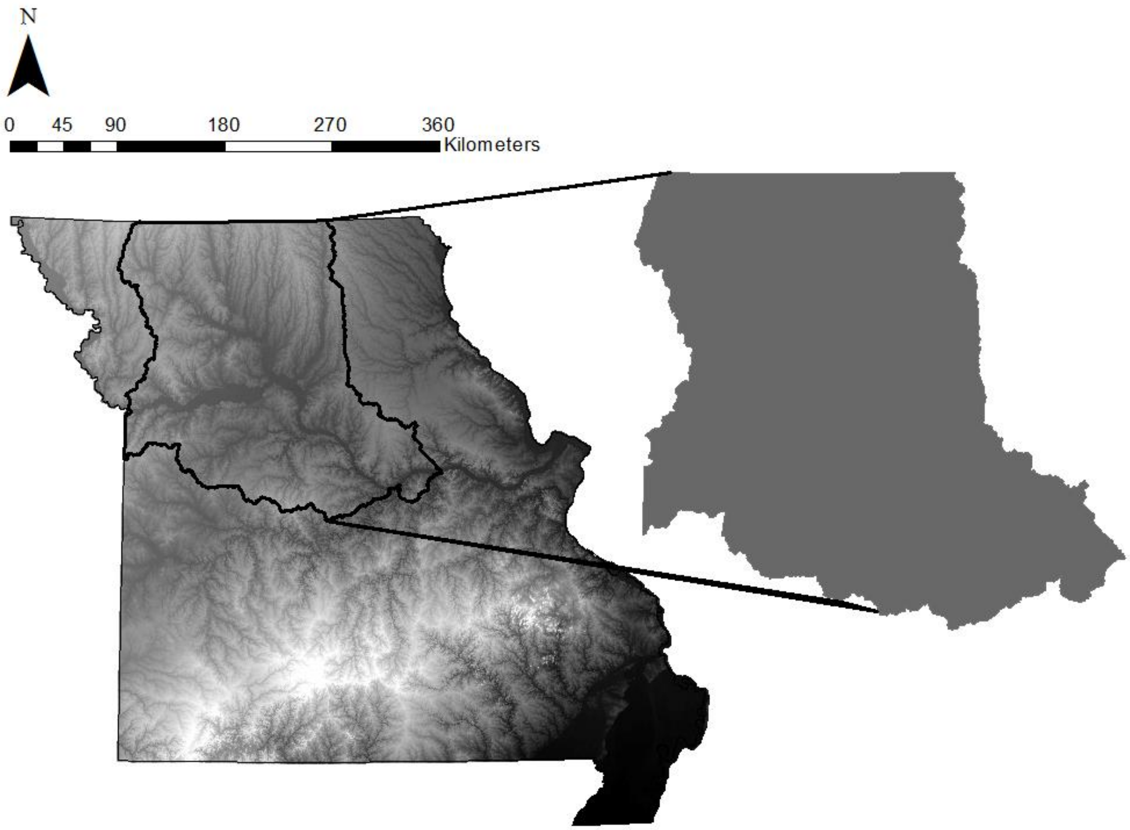


Figure 3.1. The Grand and Missouri River Ecoregion in North-central Missouri where I collected public wetland area and Wetland Reserve Program seed head and soil core samples to estimate duck food abundance in autumn. Further, I modeled factors potentially influencing peak duck abundance on public wetland areas during and post-hunting season 2004-2006 and 2008-2010.

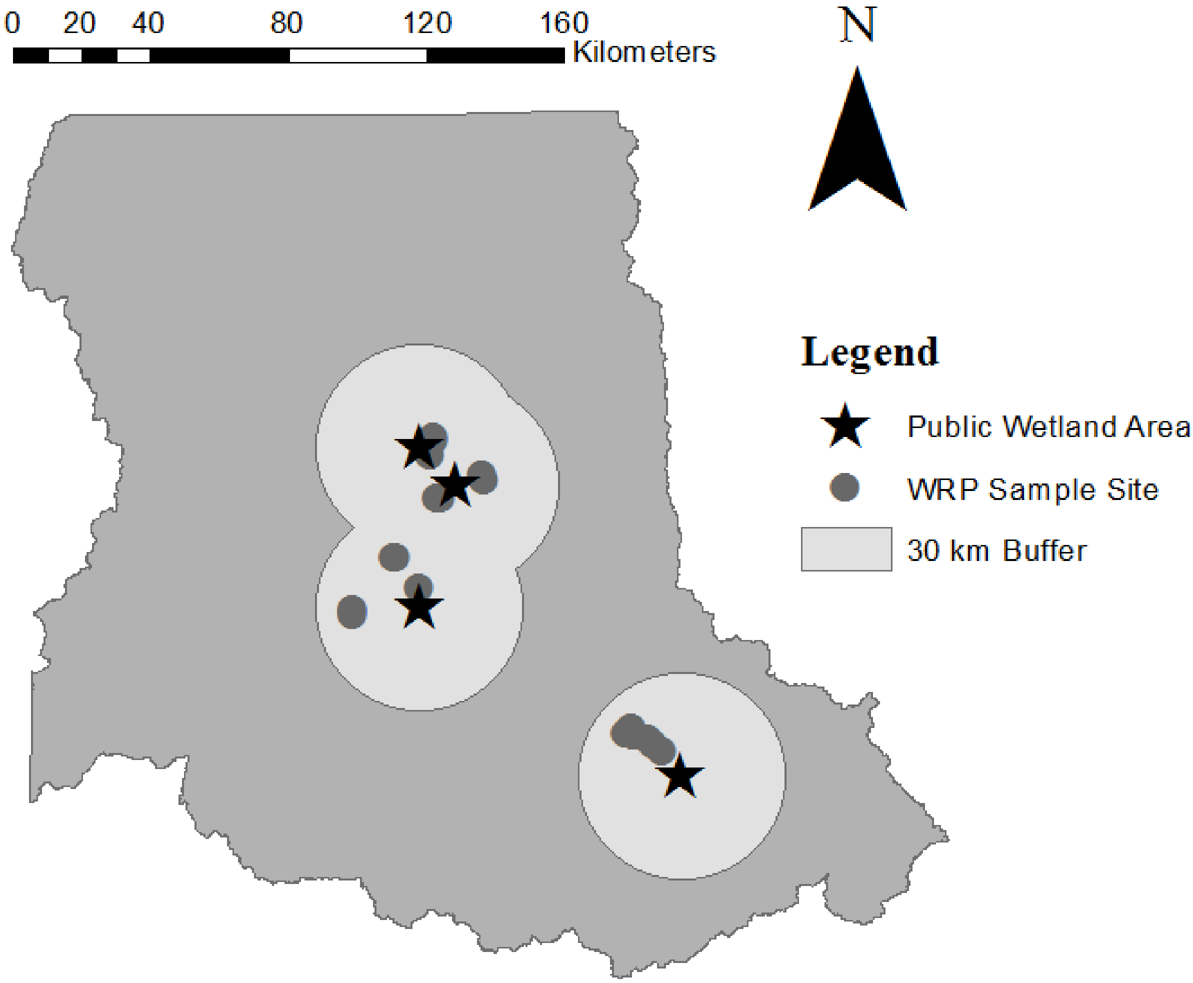


Figure 3.2. The Grand and Missouri River Ecoregion in North-central Missouri where I collected public wetland area and Wetland Reserve Program (WRP) seed head and soil core samples to estimate duck food abundance in autumn. All WRP samples were collected within 30 km of a public wetland area.

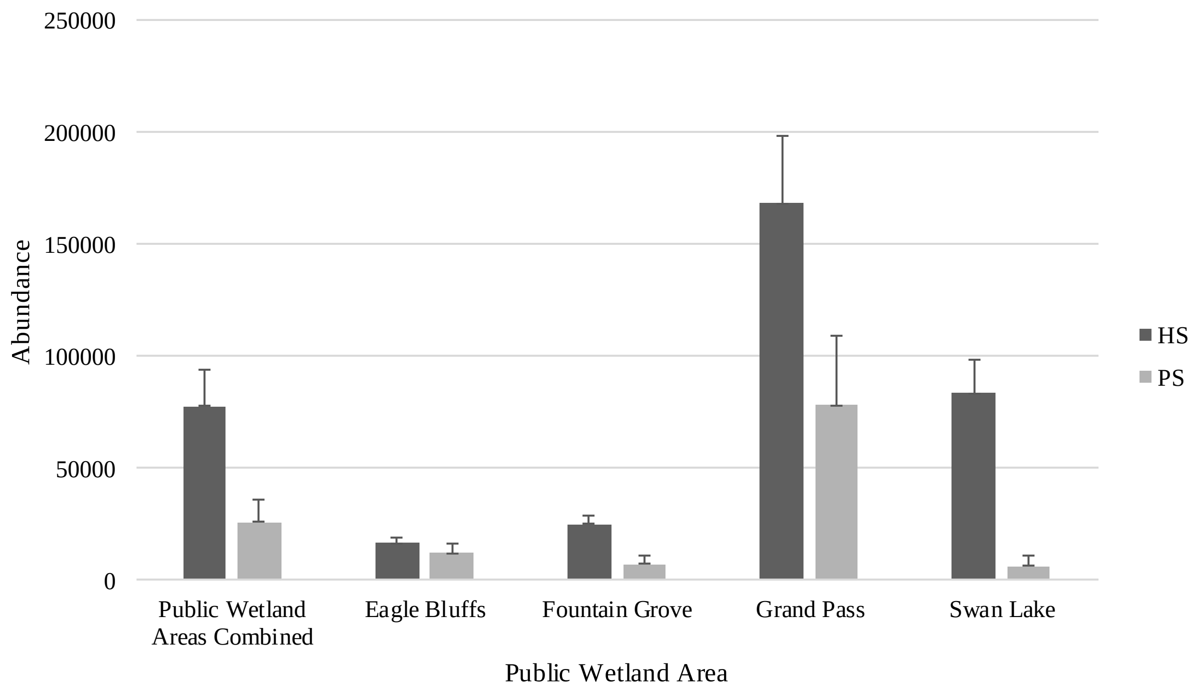


Figure 3.3. Mean and standard error (SE) peak public wetland area dabbling duck abundance estimates in the Grand and Missouri River Ecoregion, Missouri during hunting season (HS) and post-hunting season (PS). Weekly ground count estimates were collected by public wetland area managers from the onset of autumn migration in October to January 31st 2004-2010.

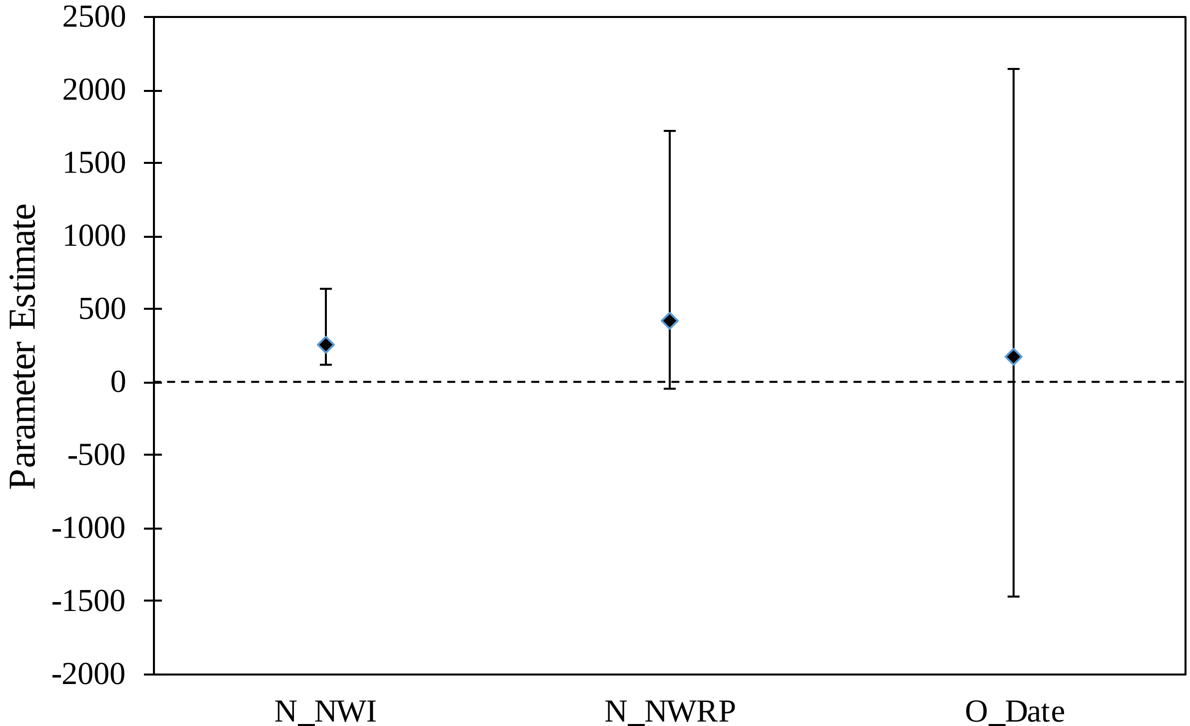


Figure 3.4. Model parameter estimates and 95% confidence intervals for Generalized Linear Mixed Models predicting peak waterfowl abundance during hunting season in the Grand and Missouri River Ecoregion, Missouri. N_NWI is number of National Wetland Inventory (NWI) wetlands within 30 km of a public wetland area, N_WRP is number of Wetland Reserve Program wetlands within 30 km of a public wetland area, and O-Date is ordinal date of peak waterfowl abundance on public wetland areas.

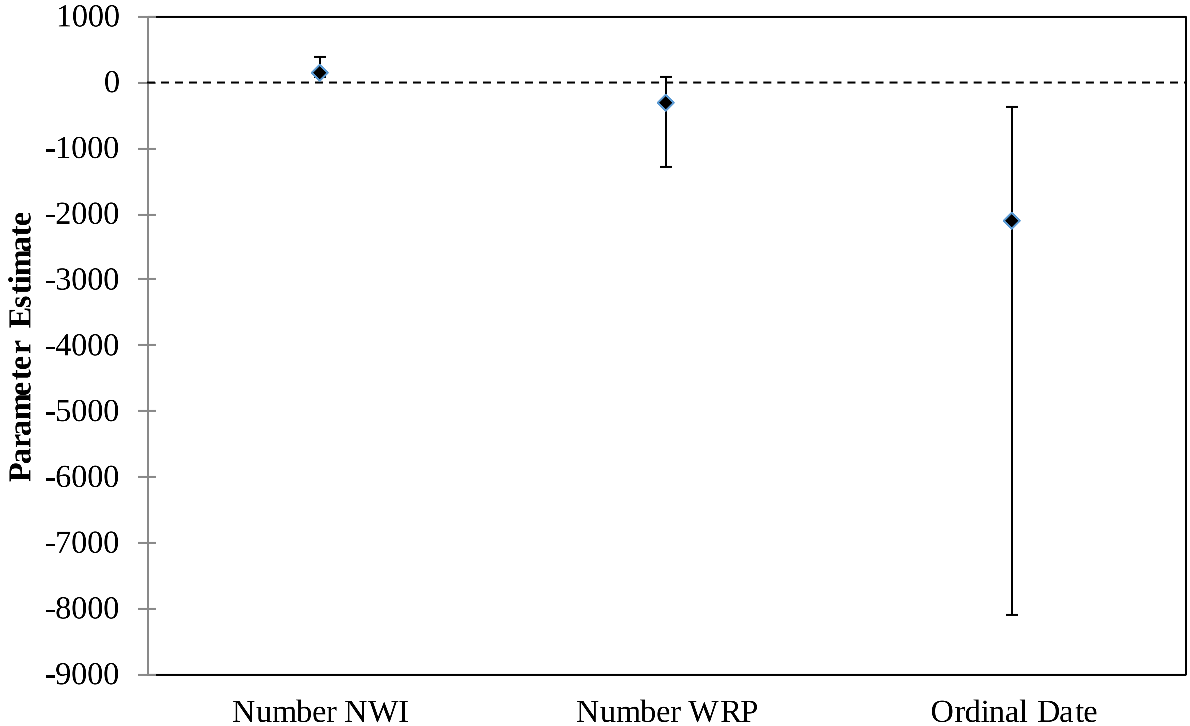
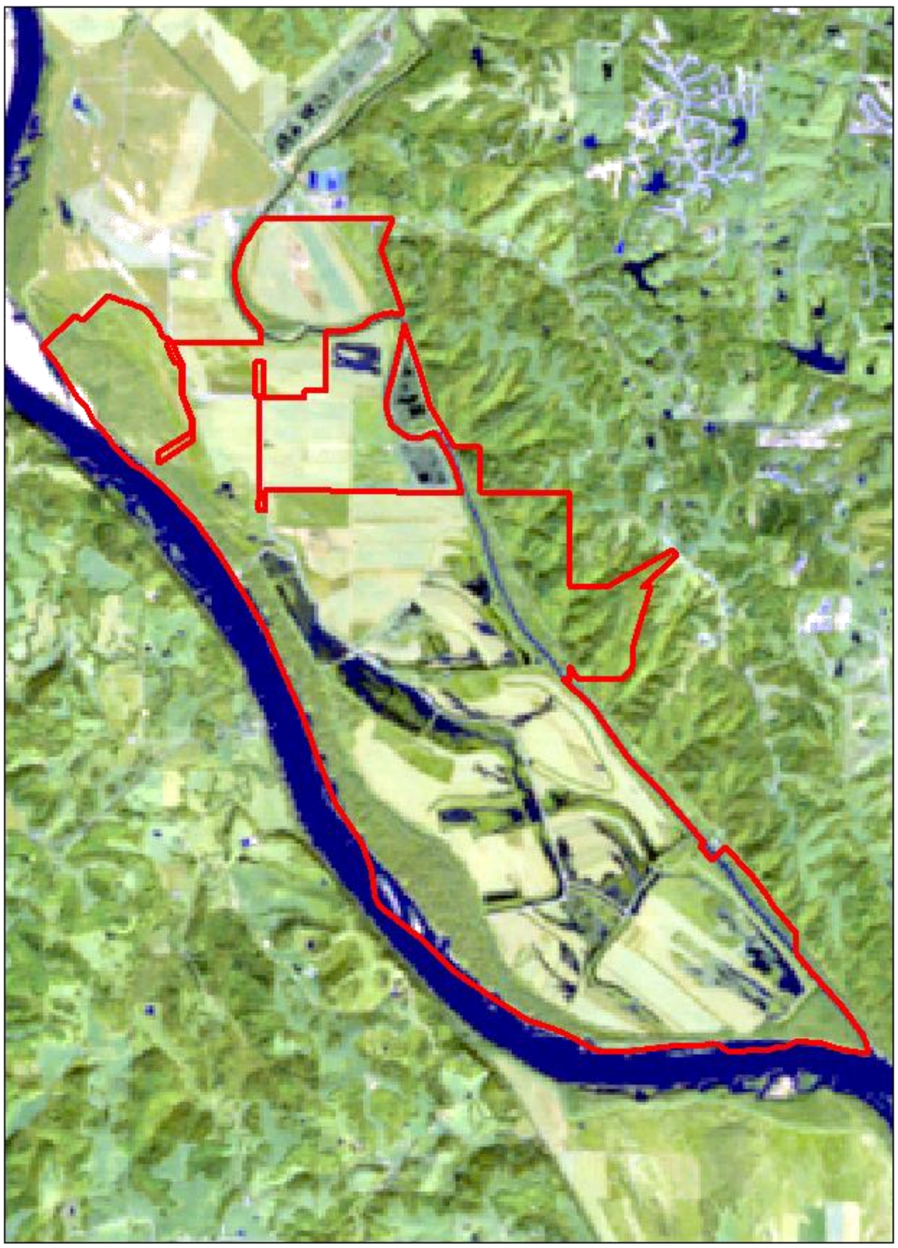


Figure 3.5. Model parameter estimates and 95% confidence intervals for Generalized Linear Mixed Models predicting peak waterfowl abundance post-hunting season in the Grand and Missouri River Ecoregion, Missouri. N_NWI is number of National Wetland Inventory (NWI) wetlands within 30 km of a public wetland area, N_WRP is number of Wetland Reserve Program wetlands within 30 km of a public wetland area, and O-Date is ordinal date of peak waterfowl abundance on public wetland areas.

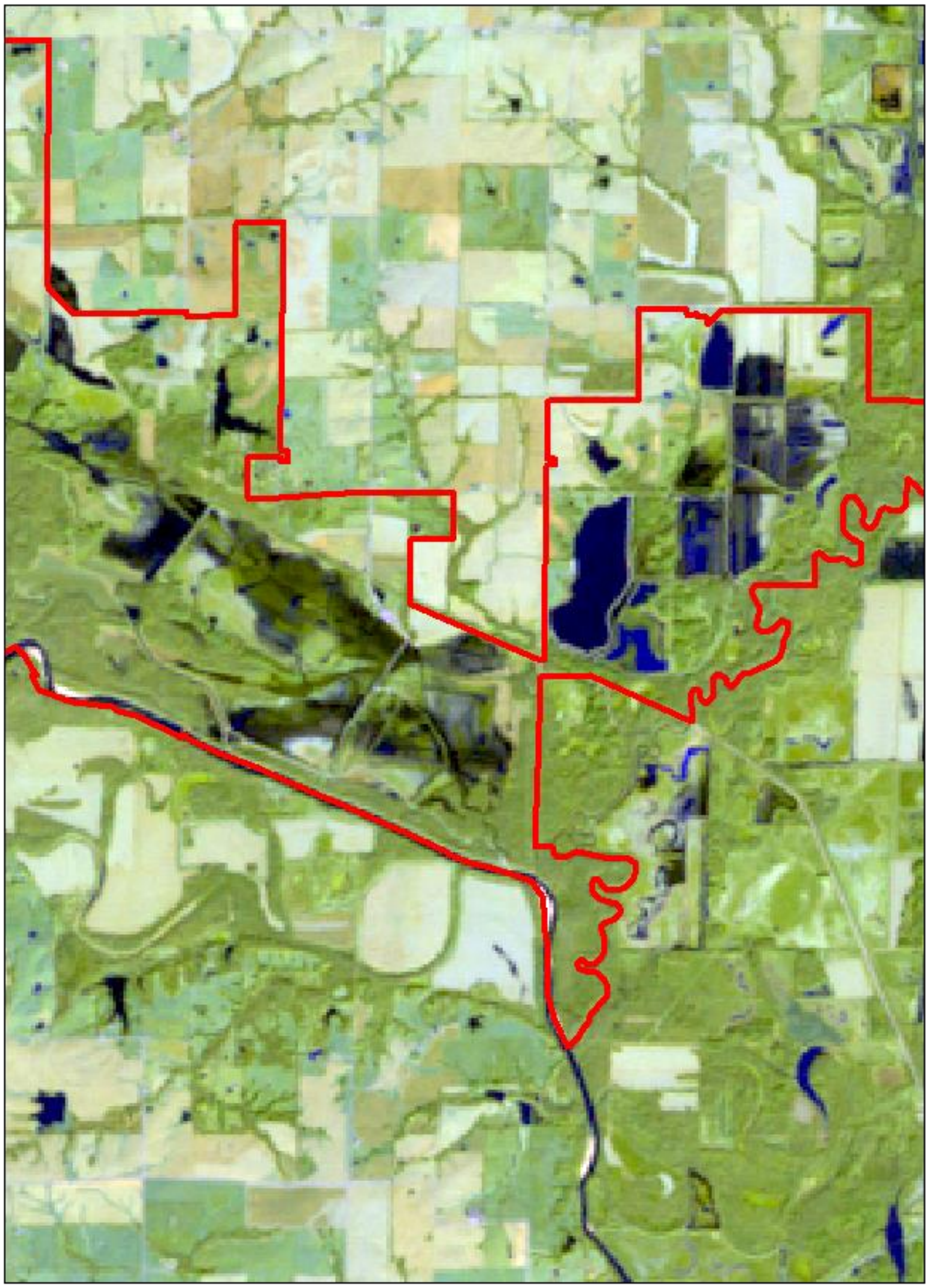
APPENDIX A

Landsat 5 Thematic Mapper satellite images from November 2004-2006 and 2007-2008 were used to identify inundated wetlands. Missouri Department of Conservation (MDC) and U.S. Fish and Wildlife Service (USFWS) public wetland areas were used as training sites to perform a supervised classification in Earth Resources Data Analysis System Imagine. I identified known land cover pixels (i.e., inundated wetland, cropland, and forest) and created specific land classes that could be converted to a raster dataset to determine National Wetland Inventory wetland basin inundation.

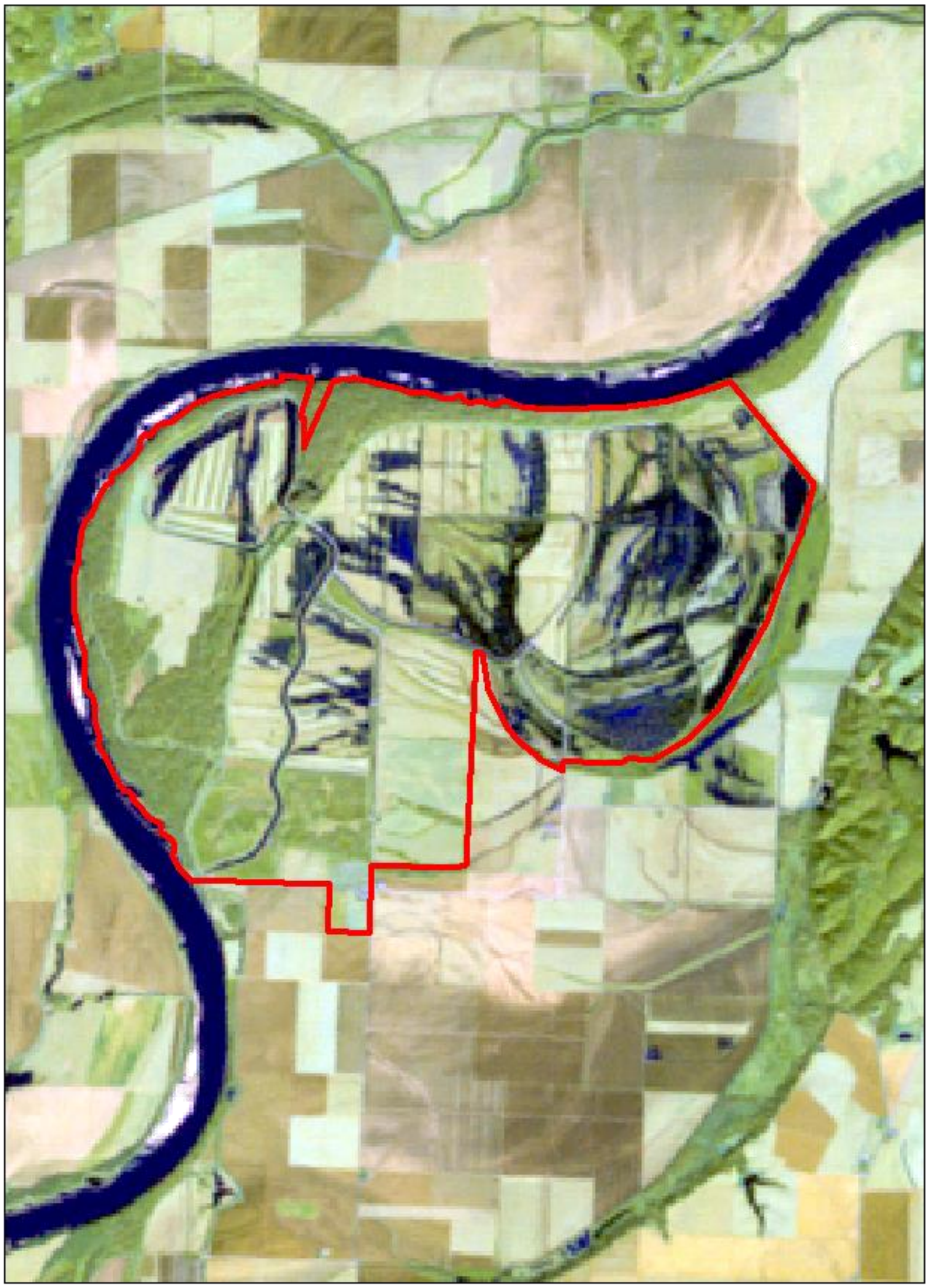
Eagle Bluffs CA



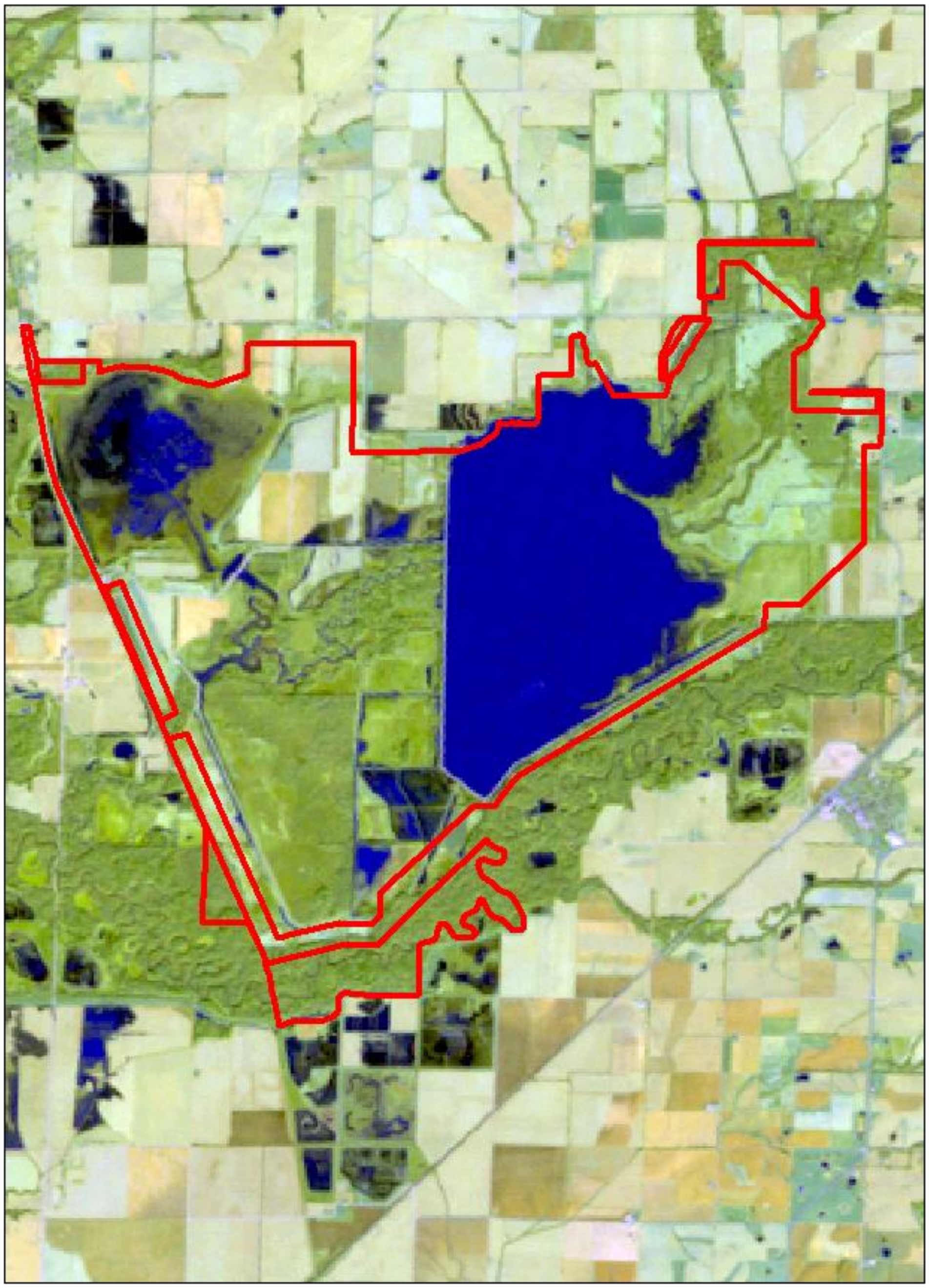
Fountain Grove CA



Grand Pass CA



Swan Lake NWR



APPENDIX B

Corrected waterfowl observations from an aerial strip-transect waterfowl survey flown in the Grand and Missouri River Ecoregion in north Missouri autumn and winter 2014 and 2015-2016. Observation were corrected for observer bias using a two sample capture-recapture Petersen estimator.

Survey Date	Ducks	CAGO[1]	TRUS[2]	Total
2014				
11 November	409	95	0	504
9 December	1,250	200	12	1,462
23 December	855	705	102	1,662
2015-2016				
2 November	378	84	0	462
20 November	757	49	12	806
4 December	2,203	387	7	2,590
30 December	260	202	0	461
11 January 2016	5,004	1036	15	6,040
26 January 2016	1,757	0	94	1,757

[1]Canada Goose

[2]Trumpeter Swan