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Informing Flood Plain Wetland Restoration Using Amphibian Monitoring

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**INFORMING FLOOD PLAIN WETLAND RESTORATION USING
AMPHIBIAN MONITORING**

by

Ashley E. VanderHam

A THESIS

Presented to the Faculty of

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INFORMING FLOOD PLAIN WETLAND RESTORATION USING AMPHIBIAN MONITORING

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Advisor: Craig Allen

Wetlands are among the most important and complex ecosystems in the world. They contribute to nutrient cycling, the hydrologic cycle, and provide critical habitat for many plants, fish, and wildlife. Channelization of Missouri River resulted in the loss of many floodplain wetlands. Despite ongoing restoration efforts, there are few ecologically-based performance guidelines, and managers need methods to quantify and assess the success of restored riverine wetland systems.

In 2008 a multi-institutional herpetofauna monitoring project, funded by the U.S. Army Corps of Engineers, was initiated in four states (Kansas, Missouri, Iowa, and Nebraska). The main goal of the project is to assess the success of previously restored wetlands and to create wetland restoration guidelines for future use. Amphibians were chosen for monitoring because they are globally declining, they integrate terrestrial and aquatic environments, and because they are good indicators of wetland restoration success. Frog call surveys and tadpole dip net surveys were conducted and analyzed using occupancy techniques to help determine restoration success. This thesis reports the results of herpetofauna monitoring from three Missouri River bends in Nebraska, a subset of the overall project.

I conducted a comparison of frog call surveys and tadpole dip net occupancy results, a novel co-occurrence analysis for frog call surveys, and a functional connectivity analysis based on anuran dispersal distances. The results of the frog call surveys and tadpole dip net surveys differed in many ways. The data from the tadpole dip net surveys produced fewer results, but these results were more accurate because of their link to reproduction and the lack of spurious data. If only one method is used it should be

the tadpole dip net surveys, but if the surveys are conducted in a drought year it is possible there will be insufficient data to conduct an occupancy analysis. Therefore, conducting both methods is the best way to produce the most accurate wetland restoration guidelines. The co-occurring species analysis of the frog call surveys was both more efficient and also produced a definition of wetland success for multiple species without the variation between species seen in the single species models. The functional connectivity analysis added a complex systems component to the wetland restoration guidelines, but also provided a method that can help focus management by identifying wetlands within and across complexes that are most important to functional connectivity. Combining the information gained from all of the analysis, it was determined that a successful flood plains wetland has aquatic vegetation, a shallow slope (less than 0.30), is ephemeral, has at least one wetland within 500 m, and is part of a compact (non-linear) wetland complex that has other successful wetlands and deeper, larger, less ideal wetlands that are all close enough to one another that habitat is provided in extreme conditions.

DEDICATION

I want to dedicate this book to my Mom and Dad for allowing me to be the idealistic, adventurous, dreamer that I am. Despite how hard it must have been, you let me take my own winding path. I would have never made it here without your love and support. Because of you I have a career that I love and believe in, built on values that run in my very blood. Grandma Sweetie Pie, without your support I would have never gotten a good night's sleep (real beds rule!), or had a fabulous live grandparent to brag about. I hope I have made you proud. Jerod, your kind, giving heart has not stopped surprising me. Your love and belief in me has helped me find my drive at my lowest, plus I think I would have drowned in dishes and laundry without you. While you may have not understood my situation, you quickly forgave and never judged my graduate student craziness. To my little loves, Lady Day and Loki, my sanity is still intact because of your fluffy support. You both constantly remind me of what is truly important, that the world does not revolve around my thesis, it revolves around you.

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INTRODUCTION

Wetlands are among the most important and complex ecosystems in the world. Although they cover less than three percent of the earth's surface, they contribute up to 40% of the global annual renewable ecosystem services (Zedler and Kercher 2005). They contribute to nutrient cycling, the hydrologic cycle and provide critical habitat for many plants, fish, and wildlife species (Mitsch and Gosselink 2007). In riverine ecosystems, wetlands aid in creating and maintaining nutrient rich soils. They also help purify water by acting as natural filters and sinks. Additionally, wetlands may contribute to climate stabilization through carbon sequestration (Zedler and Kercher, 2005).

In the Midwestern United States much of this critical habitat has been altered or destroyed (Kryzick 1998 and Leja 1998). For the wetlands in the Missouri River flood plain the main cause of loss and alteration resulted from the channelization of the Missouri River. The Bank Stabilization and Navigation Project, implemented in 1912, initiated channelization or the deepening, widening, and straightening of the Missouri River. These alterations lead to the loss of over 500,000 acres of wetland and bottomland habitat (U.S. Army Corp of Engineers Kansas City and Omaha Districts 2003).

The importance of the Missouri River for navigation has been steadily decreasing since 1977, especially above Kansas City (MRRP 2009). Concomitant with this decrease has been an increase in the recognition of the importance of wetland and wetland habitats (Mitsch and Gosselink 2007, Moll and Moll 2004, Zedler and Kercher 2005). Wetlands provide a broad range of ecosystem services, including regulating the hydrological cycle, contributing to nutrient cycling, and acting as refuge for many fish and wildlife species. In addition to those ecosystem services, wetlands and undiked floodplains can greatly reduce flooding and flood impacts (Bakacsi et al. 2011). Climate variability is expected to increase in the relatively near future, and this variability is expected to include more extreme weather events (IPCC 2003). The recognition of the importance of wetlands has guided U.S. government policy, and led to legislation such as the Clean Water Act of 1986 and the Water Resources Development Act of 1986.

Following the enactment of the Clean Water Act of 1986 the U.S. Army Corp of Engineers (USACE) instated a policy of “no net loss.” Between 1996 and 2005 the United States experienced an estimated net gain of 10,000 ha/yr of wetland and associated uplands habitat. However, it is unclear how successful these created, restored, enhanced, or preserved wetlands are: There is no information on what ecosystem functions were lost and what functions were gained (Mitsch and Gosselink 2007). Although restoration techniques continue to improve, there are still no overall guidelines for building a functional wetland outside of engineered hydrological changes. Because of ongoing restoration efforts, increasing threats to biodiversity and to wetlands, and lack of ecologically-based performance guidelines, there needs to be concerted and quantifiable efforts to assess the success of restored riverine wetland systems, so that general requirements can be put into effect. The Water Resources Development Act of 1986 led to the creation of the Missouri River Fish and Wildlife Mitigation Project (MRFWMP). The purpose of the MRFWMP is to protect the remaining critical habitat on the Missouri River and restore some of which was lost. The MRFWMP monitoring effort was expanded in 2007 to include reptiles and amphibians. In 2008 a multi-institutional project funded by the USACE was initiated in four states (Kansas, Missouri, Iowa, and Nebraska). The main goal of the overall project is to assess the success of previously restored wetlands and to create wetland restoration guidelines for future use.

To determine restoration success, herpetofauna monitoring was established on previously restored wetlands. Frog call surveys and tadpole dip net surveys were conducted and analyzed using occupancy techniques (McKenzie *et al.* 2006) to help determine restoration success. Turtle trapping was also conducted, but there was not enough data collected to contribute to the assessment of wetland function. Amphibians were chosen for monitoring because they are globally declining (Glascon *et al.* 2005; Pentranka and Holbrook 2006; Collins and Crump 2009; Crump 2010), they integrate terrestrial and aquatic environments (Collins and Crump 2009; Vitt and Caldwell 2009; Collins and Halliday 2005) and because they are good indicators of wetland restoration success (Bowers *et al.* 1998; Collins and Halliday 2005; Welsh and Ollivier 1998). Specifically, their biphasic life cycle and permeable skin make them

uniquely sensitive to conditions in the water and the surrounding terrestrial environment. Wetlands are complex ecosystems that are hard to define and consist of both aquatic and terrestrial components. They are easily altered or degraded by changes in either habitat component. Thus, amphibians, whose biphasic life cycle and permeable skin makes them dependent on these intricate and essential ecosystems, are ideal indicators of wetland restoration success. This thesis reports the results of herpetofauna monitoring from three Missouri River bends in Nebraska, a subset of the overall project. The thesis is organized around four core chapters, which focus on occupancy of wetlands by amphibians and wetland connectivity.

Chapter 2 focuses on the factors affecting adult amphibian detection and occupancy. Using frog call surveys and single species occupancy models, the environmental variables affecting both the ability to detect the adult calls and their occupancy of the wetlands were determined. The factors determining occupancy were used to define what a successful wetland restoration is, and I then provide suggestions for future restoration derived from that definition.

Chapter 3 determines the factors affecting larval amphibian detection and occupancy, and then compares this with the adult amphibian results reported in the previous chapter. I identify both the similarities and differences between the factors determining adult and larval amphibian occupancy, and make suggestions resulting from these contrasts that may help better define wetland success.

Chapter 4 uses a novel co-occurrence analysis to inform wetland restoration. As in Chapter 2, this Chapter focuses on the factors affecting adult amphibian detection and occupancy, but it uses the occupancy of different species combinations as opposed to the occupancy of single species. This chapter attempts to create a more accurate definition of wetland success by addressing the occupancy of multiple species that co-occur. This follows from the idea that a successful wetland will have the environmental factors that support a diverse amphibian population. Although individual species are important, restored wetlands should provide habitat for a diversity of species. Thus, this method may provide a more effective and robust method to determine the factors that are most important to amphibian occupancy.

Chapter 5 uses functional connectivity based on anuran dispersal distances to further inform wetland restoration. This chapter adds a complex systems component to the wetland restoration guidelines. This analysis provides a method to focus management by identifying wetlands within and across complexes that are most important to functional connectivity. This approach demonstrates a method that can determine functional connectivity of future wetlands, and my comparison of connectivity across wetland complexes can help guide restorations in the future. As our knowledge of wetlands grows it is becoming clearer that restoration is likely to be more successful at a complex, landscape, or watershed level (Zedler and Kercher, 2005). Although the restoration details of individual wetlands are important, using the landscape and hydrological connectivity to create dynamic and functionally connected wetland complexes is most critical.

The chapters are linked around the theme of assessing functional success of wetland restorations. It is hoped that recommendations will be incorporated into USACE guidelines, and that these guidelines continue to be refined through a process of adaptive management.

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CHAPTER 2: FACTORS AFFECTING ADULT AMPHIBIAN DETECTION AND OCCUPANCY ON THE MISSOURI RIVER FLOOD PLAIN

INTRODUCTION

Wetlands play a significant role in ecosystem functions. They contribute to nutrient cycling, are essential to the hydrological cycle, and provide habitat for many plants, fish and wildlife species (Mitsch and Gosselink 2007). Much of this critical habitat has been converted through drainage, stabilization and channelization activities (Kryzick 1998 and Leja 1998). Along the Missouri River the main cause for wetland habitat loss was the Bank Stabilization and Navigation Project of 1912. It was responsible for the channelization of the Missouri River and directly and indirectly led to the loss of over 500,000 acres of Missouri River habitat (U.S. Army Corp of Engineers Kansas City and Omaha Districts 2003).

The importance of the Missouri River for navigation has been steadily decreasing since 1977, especially above Kansas City (MRRP pamphlet 2009). Concomitant with this decrease has been an increase in the recognition of the importance of wetland and wetland habitats (Mitsch and Gosselink 2007, Moll and Moll 2004, Zedler and Kercher, 2005). Wetlands provide a broad range of ecosystem services, including regulating the hydrological cycle, contributing to nutrient cycling, and acting as refuge for many fish and wildlife species. In addition to those ecosystem services, wetlands and undiked floodplains can greatly reduce flooding and flood impacts (Bakacsi et al. 2011). Climate variability is expected to increase in the relatively near future, and this variability is expected to include more extreme weather events (IPCC 2003). The recognition of the importance of wetlands has guided U.S. government policy, and led to legislation such as the Clean Water Act of 1986 and the Water Resources Development Act of 1986 (U.S. Army Corp of Engineers 2006).

After the enactment of the Clean Water Act of 1986 there has been effort by the United States Army Corps of Engineers (USACE) to enforce the policy of “no net loss.” From 1996 until 2005 there was an estimated net gain of 10,000 ha/yr of wetland and associated uplands habitat. However, it is

unclear how successful these created, restored, enhanced, or preserved wetlands are in terms of restoring function and habitat. There is little information on what ecosystem functions were lost and what functions were gained (Mitsch and Gosselink 2007). Although restoration techniques continue to improve, there are still few overall guidelines for building a functional wetland outside of engineered hydrological changes. Because of ongoing restoration efforts, increasing threats to biodiversity and to wetlands, and lack of ecologically-based performance guidelines, there needs to be concerted and quantifiable efforts to assess the success of restored riverine wetland systems, so that general requirements can be implemented.

The Water Resources Development Act of 1986 led to the creation of the Missouri River Fish and Wildlife Mitigation Project (MRFWMP). The MRFWMP is dedicated to preserving existing habitat and restoring lost habitat that supports native vegetation, fish, and wildlife species within the flood plain. The requirements of the MRFWMP include monitoring restored habitat. The MRFWNP monitoring effort was expanded in 2007 to include reptiles and amphibians. In 2008 a multi-institutional project funded by the USACE was initiated in four states (Kansas, Missouri, Iowa, and Nebraska). The main goal of the overall monitoring effort is to assess the success of previously restored wetlands and to create wetland restoration guidelines for future use. Here I report the results of herpetofauna monitoring from three Missouri River bends in Nebraska, a subset of the overall project.

Amphibian monitoring was used to assess the success of previously restored wetlands. Amphibians were chosen to assess wetland success for three reasons: they are declining globally (Glascon *et al.* 2005; Pentranka and Holbrook 2006; Collins and Crump 2009; Crump 2010); they are important components of both the terrestrial and aquatic environments which they inhabit (Collins and Crump 2009; Vitt and Caldwell 2009; Collins and Halliday 2005); and they are indicators of wetland restoration success (Bowers *et al.* 1998; Collins and Halliday 2005; Welsh and Ollivier 1998). Amphibians serve as good indicators because their biphasic life cycle and permeable skin make them uniquely sensitive to conditions in the water and the surrounding terrestrial environment. Wetlands are complex ecosystems

that are hard to define and are made of both aquatic and terrestrial components. They are easily altered or degraded by changes in either habitat type. Thus, amphibians, whose biphasic life cycle and permeable skin makes them dependent on these intricate and essential ecosystems, are ideal indicators of restoration success.

Prior to the late 1990's amphibian monitoring data was mostly used to estimate abundance, birth rates, and survival probabilities. The creation of both the North American Amphibian Monitoring Program and the Amphibian Research and Monitoring Initiative led to the realization that estimating changes in abundance for large areas over time was not possible with the methods being utilized (Mossman *et al.* 1998, MacKenzie *et al.* 2006). The solution was found in measuring the presence or absence of a species at a number of wetlands. However this method also failed to account for detection differences, which, depending on the survey method, can vary by season, day, time of day, location, water temperature, air temperature, and among species (MacKenzie *et al.* 2006).

In 2002, MacKenzie *et al.* presented a method that statistically accounts for varying detection rates (MacKenzie *et al.* 2002). This method, occupancy modeling, consists of conducting two or more surveys at each site (e.g., a wetland) within a specified period of time. It is assumed that the system is closed and that the population (and therefore the probability of detection) does not change during the specified period of time. If a species is found at least once during the time period it is present at that site. However, if it is not found it is either present and undetected or not present. If one, reasonably, assumes that the detection and occupancy are not constant across sites, then it is possible to model the probability of detection and occupancy as a function of measured covariates. Occupancy analysis has often been used to compare different hypothesis about what affects amphibian distribution and detection (Mazerolle *et al.* 2005; Muths *et al.* 2005; Weir *et al.* 2005; Bailey *et al.* 2004). However, this information has rarely been used to specifically inform wetland restoration.

Therefore, I collected data that allowed for assessment of which environmental variables predict occupancy of wetlands by amphibians. Because amphibians are good indicators of wetland success, the environmental variables present at the occupied wetlands were used to help define wetland success. This definition was used to create wetland restoration guidelines. These can then be used to create higher quality restored wetlands with higher amphibian occupancy rates, as well as improve our basic understanding of these complex ecosystems.

METHODS

Study Sites

The sites used in this study consist of 55 restored wetlands distributed among three wetland complexes (river bends) between Nebraska City, NE and Nemaha, NE. Hamburg Bend is located 8 miles southeast of Nebraska City, Nebraska in Otoe County. This is the northern most wetland complex and it consists of ~638 ha of former agricultural land that was purchased by the USACE between 1993 and 2004. Hamburg Bend is adjacent to the right descending bank of the Missouri River between river miles 552 to 556. Due to the alterations of the Missouri River Bank Stabilization and Navigation Project (BSNP), the side channels were closed and this allowed the land to accrete. In response a side channel was constructed as well as several back water areas (U.S. Army Corp of Engineers 2006).

Kansas Bend is located 3 miles east of Peru, Nebraska in Nemaha County. This wetland complex is composed of ~427 ha of former agricultural land and is separated into two areas by privately owned farmland. Kansas Bend was purchased by the Corps between 1993 and 1999 and is adjacent to the right descending bank of the Missouri River between river miles 544 to 547. Due to the activities of the BSNP several side channels were closed using dikes and revetments. Two side channels have been reopened.

Langdon Bend is located 3 miles south of Brownville, Nebraska, in Nemaha County. This is the southernmost wetland complex and it is comprised of ~675 of former agricultural land that was purchased by the USACE between 1994 and 2003. Langdon Bend is the area adjacent to the right descending bank

of the Missouri River between river miles 520 and 532. Due to the Missouri River Bank Stabilization and Navigation Project the chute at Langdon Bend was closed and the side channel was cut off. The old channel could not be opened due to its proximity on the upstream end to the Cooper Nuclear Power Plant. Thus, the newly constructed channel is connected to the river at the outlet, but stops before meeting the river at the upstream end. The Corps in conjunction with the Nebraska Game and Parks Commission designed the wetland complex that was built on the west side of the levee. There are about 89 ha of wetlands that were constructed starting in July 2008 and ending May 2009.

Frog Call Surveys

Occupancy (presence/absence) data for the adult amphibians was collected using frog call surveys (MacKenzie *et al.* 2002). In 2012 and 2013 each wetland was visited twice within a two week period during each month; April, May, and June. Surveys were started a half an hour after sunset and were concluded no later than 1 am, because most temperate species peak calling hours end around midnight (Dodd 2010). To avoid disturbing the chorus, surveys were conducted 10 m from the shoreline after a 2 min acclimation period. We listened and recorded with a hand held recording device (Olympus DM-10) for five min and marked the presence of any species heard calling. The recordings were later listened to by a second observer to decrease listener bias (Lotz and Allen 2007).

Covariates

Both survey-specific covariates and site-specific covariates were collected for the frog call surveys. The survey-specific covariates (time of day, day of year, air temperature, water temperature, wind speed (m/s), water pH, relative humidity) were collected for every frog call survey. The survey-specific covariates moon phase (percent) and cloud cover (percent) were only collected for the frog call surveys. Moon phase percent was determined from the United States Naval Meteorology and Oceanography Command website (<http://aa.usno.navy.mil/data/docs/MoonFraction.php>). Percent cloud cover was visually estimated by looking at the sky at the beginning of the call survey and estimating the percent of the visible sky that had clouds. Moon phase and percent cloud cover were combined into a

covariate called moonshine by first subtracting the percent cloud cover from one and then multiplying this by the moon phase percent. This is essentially calculating the percent moonlight that is hitting the wetland.

Three of the site specific covariates, aquatic vegetation, adjacent vegetation, and the shallow water slope, were collected during tadpole dip-net surveys during the daytime. For each sweep, in addition to the presence of tadpoles, I determined whether the net was drawn through herbaceous, woody, or open water and a water height measurement was taken at 0.5 m and 1 m from the shoreline. Using all 20 net sweeps I then calculated the percentage of herbaceous, woody, and open water habitat along the 200 m surveyed. In addition, a visual survey of the entire wetland was done and aquatic vegetation was marked as present or absent. Aquatic vegetation was determined as present if it was present in the visual survey or if the percent of herbaceous and woody vegetation was $\geq 10\%$. A visual survey was also used for adjacent vegetation. We determined the percent grass, herbs and forbs, trees and shrubs, or bare ground within 1 m of the waterline. Shallow water slope was calculated using the water height measurement at 1 m and dividing by 100 cm. Two covariates, whether a wetland is ephemeral or not and wetland size, were determined at the beginning of the monitoring activity in 2009. Wetland size was recorded as small (≤ 2 ha), medium (2-5 ha), and large (> 5 ha). Land cover and the distance to the nearest wetland were calculated using ArcGIS. Land cover was determined by creating a 1000 m buffer around each wetland and determining the percentage of grass, forest, and agriculture within that buffer, and assigning land cover type to the dominant (%) land cover class. The distance to the nearest wetland was determined from digital land cover maps, and calculated from centroid to centroid.

Analysis

An occupancy modeling analysis was conducted using the package unmarked in program R (R Development Core Team 2008; Fiske and Chandler 2011). Occupancy modeling represents a major improvement to amphibian monitoring techniques because it accounts for differences in detection, which may vary across time, across wetlands, and across species. As manifest in Program Presence (Hines

2006) and in the package unmarked in program R, I used an occupancy model structured multi-model inference to select models that best fit the observed data. Like all implementations of multi-model inference, model selection is critical to sound inference. Model sets were created *a priori* and were run as single species - single season models, using month as season. A separate model set was created for tadpole detection (9 models), adult detection (12 models), and a single occupancy model set for both the adults and tadpoles (12 models) (Table 2.1 and Table 2.2). Some of the models in each model set correlated, but they were all included because they have different management implications. Unless the naïve occupancy (number of wetlands that a species was present at least once/total number of wetlands) was < 0.10 a model set was run for every species, each month (April, May, and June), both years (2012 and 2013), and for both tadpoles and adults. This cutoff was chosen because species with low detection have uncertain occupancy and most model sets below this naïve occupancy would not run in program R (Hellman 2013). Detection was assessed separately from occupancy. Detection was determined first using the adult detection model set. The occupancy portion of each model was held constant using global occupancy. Then, the covariate/s in the detection portion of the top model for each detection model set was used to hold detection constant while running the occupancy model sets. Due to drought in 2012 and the beginning of 2013, many of the wetlands dried up and both adult and tadpole occupancy was affected. This increased the number of models and variables in the global occupancy model that failed to converge. I removed variables from the global model if convergence was not achieved and any models that did not converge were removed from the model sets.

To determine what variables affect detection and occupancy, a confidence set accounting for 95% of overall model weight was considered (Burnham and Anderson 2002). The 95% confidence set was calculated by adding up the models AICc weight until it was greater than or equal to 0.95. A large confidence set was used to ensure that no important parameters were excluded. As previously mentioned, my research is part of a multi-state monitoring project that is set up to function in an adaptive management framework. If continued, an inclusive confidence set allows for more confident model set

refinement. Within the confidence set the AICc weight was used to determine the model's fit. Parameter estimates were used to determine what affect a covariate/s had on detection or occupancy. In this case, a positive parameter estimate for a continuous variable like slope or distance to the nearest wetland meant that the detection or occupancy increased as the covariate/s increased. A positive parameter estimate for a categorical variable like aquatic vegetation, bend, or size meant that occupancy or detection was higher with/at that covariate than without/not at it. For continuous variables a negative parameter estimate meant that detection or occupancy was decreasing as the covariate/s increased. For categorical variables a negative parameter estimate meant that occupancy or detection was lower with/at that covariate than without/not at it. The covariates in the top models were graphed to show how detection or occupancy was affected by that particular covariate.

Table 2.1. Detection models for adult anurans species in restored wetlands along the Missouri River in southeast Nebraska. Frog call and covariate data was collected during April, May, and June of 2012 and 2013. Detection is represented by p , and occupancy is represented by Ψ . Detection was assessed first, using the global occupancy to hold occupancy constant. The global covariates are bend, wetland size, slope, aquatic vegetation, adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest. The top detection models held the covariates that were most likely to have affected adult anuran detection. The detection covariate/s in the top detection model for each data set was used to hold detection constant when the occupancy was assessed using a second set of models (Table 2.2). The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(\text{Global Covariates}) p(.)$
Time	$\Psi(\text{Global Covariates}) p(\text{time})$
Wind (m/s)	$\Psi(\text{Global Covariates}) p(\text{wind})$
Day (julian date)	$\Psi(\text{Global Covariates}) p(\text{julian})$
Wetland Size	$\Psi(\text{Global Covariates}) p(\text{size})$
Moonshine	$\Psi(\text{Global Covariates}) p(\text{moonshine})$
Air Temperature	$\Psi(\text{Global Covariates}) p(\text{airtemp})$
Water Temperature	$\Psi(\text{Global Covariates}) p(\text{H2Otemp})$
Environmental Covariates 1	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind})$
Environmental Covariates + Day	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+julian})$
Environmental Covariates 2	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+moonshine})$
Global	$\Psi(\text{Global Covariates}) p(\text{time+wind+julian+size+moonshine+airtemp+H2Otemp})$

Table 2.2. Occupancy models for adult anurans in restored wetlands along the Missouri River in Southeast Nebraska. Frog call and covariate data was collected during April, May, and June of 2012 and 2013. Detection is on the right represented by p , and occupancy is on the left represented by Ψ . Occupancy was assessed after detection and used the detection covariate/s in the top detection model for each data set to hold detection constant. The top detection models held the covariates that were most likely to have affected adult anuran detection. The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(.) p(\text{Top Detection Covariate/s})$
Bend	$\Psi(\text{bend}) p(\text{Top Detection Covariate/s})$
Wetland Size	$\Psi(\text{size}) p(\text{Top Detection Covariate/s})$
Slope	$\Psi(\text{slope}) p(\text{Top Detection Covariate/s})$
Aquatic Vegetation	$\Psi(\text{aquatic}) p(\text{Top Detection Covariate/s})$
Adjacent Vegetation	$\Psi(\text{adjacent}) p(\text{Top Detection Covariate/s})$
Ephemeral Wetland	$\Psi(\text{ephemeral}) p(\text{Top Detection Covariate/s})$
Distance to Nearest Wetland	$\Psi(\text{distwet}) p(\text{Top Detection Covariate/s})$
Land Cover	$\Psi(\text{agr+forest}) p(\text{Top Detection Covariate/s})$
Wetland Vegetation	$\Psi(\text{aquatic+adjacent}) p(\text{Top Detection Covariate/s})$
Connectivity	$\Psi(\text{distwet+agr+forest}) p(\text{Top Detection Covariate/s})$
Global	$\Psi(\text{bend+size+slope+aquatic+adjacent+ephemeral+distwet+agr+forest}) p(\text{Top Detection Covariate/s})$

RESULTS

Frog Call Survey Detection

The number of wetlands holding water varied from month to month during 2012 and 2013. Annual variation in flood plains wetlands is typical; however the variation in 2012 and 2013 was extreme. In 2011 the Missouri River experienced historic flooding due to both higher than average snowpack melt and higher than average amounts of rainfall during the spring and summer months. This was followed by a historic drought in 2012 that continued into 2013. The drought was caused by record high temperatures combined with below average snowpack melt and precipitation. From mid-April until the beginning of May 2012 the water table was still high enough that average rainfall caused an increase in the number of wetlands holding water. However, from that point on the high temperatures continued to lower the number of wetlands that were holding water. Southeast Nebraska received average to above average rainfall in March and April of 2013, but the soil was not saturated enough for most of the wetlands to hold water until the extreme rainfall events in late May of 2013. Thus, frog call surveys were conducted at 26 wetlands in April 2012, 30 wetlands in May 2012, 21 wetlands in June 2012, 22 wetlands in April 2013, 20 wetlands in May 2013, and 43 wetlands in June 2013.

There were 12 models in the detection model set for adult anurans (Table 2.1). Ideally, these would have been conducted for all seven species, all months of April, May, and June, and both years 2012 and 2013. However, only the Plains Leopard Frog had naïve occupancy high enough for all three months and both years. Thus, the models were run 28 times and there were 28 top models and confidence sets. There was variance between the species, months, and years and many of the models had weak inference. The confidence sets contained between 1 to 7 models (Appendix A). Overall, detection was affected by water temperature, air temperature, and moonshine. However, the null model was the top-ranked model in 15 of the 28 cases, and was in the confidence set 23 times (Table 2.3, Appendix A). This suggests that within the methodological constraints that the data was surveyed, the covariates measured are not affecting the detection of the adult anurans, or that inference was weak relative to covariate affect, so that

it was not possible to clearly identify covariates. Below I address those covariates that were in the confidence set as well as individual species.

Water Temperature

The water temperature model was the top model for the detection model sets five times and among the detection confidence sets sixteen times. Three out of the five top models had an AICc weight over 0.90 (Table 2.3, Appendix A). For thirteen out of the sixteen occurrences in the confidence set the parameter estimate was positive, thus the probability of detection increased as the water temperature increased (Figure 2.1, Appendix A).

Air Temperature

The model for air temperature was the top model for the detection model sets three times and among the detection confidence sets fourteen times (Table 2.3, Appendix A). For eight of the fourteen occurrences in the confidence set the parameter estimate was positive, but for six of the fourteen occurrences the parameter estimate was negative (Appendix A). This disparity is seen in the top models as well. For both the Great Plains Toad and the American Bullfrog in June 2012 the parameter estimate is negative and the probability of detection decreased with the increase in air temperature (Figure 2.1, Appendix A). However, for the Blanchard's Cricket Frog in April 2012 the parameter estimate is positive and the probability of detection increased as the air temperature increased (Figure 2.1, Appendix A). All of the air temperature models in confidence set for the month of April had positive parameter estimates and all of the parameter estimates for the air temperature models in confidence sets for the month of June, except for Plains Leopard Frog in June 2013, had negative parameter estimates. The positive parameter estimates at the start of the calling season and the negative parameter estimates at the end suggest that there is a temperature range that adult anurans will call in. We know that anuran species have a threshold temperature that must be reached before they start calling (Oseen and Wassersug 2002); however there is no evidence that there is a high temperature threshold for temperate species. The decrease in detection for the higher temperatures could also be due to the lower water levels and rapidly shrinking wetlands.

Moonshine

The model for moonshine was the top model for the detection model sets three times and among the detection confidence sets nineteen times (Table 2.3, Appendix A). It often was ranked after the null model or among model sets where many of the models had low weights. However, it was the top model three times and two of those times it had an AICc weight over 0.80 (Table 2.3, Appendix A). Both models with the AICc weight over 0.80 were for the Plains Leopard Frog, one for June 2012 and the other for April 2013. They also both had a positive parameter estimate that showed an increase in detection as the moonshine percent increased. However, the opposite effect was found for the Boreal Chorus Frog in June (Figure 2.1). The model for moonshine had a positive parameter estimate in thirteen of the nineteen confidence sets and a negative parameter estimate in six of the confidence sets.

Time

The model for time was among the detection confidence sets twelve times. However, it was the top model only once for the Plains Leopard Frog in May of 2012 with an AICc weight of 0.96 (Table 2.3, Appendix A). For that model the parameter estimate was negative and therefore the probability of detection decreased as the time got closer to midnight (Figure 2.1, Appendix A). The model for time had a negative parameter estimate in seven of the twelve confidence sets it was present, while it had a positive parameter estimate five times.

Wind

Wind was the top model once, but was among the detection confidence sets fifteen times (Table 2.3, Appendix A). It was the top model for Gray Treefrogs in June 2013, with a negative parameter estimate (Appendix A). However, when graphed there is no clear trend (Figure 2.1). While the top model it only has 0.40 AICc weight, followed closely by the null model with 0.34 AICc weight (Table 2.3, Appendix A).

Other Covariates

The model environmental covariates 1 (water temperature + air temperature + wind) was among the detection confidence sets twice and environmental covariates 2 (water temperature + air temperature + wind + moonshine) was among the confidence sets once. The models for day, wetland size, environmental covariates + day, and the global model were never among the confidence sets (Appendix A).

Blanchard's Cricket Frog (*Acris blanchardi*)

There was sufficient data to assess detection models for Blanchard's Cricket Frog adults in all three months of 2012 and in May and June of 2013. In four of the five model sets null was the top model. In April of 2012 there were three models in the confidence set and air temperature was the top model with 0.38 AICc weight, but Null was the second model with the same AICc weight (Table 2.3, Appendix A). The parameter estimate was positive and the probability of detection increased as the air temperature increased (Figure 2.1, Appendix A).

Woodhouse's Toad (*Anaxyrus woodhousii*)

For the Woodhouse's Toad adults there was sufficient data to assess detection models for April 2012 and May and June in 2013. There were three models in the confidence set for April 2012 and the top model was water temperature with AICc weight 0.79. The parameter estimate was positive and the probability of detection increased as the water temperature increased (Table 2.3, Figure 2.1, Appendix A). For May and June of 2013 null was the top model. There were six models in the confidence set for May 2013 and four in the confidence set for June 2013. For both May 2013 and June 2013 the model for moonshine was the second model with AICc weights of 0.19 and 0.33 respectively. However, May 2013 has positive parameter estimate and June 2013 has a negative estimate. Thus, detection has opposing relationships with moonshine. It is possible that these covariates effect detection, but they also fall after the null model so it is also possible that none of the models in the model set contain the covariates that are effecting adult detection.

Great Plains Toad (*Anaxyrus cognatus*)

For the Great Plains Toad there was sufficient data to assess detection models for June 2013. There were four models in the confidence set and the top model was air temperature with 0.62 AICc weight, followed by water temperature with AICc weight 0.28 (Table 2.3, Appendix A). The probability of detection decreases as both air temperature and water temperature increase (Figure 2.1). The water temperature shows the opposite trend from the rest of the water temperature models, but this is possibly because most of the models are in April and don't go above 22C while these are in June. In addition, the Great Plains Toad is a more intermittent breeder than most of the other species in Nebraska. It tends to only breed in ephemeral wetlands that form after a heavy rain. Thus, the water and air would be cooler after rain fall and they would be detected more often in these cooler temperatures.

Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*)

For the Gray Treefrog Complex there was sufficient data to assess detection models for April and June in 2012 and May and June in 2013. Null was the top model for June 2012 and May 2013 (Table 2.3, Appendix A). Water temperature was the top model for April 2012 with AICc weight of 0.94 (Table 2.3, Appendix A). There were only two models in this confidence set with null as the second model (Appendix A). Like a majority of the water temperature models the parameter estimate was positive and the probability of detection increased as the water temperature increased (Figure 2.1). In June of 2013 there were five models in the confidence set and wind was the top model with AICc weight 0.4, followed closely by the null model with AICc weight 0.34 (Table 2.3, Appendix A). The parameter estimate for the model of wind was negative, suggesting that the probability of detection decreases as wind increases. However, the standard errors are large which implies weak inference and low affect (Figure 2.1).

Plains Leopard Frog (*Lithobates blairi*)

For the Plains Leopard Frog there was sufficient data to assess detection models for all months, both years. Moonshine was the top model twice in June 2012 and April 2013 with AICc weights 0.91 and 0.87 respectively (Table 2.3, Appendix A). For both models there were only two models in the

confidence set and the probability of detection increased as the percent moonshine increased (Appendix A, Figure 2.1). Water temperature was the top model and only model in the confidence set for April 2012 with AICc weight 1. The water temperature model had a positive parameter estimate and the probability of detection increased with the increase in water temperature (Table 2.3, Appendix A, Figure 2.1). Time was the top model and only model in the confidence set for May 2012 with AICc weight 0.96. (Table 2.3, Appendix A). The model of time had a negative parameter estimate and thus the probability of detection decreasing as it nears midnight (Figure 2.1). It is known that the peak calling hour for temperate species ends around midnight (Dodd 2010), so it is possible this is the trend seen in the time model predictions. Null was the top model May and June of 2013 with confidence sets containing five and seven models.

American Bullfrog (*Lithobates catesbeianus*)

For the American Bullfrog there was sufficient data to assess detection models for all months in 2012 and in May and June in 2013. For April and May 2012 and May 2013 the best model was the null model with 6, 5, and 4 models in the confidence set respectively (Table 2.3, Appendix A). There were other models after the null model for each, but due to the low AICc weights and the fact that they fall under the null model they probably do not have a significant effect on detection. For June 2012 the top model was air temperature with AICc weight 0.42, followed by null with AICc weight 0.26, moonshine with AICc weight 0.25, and wind with AICc weight 0.03 (Table 2.3, Appendix A). When graphed the air temperature model showed the same trend as the Great Plains Toad with the probability of detection decreasing as the air temperature increased (Figure 2.1). For June 2013 the best model was water temperature with AICc weight 0.58, followed by the environmental covariates model (water temperature + air temperature + wind) with AICc weight 0.35, and the environmental covariates model 2 (water temperature + air temperature + wind + moonshine) with AICc weight 0.06 (Table 2.3, Appendix A). The weights of the second and third models are likely due to the presence of the covariate water temperature. The parameter estimate was positive and so the probability of detection increased as the water temperature increased (Figure 2.1).

Boreal Chorus Frog (*Pseudacris maculata*)

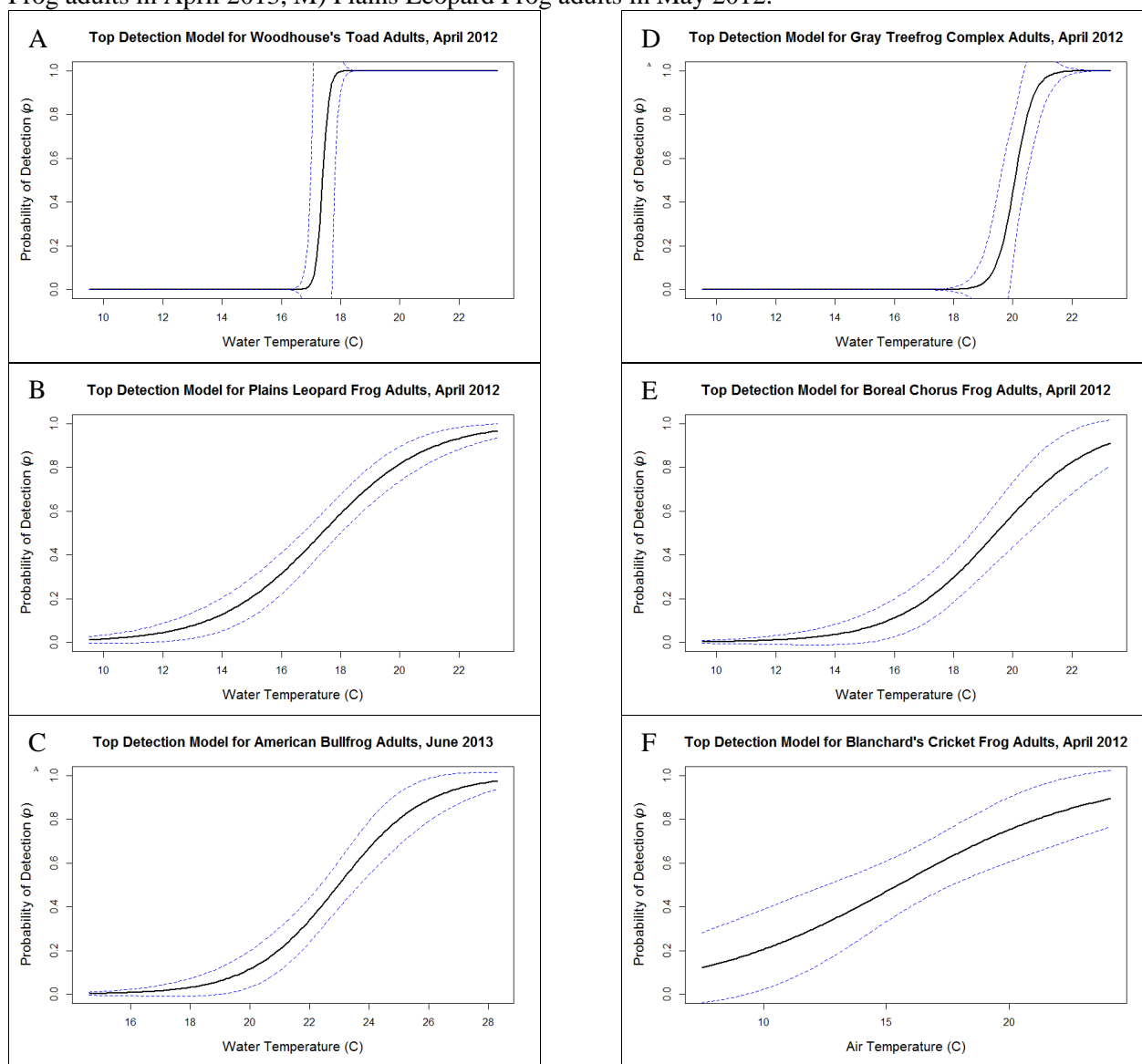
For the Boreal Chorus Frog there was sufficient data to assess detection models for April 2012 and all three months in 2013. In April 2012 the top model and only model in the confidence set was water temperature with AICc weight 0.96 (Table 2.3, Appendix A). The probability of detection increasing as the water temperature increased (Figure 2.1). For April and May of 2013 the top model was the null model with five models in each confidence set and AICc weights 0.62 and 0.69 respectively. (Table 2.3, Appendix A). For June 2013 the top model was moonshine with six models in the confidence set and AICc weight of 0.42 (Table 2.3, Appendix A). The probability of detection decreased as percent moonshine increased (Figure 2.1).

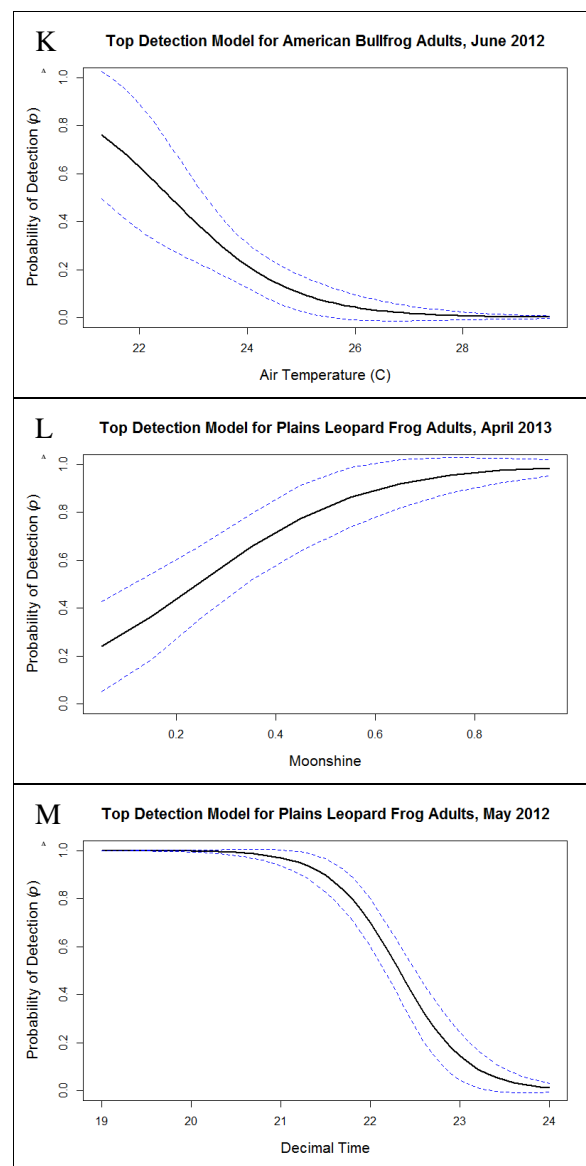
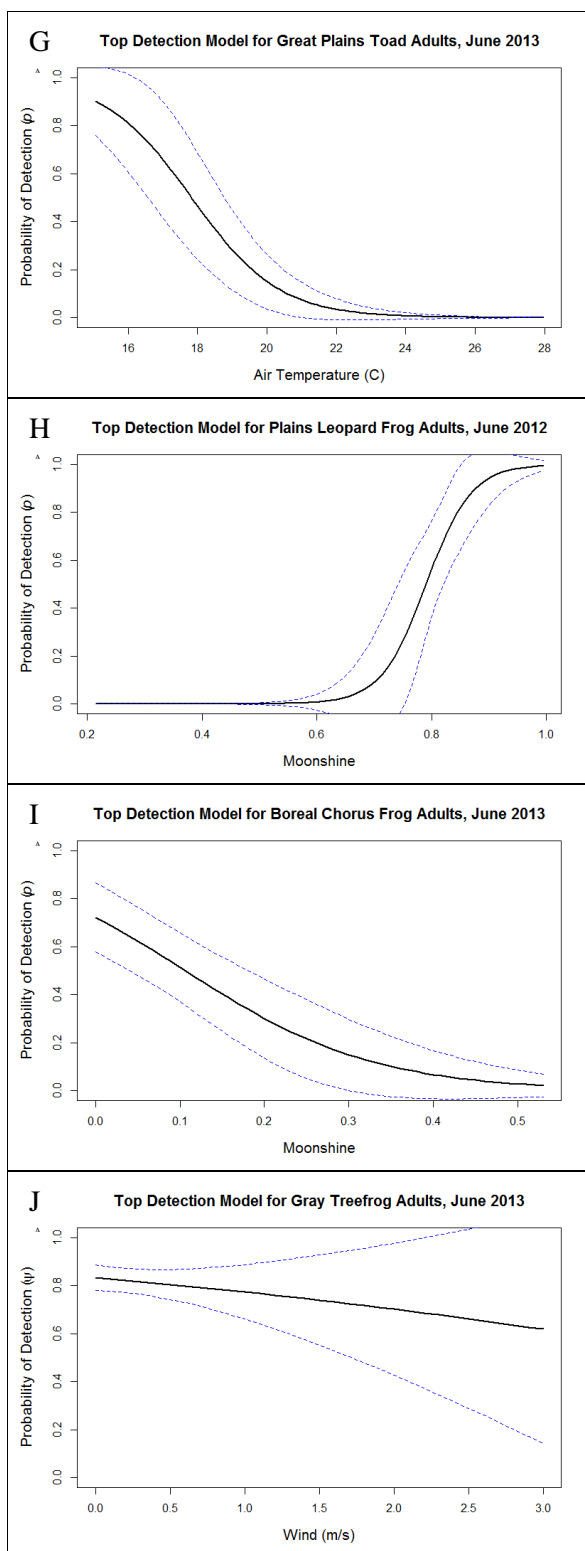
Table 2.3. Top detection models for adult anuran species in restored wetlands along the Missouri River in southeast Nebraska. A set of detection models was assessed for every species with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top detection models contain the measured covariates that affected frog call detection the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the detection models had a positive or negative relationship with detection. The detection estimate (\hat{p}) also includes the standard error. For the last column CL refers to confidence limits.

Species	Month	Year	Model	AICcWt	$\hat{\beta}$	\hat{p}	95% CL
Blanchard's Cricket Frog	April	2012	$p(\text{airtemp})$	0.38	+	0.520 \pm 0.130	0.310-0.720
Blanchard's Cricket Frog	May	2012	$p(.)$	0.36	NA	0.500 \pm 0.072	0.380-0.620
Blanchard's Cricket Frog	June	2012	$p(.)$	0.79	NA	0.660 \pm 0.100	0.470-0.800
Woodhouse's Toad	April	2012	$p(\text{H2Otemp})$	0.79	+	0.000 \pm 0.004*	0.000-1.000*
Gray Treefrog Complex	April	2012	$p(\text{H2Otemp})$	0.94	+	0.000 \pm 0.000*	0.000-0.999
Gray Treefrog Complex	June	2012	$p(.)$	0.89	NA	0.750 \pm 0.150	0.440-0.920
Plains Leopard Frog	April	2012	$p(\text{H2Otemp})$	1.00	+	0.370 \pm 0.094	0.230-0.530
Plains Leopard Frog	May	2012	$p(\text{time})$	0.96	-	0.420 \pm 0.120	0.250-0.620
Plains Leopard Frog	June	2012	$p(\text{moonshine})$	0.91	+	0.001 \pm 0.004	0.000-0.970*
American Bullfrog	April	2012	$p(.)$	0.31	NA	0.170 \pm 0.087	0.070-0.360
American Bullfrog	May	2012	$p(.)$	0.55	NA	0.340 \pm 0.093	0.200-0.500
American Bullfrog	June	2012	$p(\text{airtemp})$	0.42	-	0.069 \pm 0.065	0.010-0.280
Boreal Chorus Frog	April	2012	$p(\text{H2Otemp})$	0.96	+	0.140 \pm 0.094	0.040-0.370
Blanchard's Cricket Frog	May	2013	$p(.)$	0.54	NA	0.650 \pm 0.092	0.490-0.780
Blanchard's Cricket Frog	June	2013	$p(.)$	0.47	NA	0.610 \pm 0.059	0.510-0.700
Woodhouse's Toad	May	2013	$p(.)$	0.55	NA	0.530 \pm 0.160	0.280-0.770
Woodhouse's Toad	June	2013	$p(.)$	0.40	NA	0.230 \pm 0.055	0.150-0.330
Great Plains Toad	June	2013	$p(\text{airtemp})$	0.62	-	0.050 \pm 0.056	0.007-0.270
Gray Treefrog Complex	May	2013	$p(.)$	0.86	NA	0.660 \pm 0.099	0.480-0.800
Gray Treefrog Complex	June	2013	$p(\text{wind})$	0.40	-	0.740 \pm 0.190	0.350-0.940
Plains Leopard Frog	April	2013	$p(\text{moonshine})$	0.87	+	0.820 \pm 0.130	0.510-0.950
Plains Leopard Frog	May	2013	$p(.)$	0.63	NA	0.840 \pm 0.110	0.570-0.950
Plains Leopard Frog	June	2013	$p(.)$	0.34	NA	0.500 \pm 0.059	0.400-0.590
American Bullfrog	May	2013	$p(.)$	0.58	NA	0.170 \pm 0.110	0.053-0.420
American Bullfrog	June	2013	$p(\text{H2Otemp})$	0.58	+	0.260 \pm 0.100	0.130-0.460
Boreal Chorus Frog	April	2013	$p(.)$	0.62	NA	0.660 \pm 0.089	0.500-0.790
Boreal Chorus Frog	May	2013	$p(.)$	0.69	NA	0.630 \pm 0.120	0.420-0.800
Boreal Chorus Frog	June	2013	$p(\text{moonshine})$	0.42	-	0.028 \pm 0.057	0.000-0.470*

*Number is greater than zero but is smaller than 0.0005

Figure 2.1. Graphs of top detection model predictions for frog call surveys. The graphs are for A) Woodhouse's Toad adults in April 2012, B) Plains Leopard Frog adults in April 2012, C) American Bullfrog adults in June 2013, D) Gray Treefrog Complex adults in April 2012, E) Boreal Chorus Frog adults in April 2012, F) Blanchard's Cricket Frog adults in April 2012, G) Great Plains Toad adults in June 2013, H) Plains Leopard Frog in June 2012, I) Boreal Chorus Frog adults in June 2013, J) Gray Treefrog Complex adults in June 2013, K) American Bullfrog adults in June 2012, L) Plains Leopard Frog adults in April 2013, M) Plains Leopard Frog adults in May 2012.





Frog Call Survey Occupancy

There were 12 models in the occupancy model set for adult anurans (Table 2.2). Ideally, these would have been conducted for all seven species, each month of April, May, and June, and both years 2012 and 2013. However, only the Plains Leopard Frog had naïve occupancy high enough to be run for all three months and both years. Thus, the models were conducted 28 times and there were 28 top models and confidence sets. There was variation among the species, months, and years. There was also weak inference for many of the models. The confidence sets contained between 2 to 9 models (Appendix A). Overall, occupancy was affected by aquatic vegetation, slope, bend, whether or not a wetland was ephemeral, and the distance to the nearest wetland plus land cover. However, null was the top model 5 of the 28 model runs and was in the confidence set 26 times (Table 2.4, Appendix A). Below I address those occupancy covariates that were in the confidence set as well as individual species.

Aquatic Vegetation

The model for aquatic vegetation was the top occupancy model seven times and among the confidence sets 27 times (Table 2.4, Appendix A). This is not only the covariate that is the top model and part of the confidence sets most often, but it also is one of the few covariates that has the same effect on occupancy for almost every species, months, and years. Out of the 27 confidence sets that the model was present in it had a negative parameter estimate only twice. The model for aquatic vegetation had a positive parameter estimate 25 out of 27 confidence sets and thus the probability of occupancy was highest when aquatic vegetation is present (Table 2.4, Appendix A, Figure 2.2).

Slope

Slope was the top model six times and was among the occupancy confidence sets 24 times (Table 2.4, Appendix A). Out of the 24 occurrences in the confidence set fifteen had negative parameter estimates, eight had positive parameter estimates, and one had such low sample size that the parameter estimate was unattainable. Four of the eight positive parameter estimates were for the American Bullfrog (out of 5 American Bullfrog confidence sets with the slope model) and two were for the Blanchard's

Cricket Frog (out of 3 Blanchard's Cricket Frog confidence sets that included the slope model) (Appendix A). For the three top models that were not for either the Blanchard's Cricket Frog or the American Bullfrog, the model for slope had a negative parameter estimate and the probability of occupancy decreased as the percent slope increased (Figure 2.2).

Bend

Bend was the top occupancy model four times and among the confidence sets 15 times (Table 2.4, Appendix A). The model for this graph simply showed that certain species were more likely to be present at certain bends during certain months (Figure 2.2).

Ephemeral

The model for whether or not a wetland is ephemeral was the top model twice and was among the occupancy confidence sets 24 times (Table 2.4, Appendix A). Out of the 24 occurrences in the confidence sets 16 had positive parameter estimates, six had negative parameter estimates, and two had inferences too low to produce a parameter estimate (Appendix A). For both top models the parameter estimate was positive and the probability of occupancy was higher when a wetland was ephemeral (Figure 2.2).

Distance to the Nearest Wetland

Distance to the nearest wetland was the top occupancy model once and was among the confidence sets 15 times (Table 2.4, Appendix A). Out of the 15 confidence sets seven had a negative parameter estimate, seven had a positive parameter estimate, and one had sample size too low to produce a parameter estimate (Appendix A). All four potential Boreal Chorus Frog confidence sets contained the model distance to the nearest wetland, and all had a negative parameter estimate. This model was also in four of the American Bullfrog confidence sets and three of the four models had a positive parameter estimate. (Appendix A). This was the top model for the Plains Leopard Frog in May 2012 and it had a

negative parameter estimate, thus the probability of occupancy decreased as the distance to the next wetland increased (Figure 2.2).

Distance to the Nearest Wetland + Land Cover

The model for distance to the nearest wetland plus land cover was the top model twice and among the occupancy confidence sets once. Both top models produced parameter estimates, but no standard errors. The first top model was for the Blanchard's Cricket Frog in April 2012 and the second was for the American Bullfrog in May 2013. Both had negative parameter estimates for forest and positive parameter estimates for agriculture. However, the model for Blanchard's Cricket Frog had a negative parameter estimate for the distance to the nearest wetland and the American Bullfrog had a positive parameter estimate. Therefore, for the Blanchard's Cricket Frog the probability of occupancy was highest when the distance to the nearest wetland was short and the wetland was surrounded by agriculture. For the American Bullfrog the probability of occupancy was highest when the slope was steep and the wetland was surrounded by agriculture.

Wetland Size

The model for size was the top occupancy model once and among the confidence sets 10 times (Table 2.4, Appendix A). Small wetlands had a positive parameter estimate three times, medium wetlands had a positive parameter estimate seven times, and large wetlands had a positive parameter estimate five times (Appendix A). Two of the positive parameter estimates for the large wetlands, and one for the medium wetlands were for the American Bullfrog, and because they are an invasive species their needs should not be considered. Therefore, occupancy was highest at medium sized wetlands. However, this could have been skewed by the drought conditions in 2012 and the beginning of 2013. If all the small wetlands were dry they would have to inhabit larger wetlands.

Other Covariates

The model of adjacent vegetation was never a top model and was among the occupancy confidence sets five times. It was the second model three times and never had an AICc weight higher than 0.26 (Table 2.4, Appendix A). Due to its low AICc weights this covariate likely has little effect on occupancy. The model for land cover was never a top occupancy model and was among the confidence sets eight times. It was the last model in the confidence set five of the eight times and never had an AICc weight higher than 0.11 (Appendix A). Given this it is not likely to play an important role in determining occupancy. The model of aquatic vegetation plus adjacent vegetation was never a top model and was among the occupancy confidence sets seven times (Appendix A). Five of the seven times it was the last model in the confidence set and it never had an AICc weight higher than 0.07. Like the model for land cover, the low placement in the confidence sets and low AICc weights suggests that this is not likely to play a role in determining occupancy. The global model was the only occupancy model that was never among the occupancy confidence sets (Appendix A).

Blanchard's Cricket Frog (*Acris blanchardi*)

There was sufficient data to assess occupancy models for Blanchard's Cricket Frog adults all three months in 2012 and May and June in 2013. For April 2012 there were five models in the confidence set and the top model was distance to nearest wetland plus land cover with AICc weight 0.67 (Table 2.4). The models for distance to nearest wetland and land cover rank below this model, suggesting that the combination of the two is more important than the individual covariates. The model produced parameter estimates, but did not produce standard errors. Distance to the nearest wetland had a negative parameter estimate and the land cover covariate agriculture had a positive parameter estimate, while forest had a negative parameter estimate. Therefore, the probability of occupancy is highest when a wetland is surrounded by agriculture and the distance to the nearest wetland is short (Appendix A: Table 29). Bend is the top model in May of 2012 with AICc weight 0.28 (Table 2.4). The confidence set contains nine models and the second model in the null model with AICc weight 0.23 (Appendix A: Table

30). This suggests that most of the variables play some role in determining amphibian occupancy but in this case none of them are the sole reason. When the predictions for the bend model are graphed May 2012 shows that Langdon Bend had a higher probability of occupancy than the other two bends (Figure 2.2). Aquatic vegetation is the top model twice, in June 2012 and June 2013. For June 2012 the confidence set contains six models and aquatic vegetation only has 0.33 AICc weight, but in June 2013 there are four models in the confidence set and aquatic vegetation had 0.70 AICc weight (Table 2.4, Appendix A: Table 31 and 43). In both cases the probability of occupancy is higher when aquatic vegetation is present (Figure 2.2). The top model for May 2013 was slope with five other models in the confidence set and an AICc weight of 0.49 (Table 2.4, Appendix A: Table 32). As opposed to most of the other species except Bullfrogs, the parameter estimate is positive and therefore the probability of occupancy increased as the slope percent increased (Figure 2.2).

Woodhouse's Toad (*Anaxyrus woodhousii*)

There was sufficient data to assess occupancy models for Woodhouse's Toad April 2012 and May and June of 2013. Bend was the top model for April 2012 with eight models in the confidence set and AICc weight 0.33, and May 2013 with five models in the confidence set and AICc weight 0.33 (Table 2.4, Appendix A: Tables 32 and 44). Presence was indicated by a positive parameter estimate. Woodhouse's Toads were only heard at Hamburg Bend in April 2012 and were heard at Hamburg Bend and Langdon Bend in May 2013, but not Kansas Bend (Figure 2.2). Size was the top model for June 2013, but there were seven models in the confidence set and it only had an AICc weight of 0.23 (Table 2.4, Appendix A: Table 35). It is clear that there is no difference in occupancy between the different wetland sizes (Figure 2.2).

Great Plains Toad (*Anaxyrus cognatus*)

There was only sufficient data to assess occupancy models for the Great Plains Toad in June 2013. The top model was slope with AICc weight 0.64. The confidence set contained four models and the second model was the distance to the nearest wetland with AICc weight 0.24. This was followed by

the null model with AICc weight 0.05 and size with AICc weight 0.02 (Table 2.4, Appendix A: Table 46). The parameter estimate was negative and the probability of occupancy decreased as the slope percentage increased (Appendix A, Figure 2.2). For the model of the distance to the nearest wetland the parameter estimate was negative and the probability of occupancy decreased as the distance increased (Appendix A: Table 46, Figure 2.2).

Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*)

There was sufficient data to assess occupancy models for Gray Treefrog adults April and June of 2012 and May and June of 2013. Slope was the top model for April 2012 with an AICc weight of 0.85 and a confidence set containing only two models (Table 2.4, Appendix A: Table 33). It is also the second model, with AICc weight 0.25, in June 2013 with a confidence set of five (Appendix A: Table 38). For both instances the parameter estimate is negative and thus the probability of occupancy decreases as the percent slope increases (Appendix A: Table 33, Figure 2.2). Ephemeral was the top model once for June 2012, with a confidence set of six but an AICc weight of 0.76 (Table 2.4, Appendix A: Table 34). The parameter estimate was positive and the probability of occupancy was higher when a wetland was ephemeral (Appendix A, Figure 2.2). The model for aquatic vegetation was the top model for both May and June of 2013, with AICc weights of 0.62 and 0.51 respectively (Table 2.4). May 2013 had a confidence set of six and June 2013 had a confidence set of five. In both instances the parameter estimates were positive and the probability of occupancy was higher when aquatic vegetation was present in a wetland (Figure 2.2).

Plains Leopard Frog (*Lithobates blairi*)

There was sufficient data to assess occupancy models for Plains Leopard Frogs all months in both 2012 and 2013. Null was top model twice, in April 2012 with a confidence set of six and AICc weight 0.68 and April 2013 with a confidence set of eight and AICc weight 0.29 (Table 2.4, Appendix A: Tables 35 and 49). The model for distance to the nearest wetland was the top model for May 2012 with AICc weight 0.31 and the third model for April 2013 with AICc weight 0.13. May 2012 had a confidence set of

seven with null as the second model with AICc weight 0.24 (Table 2.4, Appendix A: Tables 36 and 49). For May 2012 the parameter estimate was negative therefore the probability of occupancy decreased as the distance increased (Appendix A, Figure 2.2). June 2012 had a confidence set of five and the top model was ephemeral with AICc weight 0.47, followed closely by size with AICc weight 0.3 (Table 2.4, Appendix A: Table 37). The parameter estimate for ephemeral was positive and the probability of occupancy was higher when a wetland was ephemeral (Appendix A, Figure 2.2). Adjacent vegetation was the top model for May 2013 with AICc weight 0.79, and the second model for April and May 2013 with AICc weights 0.18 and 0.26 (Table 2.4, Appendix A: Tables 49-51). The confidence set for May 2013 contained three models and given that the parameter estimate was positive, the probability of occupancy was higher when a wetland had aquatic vegetation (Appendix A: Table 50, Figure 2.2). Slope was the top model for June 2013 with AICc weight 0.3 and a confidence set of eight (Table 2.4, Appendix A: Table 51). The parameter estimate for this model was negative, therefore the probability of occupancy decreased as the slope percentage increased (Appendix A, Figure 2.2).

American Bullfrog (*Lithobates catesbeianus*)

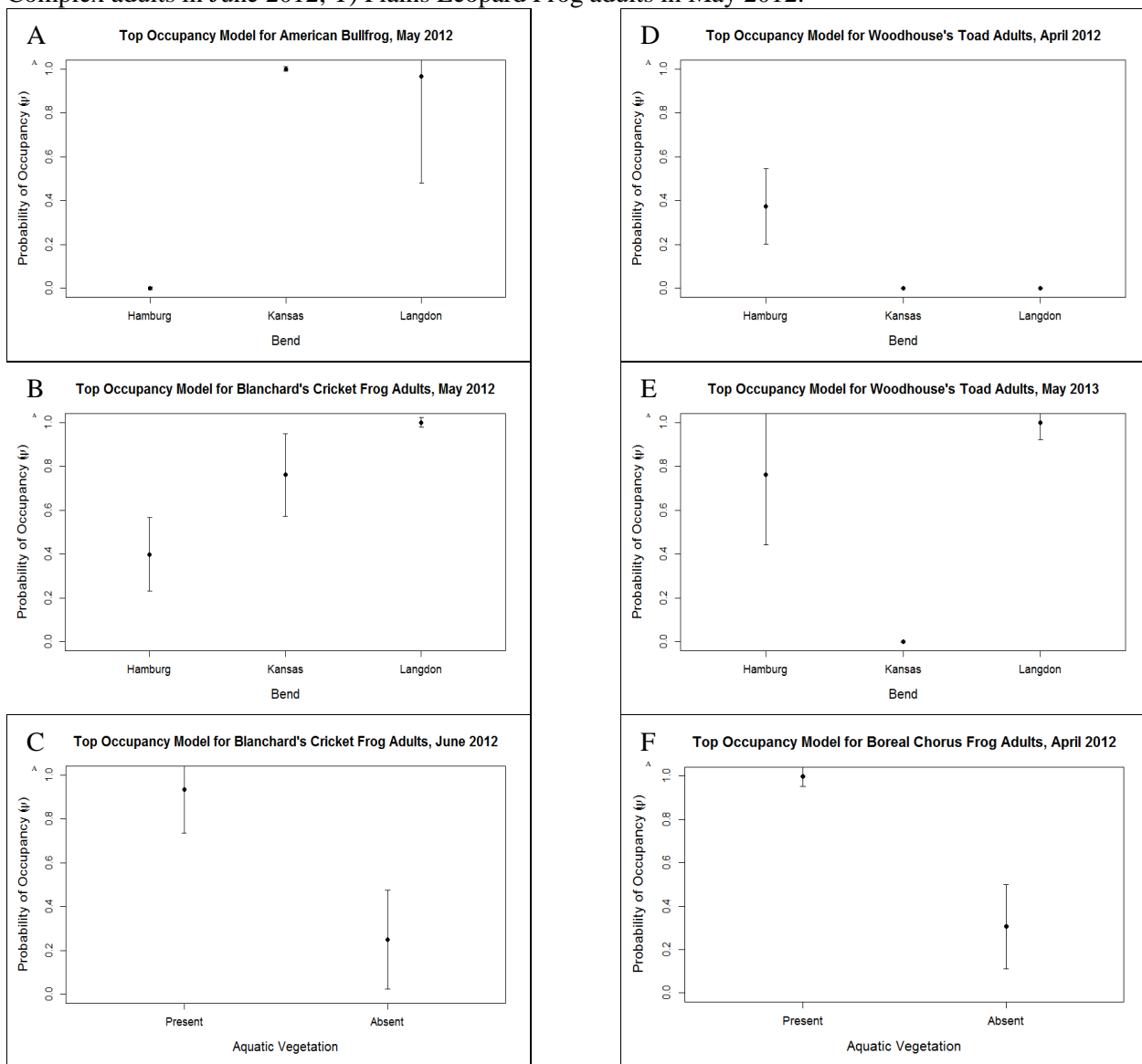
There was sufficient data to assess occupancy models for American Bullfrog adults for all months in 2012 and May and June in 2013. The top model for April 2012 was the null model with AICc weight 0.26 and a confidence set containing six models (Table 2.4, Appendix A: Table 38). Slope was the top model for June 2012 and June 2013 with AICc weights 0.39 and 0.93 respectively. June 2012 had a confidence set of six models, while June 2013 had a confidence set containing two models (Table 2.4, Appendix A: Table 30 and 53). For both June 2012 and June 2013 the parameter estimate was positive and the probability of occupancy increased as the percent slope increased (Appendix A, Figure 2.2). Bend was the top model for May 2012 with AICc weight 0.74 with a confidence set of six models. In May 2012 the American Bullfrog was more likely to occupy Kansas Bend and Langdon Bend and was not present at Hamburg Bend. The top model for May 2013 was distance to the nearest wetland plus land cover with AICc weight 0.2 and the second model was slope with the same AICc weight. The confidence

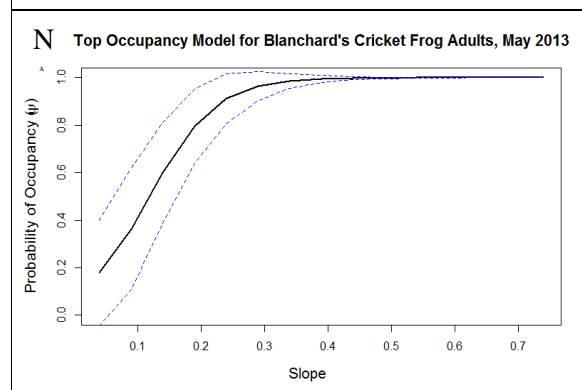
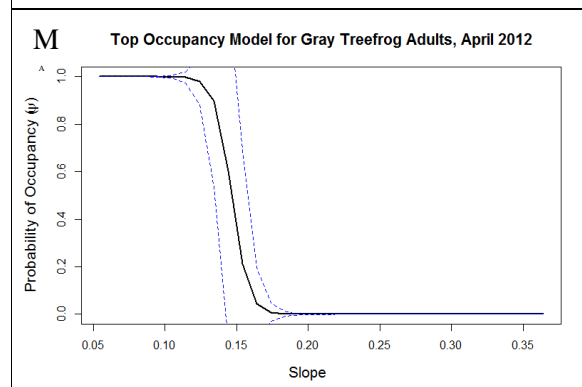
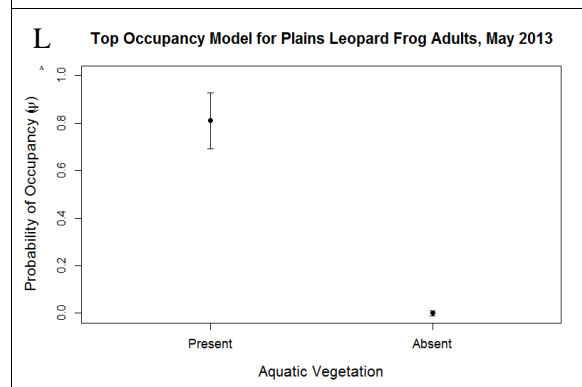
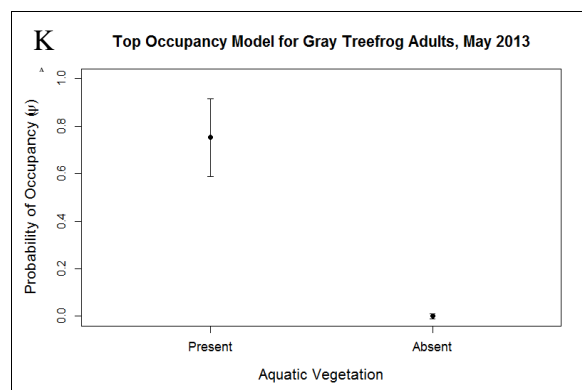
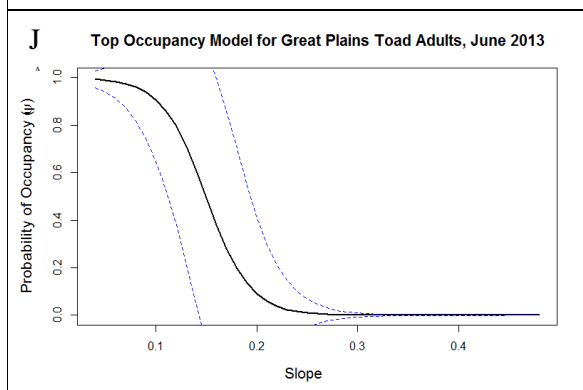
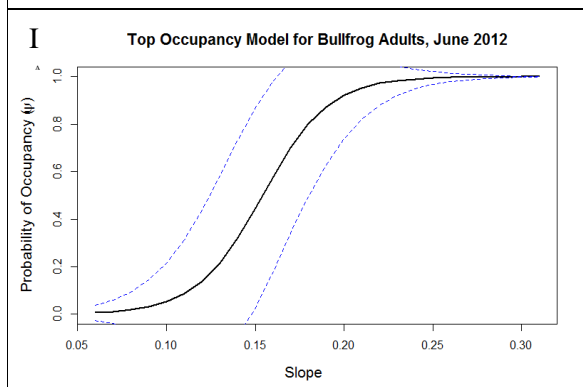
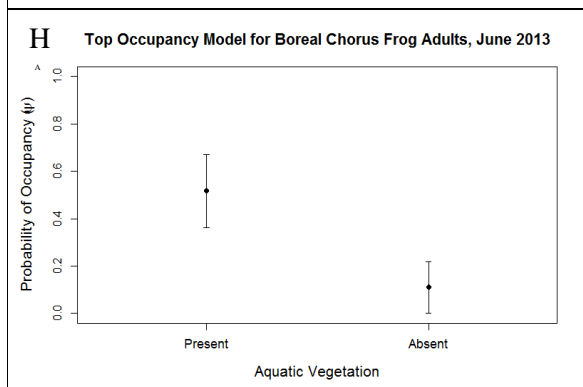
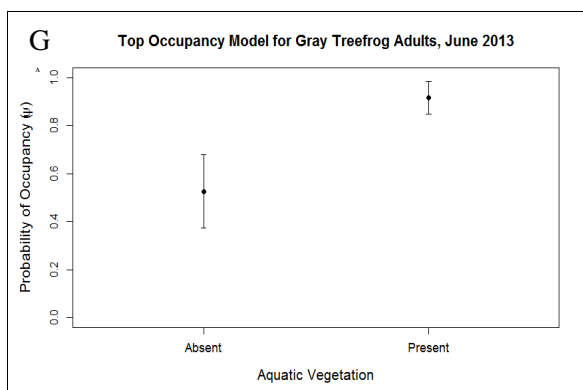
set contained eight models (Table 2.4, Appendix A: Table 42). For the top model parameter estimates were produced, but standard errors were not. Distance to the nearest wetland had a positive parameter estimate and the land cover covariate forest had a positive parameter estimate, while forest had a negative parameter estimate. This suggests that the probability of occupancy was highest when a wetland was isolated and surrounded forest. The model for slope had a positive parameter estimate and so the probability of occupancy increased as the slope percent increased (Appendix A: Table 42).

Boreal Chorus Frog (*Pseudacris maculata*)

There was sufficient data to assess occupancy models for Boreal Chorus Frogs April 2012 and all three months of 2013. Null was the top model twice, for April 2013 and May 2013. April 2013 had a confidence set of seven and an AICc weight of 0.22, while May 2013 had a confidence set of six and an AICc weight of 0.43 (Table 2.4, Appendix A: Tables 54 and 55). The model for aquatic vegetation was the top model for April 2012 with AICc weight 0.43 and June 2013 with AICc weight 0.45. The confidence set for April 2012 had six models and the confidence set for June 2013 had seven models (Table 2.4, Appendix A: Tables 41 and 56). In both cases the parameter estimate was positive and the probability of occupancy was higher when there was aquatic vegetation present at a wetland (Appendix A, Figure 2.2).

Figure 2.2. Graphs of top occupancy model predictions for frog call surveys. The graphs are for A) American Bullfrog adults in May 2012, B) Blanchard's Cricket Frog adults in May 2012, C) Blanchard's Cricket Frog adults in June 2012, D) Woodhouse's Toad adults in April 2012, E) Woodhouse's Toad adults in May 2013, F) Boreal Chorus Frog adults in April 2012, G) Gray Treefrog Complex adults in June 2013, H) Boreal Chorus Frog adults in June 2013, I) American Bullfrog adults in June 2012, J) Great Plains Toad adults in June 2013, K) Gray Treefrog Complex adults in May 2013, L) Plains Leopard Frog adults in May 2013, M) Gray Treefrog Complex adults in April 2012, N) Blanchard's Cricket Frog adults in May 2013, O) Plains Leopard Frog adults in June 2013, P) Plains Leopard Frog adults in June 2012, Q) Woodhouse's Toad adults in June 2013, R) American Bullfrog adults in June 2013, S) Gray Treefrog Complex adults in June 2012, T) Plains Leopard Frog adults in May 2012.





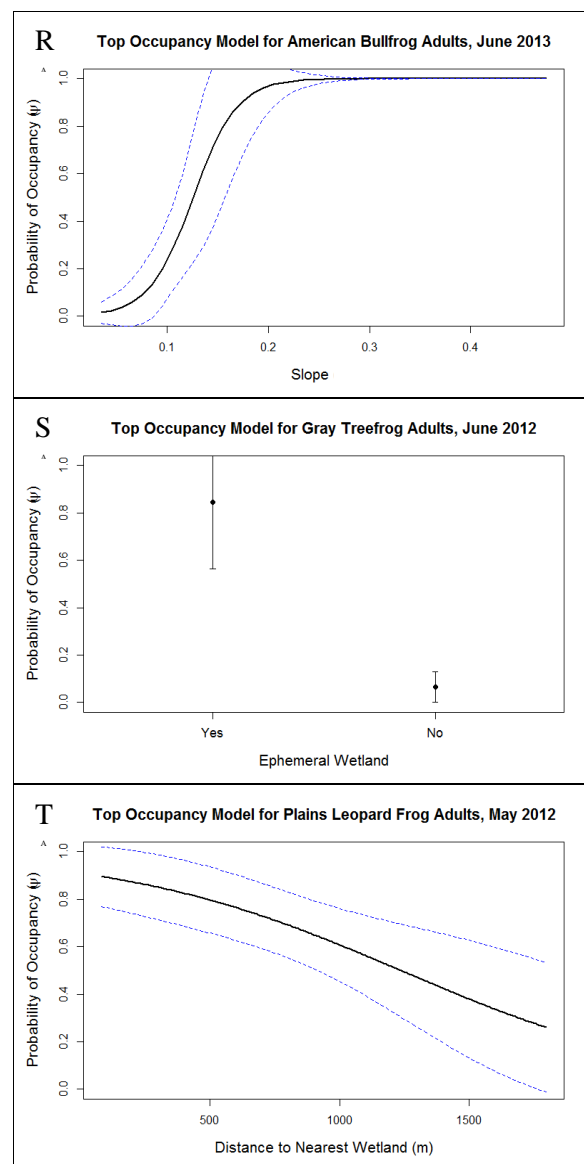
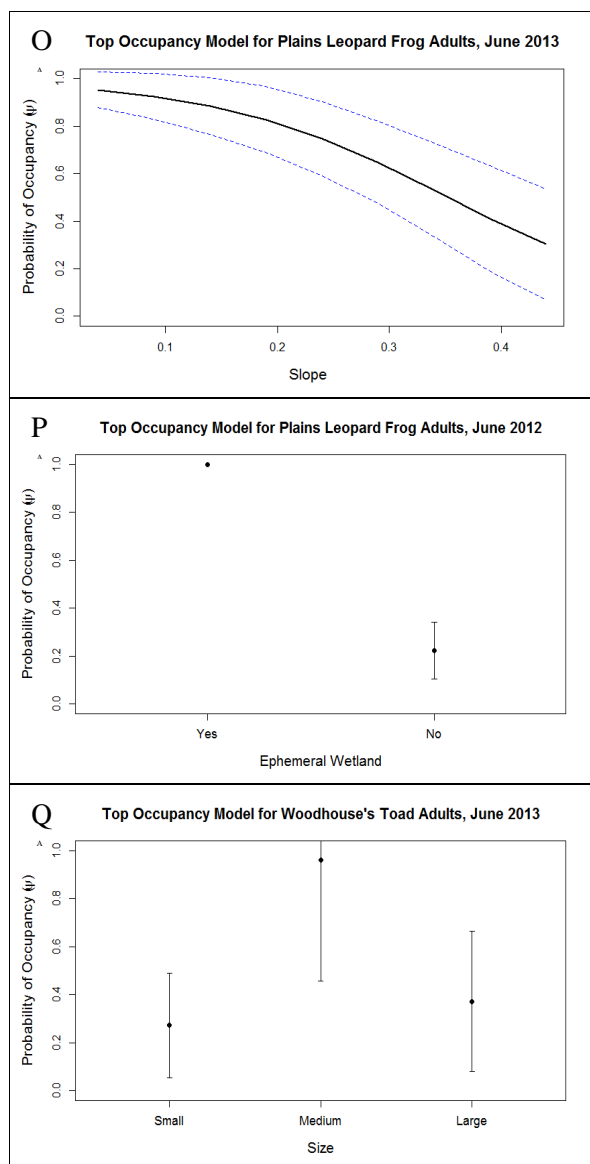


Table 2.4. Top occupancy models for adult anuran species in restored wetlands along the Missouri River in southeast Nebraska. A set of occupancy models was assessed for every species with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top occupancy models contain the measured covariates that affected frog call occupancy the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the occupancy models had a positive or negative relationship with occupancy. For $\hat{\beta}$ D represents distance to the nearest wetland (distwet), A represents agriculture (agr), F represents forest, H represents Hamburg Bend, K represents Kansas Bend, and L represents Langdon Bend. The occupancy estimate ($\hat{\Psi}$) also includes the standard error. For the last column CL refers to confidence limits.

Species	Month	Year	Model	AICcWt	$\hat{\beta}$	$\hat{\Psi}$	95% CL
Blanchard's Cricket Frog	April	2012	$\Psi(\text{distwet}+\text{agr}+\text{forest})$	0.67	D(-)+A(+)+F(-)	0.340 \pm 0.230	0.090-0.730
Blanchard's Cricket Frog	May	2012	$\Psi(\text{bend})$	0.28	H (-), K (+), L (+)	0.999 \pm 0.240	0.000-1.000*
Blanchard's Cricket Frog	June	2012	$\Psi(\text{aquatic})$	0.33	+	0.690 \pm 0.390	0.100-0.980
Woodhouse's Toad	April	2012	$\Psi(\text{bend})$	0.33	H (-), K (-), L (-)	0.000 \pm 0.000*	0.000-1.000*
Gray Treefrog Complex	April	2012	$\Psi(\text{slope})$	0.85	-	0.000 \pm 0.000*	0.000-0.999*
Gray Treefrog Complex	June	2012	$\Psi(\text{ephemeral})$	0.76	+	0.380 \pm 0.290	0.076-0.820
Plains Leopard Frog	April	2012	$\Psi(.)$	0.68	NA	0.830 \pm 0.100	0.600-0.940
Plains Leopard Frog	May	2012	$\Psi(\text{distwet})$	0.31	-	0.630 \pm 0.150	0.380-0.830
Plains Leopard Frog	June	2012	$\Psi(\text{ephemeral})$	0.47	+	1.000 \pm 0.530	0.000-1.000*
American Bullfrog	April	2012	$\Psi(.)$	0.26	NA	0.999 \pm 0.180	0.000-1.000*
American Bullfrog	May	2012	$\Psi(\text{bend})$	0.74	H (-), K (+), L (+)	1.000 \pm 0.003	0.000-1.000*
American Bullfrog	June	2012	$\Psi(\text{slope})$	0.39	+	0.840 \pm 0.280	0.150-0.999
Boreal Chorus Frog	April	2012	$\Psi(\text{aquatic})$	0.43	+	0.940 \pm 0.780	0.000-1.000*
Blanchard's Cricket Frog	May	2013	$\Psi(\text{slope})$	0.49	+	0.999 \pm 0.014	0.670-0.999
Blanchard's Cricket Frog	June	2013	$\Psi(\text{aquatic})$	0.7	+	0.820 \pm 0.220	0.290-0.980
Woodhouse's Toad	May	2013	$\Psi(\text{bend})$	0.33	H (+), K (-), L (+)	0.120 \pm 4.550	0.000-1.000*
Woodhouse's Toad	June	2013	$\Psi(\text{size})$	0.23	H (-), K (+), L (+)	0.860 \pm 0.800	0.000-0.999*
Great Plains Toad	June	2013	$\Psi(\text{slope})$	0.64	-	0.006 \pm 0.041	0.000-0.999*
Gray Treefrog Complex	May	2013	$\Psi(\text{aquatic})$	0.62	+	0.029 \pm 0.510	0.000-1.000*
Gray Treefrog Complex	June	2013	$\Psi(\text{aquatic})$	0.51	+	0.780 \pm 0.095	0.590-0.900
Plains Leopard Frog	April	2013	$\Psi(.)$	0.29	NA	0.330 \pm 0.100	0.180-0.510
Plains Leopard Frog	May	2013	$\Psi(\text{aquatic})$	0.79	+	0.032 \pm 0.580	0.000-1.000*
Plains Leopard Frog	June	2013	$\Psi(\text{slope})$	0.3	-	0.710 \pm 0.160	0.400-0.900
American Bullfrog	May	2013	$\Psi(\text{distwet}+\text{agr}+\text{forest})$	0.2	D(+)+A(-)+F(+)	0.430 \pm 1.250	0.000-0.999*
American Bullfrog	June	2013	$\Psi(\text{slope})$	0.93	+	0.999 \pm 0.012	0.034-1.000
Boreal Chorus Frog	April	2013	$\Psi(.)$	0.22	NA	0.650 \pm 0.130	0.430-0.820
Boreal Chorus Frog	May	2013	$\Psi(.)$	0.43	NA	0.560 \pm 0.140	0.340-0.770
Boreal Chorus Frog	June	2013	$\Psi(\text{aquatic})$	0.45	+	0.270 \pm 0.130	0.110-0.520

*Number is greater than zero but is smaller than 0.0005

DISCUSSION

To determine what a successful wetland is and create the wetland restoration guidelines it is essential to consider the needs of as many amphibian species as possible. None of the species assessed responded in the same way to covariates potentially affecting detection and occupancy of Missouri River wetland. In addition, even with the multiple years, many wetlands, and many species, inference from occupancy modeling was almost always weak. The confidence sets often had many models and so most models had low AICc weights, including many top models. This suggests that frogs are either poor candidates for this statistical approach or that they have a wide tolerance to the wetland environmental conditions, at least within the ranges that they were sampled, or both. However, there were some variables that were consistently among the top models for many species.

The covariates that are most frequent in the model confidence sets for adult anuran detection are important to improving survey techniques. The null model was the top model most often. It was the top model three times more often than the model for water temperature, which was the next most frequent model for detection. This suggests that the detection covariates used in the models do not affect detection, or that inference was too weak to clearly determine covariate effect. If the covariates did not affect detection within the survey techniques that I used, the only way to increase frog call detection is to increase the number of surveys conducted at each wetland within each season (two week period).

While the null model was the top-ranked detection model most often, there were other models that were important to detection for several species. The model for water temperature was the top-ranked model for five species and all had a positive relationship with water temperature. This means that as the water temperature increased the probability of detection increased. Therefore, detection of these species (Figure 2.1 A-E) is going to be higher when call surveys are conducted when the water temperatures are warmer. The best water temperature or threshold water temperature for each species will differ, but the graphs of the detection probabilities (Figure 2.1 A-E) suggest that conducting surveys when the water temperature is 20°C or above will have the highest detection probabilities. The model for air temperature

was the top-ranked model for detection three times (representing three species) and among the confidence sets 14 times. There was a lot of variance in how air temperature affected the probability of detection, but all of the air temperature models in the month of April had positive parameter estimates and all of the air temperature models in June (except for one) had negative parameter estimates. While there could be several explanations for the decrease in calling at the higher air temperatures of June, including the decrease in water in the wetlands (habitat) as the temperature increases, but it is known that anurans have a threshold air temperature that must be reached before they start calling. This will vary from species to species, but conducting frog call surveys on warmer nights in the beginning of the call season will produce better detection rates. In addition, whatever the reason for the negative relationship with temperature in June, detection rates should be higher on cooler nights in June.

The covariate most frequently in model confidence sets for frog occupancy was aquatic vegetation. Overall, the adult anurans were more likely to be present at wetlands with aquatic vegetation. The second most frequent occupancy model was slope. The occupancy probability for the model slope decreased as the percent slope increased for most months, years, and species. The main exception was the American Bullfrog. However, the American Bullfrog is a common and widespread species, invasive in western Nebraska and it is not a priority species for conservation or restoration efforts. The last variable important to many species of adults was whether or not a wetland is ephemeral. For most species ($n=5$), months, and years the probability of occupancy was higher when a wetland was ephemeral. There were other models in the confidence sets that had opposing results for different species or that didn't have enough support to make overall generalizations. In addition, null was a top model five times and among the confidence sets 26 times. The high variance, low inference, and the presence of the null model in most confidence sets, may suggest that amphibian populations need a diversity of wetlands and wetland variables to support a diverse amphibian population. A diversity of wetlands would also support bird and fish populations. The use of shallow, ephemeral wetlands with aquatic vegetation is compatible with management strategies for waterfowl. The deeper tributaries, backwaters, and ground fed, permanent

wetlands, while not ideal or successful, provide habitat for many native fish and in drought conditions they may provide refuges for amphibians (Hellman 2013). Additionally, the model for distance to the nearest wetland was a top model only once, but was among the confidence sets 15 times. For the top model and six others, the parameter estimate was negative and therefore the probability of occupancy decreased as the distance to the nearest wetland increased. This suggests that functional connectivity may also be important to amphibian occupancy.

Therefore, a successful wetland is a wetland that has a shallow slope, contains aquatic vegetation, is ephemeral, and is potentially part of a functionally connected complex with a variety of different wetland types. A more specific idea of what slope percentage to use can be found using the equations made using the parameter estimates (Table 2.5). Picking a representative species with slope as a top model, such as the Gray Treefrog complex, the Great Plains Toad, or the Plains Leopard frog, and then entering the desired occupancy for y and solving for the percent slope (x) (Table 2.5) will provide the recommended slope for restoration. However, the graphs of the top models demonstrate that most species have higher occupancies when the percent slope is less than 0.20. Aquatic vegetation refers to anything that provided habitat in the water to call from, attach eggs to, and use as protection from predators. This includes emergent vegetation like Cattails, Bulrush, and the Common Reed, submerged vegetation like the bladderwort, and woody vegetation like the willow. The flood plains used to flood annually, so natural colonization of plant species is possible. However, the best way to ensure that aquatic vegetation becomes a presence at a wetland is through monitoring and adaptive management. Ideal wetland variety and functional connectivity is largely determined by the landscape itself. However, understanding that wetland success has a landscape level component is the first step. The second is to continue monitoring, analyzing, and learning from the previously restored wetland complexes.

River flood plains are variable and complex systems. Traditionally, flooding occurred annually altering and renewing the landscape. In the future, global climate change is likely to increase extreme conditions, including both flood and drought (IPCC 2003, Erwin 2009). Thus, more monitoring is

needed to increase our understanding of the restored wetlands. However, monitoring data is useless without analysis and the application of the knowledge gained from analysis. Thus, continued longitudinal monitoring in an adaptive management framework is needed to build upon the wetland restoration guidelines. It is our hope that this research will continue and lead to more specific wetland restoration guidelines and the creation of more successfully restored wetlands. Regardless, this study has provided suggestions to increase frog call survey detection rates, a general definition of a successful flood plains wetlands, and equations for individual species that will determine the probability of occupancy for a specific slope percent (Table 2.5). While general, if these guidelines are used they will be far more likely to produce successful flood plains wetlands than the less informed restorations of the past.

Table 2.5. Equations for the top slope and distance to nearest wetland models. Where y is occupancy and x is a distance or percent slope.

Species	Month	Year	Covariate	Equation
Gray Treefrog Complex	April	2012	Slope	$y = -173.60x + 25.50$
American Bullfrog	June	2012	Slope	$y = 53.66x - 8.26$
Blanchard's Cricket Frog	May	2013	Slope	$y = 19.30x - 2.30$
Great Plains Toad	June	2013	Slope	$y = -45.98x + 6.87$
Plains Leopard Frog	June	2013	Slope	$y = -9.60x + 3.39$
American Bullfrog	June	2013	Slope	$y = 46.06x - 5.78$
Plains Leopard Frog	May	2012	Distance to Nearest Wetland	$y = -0.0019x + 2.29$

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CHAPTER 3: DETECTION AND OCCUPANCY OF ANURANS: ADULTS OR TADPOLES?

INTRODUCTION

Wetlands are among the most important and complex ecosystems in the world. Although they cover less than three percent of the earth's surface, they contribute up to 40% of the global annual renewable ecosystem services (Zedler and Kercher 2005). They contribute to nutrient cycling, the hydrologic cycle and provide critical habitat for many plants, fish, and wildlife species (Mitsch and Gosselink 2007). In riverine ecosystems, wetlands aid in creating and maintaining nutrient rich soils. They also help purify water by acting as natural filters and sinks. Additionally, wetlands may contribute to climate stabilization through carbon sequestration (Zedler and Kercher, 2005).

Amphibian monitoring is a method of assessing restored wetlands that is directly related to ecological function. Amphibians are indicators of wetland function because they are declining world-wide (Glascon *et al.* 2005; Pentranka and Holbrook 2006; Collins and Crump 2009; Crump 2010), they are important parts of both the terrestrial and aquatic environments (Collins and Crump 2009; Vitt and Caldwell 2009; Collins and Halliday 2005), and they are good indicators of wetland restoration success (Bowers *et al.* 1998; Collins and Halliday 2005; Welsh and Ollivier 1998). Their biphasic life cycle and permeable skin make them uniquely sensitive to conditions in the water and the surrounding terrestrial environment. Wetlands are complex ecosystems that are hard to define and consist of both aquatic and terrestrial components. This complexity means that these critical ecosystems are easily altered or degraded by changes in either. Thus, amphibians, whose biphasic life cycle and permeable skin makes them dependent on these intricate and essential ecosystems, are ideal indicators of wetland restoration success.

However, there are many ways to monitor amphibians. For larval amphibians examples include box/piper samplers, dip net surveys, seining, leaf litter bags, minnow traps or funnel traps, mark-recapture

using visible implant elastomer (VIE) tagging, and visual encounter surveys. For juvenile and adults examples include mark-recapture using toe clipping, VI-Alpha tags, passive integrated transponder (PIT) tags, VIE, or a variety of other potential marking methods, egg mass and nest counts, call surveys, drift fences, cover boards, funnel traps, pitfall traps, and vertical PVC pipes (Dodd 2010). Prior to the late 1990's monitoring data was mostly used to estimate abundance, birth rates, and survival probabilities. The creation of both the North American Amphibian Monitoring Program and the Amphibian Research and Monitoring Initiative led to the realization that estimating changes in abundance for large areas over time was not possible with the methods being utilized (Mossman *et al.* 1998, MacKenzie *et al.* 2006). The solution was found in measuring the presence or absence of a species at a number of wetlands. However this method also failed to account for detection differences, which, depending on the survey method, can vary by season, day, time of day, location, water temperature, air temperature, and among species (MacKenzie *et al.* 2006).

In 2002, MacKenzie *et al.* presented a method that statistically accounts for varying detection rates (MacKenzie *et al.* 2002). Occupancy modeling consists of conducting two or more surveys at each site (e.g., a wetland) within a specified period of time. It is assumed that the system is closed and that the population (and therefore the probability of detection) does not change during the specified period of time. If a species is found at least once during the time period it is present at that site. However, if it is not found it is either present and undetected or not present. If one, reasonably, assumes that the detection and occupancy are not constant across sites, then it is possible to model the probability of detection and occupancy as a function of measured covariates.

Occupancy techniques for amphibians have almost always been applied to adult surveys of calling, because calling is easy to detect, though variable. However, wetlands may be used by calling adults but not contribute to population viability, because some wetlands may serve as ecological traps in some years, and may induce reproduction but not allow for larval development. In ephemeral wetlands this generally occurs when hydro periods are too short. Thus, to document function, scientists generally

seek to document successful reproduction and / or successive generations of successful reproduction from the same wetland. The latter approach requires expensive and difficult techniques requiring marking, capture, and recapture of large numbers of adults and neonates. Compared to tadpole surveys, frog call surveys are quick, making it easier to gather a large amount of data. Tadpoles are surveyed with dip nets, which is more time consuming, but tadpole surveys provide evidence that a wetland is occupied at all life stages and provides data for non-calling amphibians and intermittent breeders as well as regular breeders. Biologists could potentially use both tadpole and adult surveys, but using both is time consuming. Often, monitoring data is gathered by managers who are time-limited. Thus, it is important to determine if frog call surveys and tadpole dip net surveys are comparable, and where and how detection and occupancy estimates may differ between the two approaches. Here, I report on the comparison of these two techniques, which were tested using data from 55 wetlands across two years.

METHODS

Study Sites

The sites used in this study consist of 55 restored wetlands distributed among three wetland complexes (river bends) between Nebraska City, NE and Nemaha, NE. Hamburg Bend is located 8 miles southeast of Nebraska City, Nebraska in Otoe County. This is the northern most wetland complex and it consists of ~638 ha of former agricultural land that was purchased by the USACE between 1993 and 2004. Hamburg Bend is adjacent to the right descending bank of the Missouri River between river miles 552 to 556. Due to the alterations of the Missouri River Bank Stabilization and Navigation Project (BSNP), the side channels were closed and this allowed the land to accrete. In response a side channel was constructed as well as several back water areas (U.S. Army Corp of Engineers 2006).

Kansas Bend is located 3 miles east of Peru, Nebraska in Nemaha County. This wetland complex is composed of ~427 ha of former agricultural land and is separated into two areas by privately owned

farmland. Kansas Bend was purchased by the Corps between 1993 and 1999 and is adjacent to the right descending bank of the Missouri River between river miles 544 to 547. Due to the activities of the BSNP several side channels were closed using dikes and revetments. Two side channels have been reopened.

Langdon Bend is located 3 miles south of Brownville, Nebraska, in Nemaha County. This is the southernmost wetland complex and it is comprised of ~675 of former agricultural land that was purchased by the USACE between 1994 and 2003. Langdon Bend is the area adjacent to the right descending bank of the Missouri River between river miles 520 and 532. Due to the Missouri River Bank Stabilization and Navigation Project the chute at Langdon Bend was closed and the side channel was cut off. The old channel could not be opened due to its proximity on the upstream end to the Cooper Nuclear Power Plant. Thus, the newly constructed channel is connected to the river at the outlet, but stops before meeting the river at the upstream end. The Corps in conjunction with the Nebraska Game and Parks Commission designed the wetland complex that was built on the west side of the levee. There are about 89 ha of wetlands that were constructed starting in July 2008 and ending May 2009.

Frog Call Surveys

Occupancy data for the adult amphibians was collected using frog call surveys. In 2012 and 2013 each wetland was visited twice within a two week period during April, May, and June. Surveys were started a half an hour after sunset and were concluded by 0100H because most temperate species peak calling hours end around midnight (Dodd 2010). To avoid disturbing the chorus, surveys were conducted 10 m from the shoreline after a two-min acclimation period. We listened and recorded with a hand held recording device (Olympus DM-10) for five min and marked the presence of any species heard calling. The recordings were later listened to by a second observer to decrease listener bias (Lotz and Allen, 2007).

Tadpole Dip net Surveys

Tadpole dip net surveys were conducted to provide occupancy data for species that don't call, such as the Smallmouth Salamander (*Ambystoma texanum*), or species that are rare or breed intermittently

like the Plains Spadefoot (*Spea bombifrons*), and to help determine whether the amphibians at a given wetland were successfully breeding. Dip net surveys were conducted in 2012 and 2013 and each wetland was visited twice within a two-week period during April, May and June. At each wetland I conducted 20 sweeps of the dip-net every 10 m for a total coverage of 200 m of wetland shoreline. Following each sweep of the dip-net, captured individuals were identified and the presence of all species found was recorded. The tadpoles for both potential toad species (Woodhouse's Toad and Great Plains Toad) were combined into one category because it is difficult to identify the difference between them at that stage.

Covariates

Both survey specific covariates and site specific covariates were collected for the frog call surveys and the tadpole dipnet surveys. The survey specific covariates time of day, day of year, air temperature, water temperature, wind speed (m/s), water pH, relative humidity, were collected for every frog call survey and tadpole dip net survey. The survey specific covariates moon phase (percent) and cloud cover (percent) were only collected for the frog call surveys. Moon phase percent was determined from the United States Naval Meteorology and Oceanography Command website (<http://aa.usno.navy.mil/data/docs/MoonFraction.php>). Percent cloud cover was visually estimated. Moon phase and percent cloud cover were combined into a covariate called moonshine by first subtracting the percent cloud cover from one and then multiplying this by the moon phase percent.

Three of the site specific covariates, aquatic vegetation, adjacent vegetation, and the shallow water slope, were collected during tadpole dip-net surveys during the daytime. For each sweep, in addition to the presence of tadpoles, I determined whether the net was drawn through herbaceous, woody, or open water and a water height measurement was taken at 0.5 m and 1 m from the shoreline. Using all 20 net sweeps I then calculated the percentage of herbaceous, woody, and open water habitat along the 200 m surveyed. In addition, a visual survey of the entire wetland was done and aquatic vegetation was marked as present or absent. Aquatic vegetation was determined as present if it was present in the visual survey or if the percent of herbaceous and woody vegetation was $\geq 10\%$. A visual survey was also used

for adjacent vegetation. We determined the percent grass, herbs and forbs, trees and shrubs, or bare ground within 1 m of the waterline. Shallow water slope was calculated using the water height measurement at 1 m and dividing by 100 cm. Two covariates, whether a wetland is ephemeral or not and wetland size, were determined at the beginning of the monitoring activity in 2009. Wetland size was recorded as small (≤ 2 ha), medium (2-5 ha), and large (> 5 ha). Land cover and the distance to the nearest wetland were calculated using ArcGIS. Land cover was determined by creating a 1000 m buffer around each wetland and determining the percentage of grass, forest, and agriculture within that buffer, and assigning land cover type to the dominant (%) land cover class. The distance to the nearest wetland was determined from digital land cover maps, and calculated from centroid to centroid.

Analysis

An occupancy modeling analysis was conducted using the package *unmarked* in program R (R Development Core Team 2008; Fiske and Chandler 2011). Occupancy modeling represents a major improvement to amphibian monitoring techniques because it accounts for differences in detection, which may vary across time, across wetlands, and across species. As manifest in Program Presence (Hines 2006) and in the package *unmarked* in program R, I used an occupancy model structured multi-model inference to select models that best fit the observed data. Like all implementations of multi-model inference, model selection is critical to sound inference. Model sets were created *a priori* and were run as single species - single season models, using month as season. A separate model set was created for tadpole detection (9 models), adult detection (12 models), and a single occupancy model set for both the adults and tadpoles (12 models) (Table 2.1 and Table 2.2). Some of the models in each model set correlated, but they were all included because they have different management implications. Unless the naïve occupancy (number of wetlands that a species was present at least once/total number of wetlands) was < 0.10 a model set was run for every species, each month (April, May, and June), both years (2012 and 2013), and for both tadpoles and adults. This cutoff was chosen because species with low detection have uncertain occupancy and most model sets below this naïve occupancy would not run in program R

(Hellman 2013). Detection was assessed separately from occupancy. Detection was determined first using the adult detection model set. The occupancy portion of each model was held constant using global occupancy. Then, the covariate/s in the detection portion of the top model for each detection model set was used to hold detection constant while running the occupancy model sets. Due to drought in 2012 and the beginning of 2013, many of the wetlands dried up and both adult and tadpole occupancy was affected. This increased the number of models and variables in the global occupancy model that failed to converge. I removed variables from the global model if convergence was not achieved and any models that did not converge were removed from the model sets.

To determine what variables affect detection and occupancy, a confidence set accounting for 95% of overall model weight was considered (Burnham and Anderson 2002). The 95% confidence set was calculated by adding up the models AICc weight until it was greater than or equal to 0.95. A large confidence set was used to ensure that no important parameters were excluded. As previously mentioned, my research is part of a multi-state monitoring project that is set up to function in an adaptive management framework. If continued, an inclusive confidence set allows for more confident model set refinement. Within the confidence set the AICc weight was used to determine the model's fit. Parameter estimates were used to determine what affect a covariate/s had on detection or occupancy. In this case, a positive parameter estimate for a continuous variable like slope or distance to the nearest wetland meant that the detection or occupancy increased as the covariate/s increased. A positive parameter estimate for a categorical variable like aquatic vegetation, bend, or size meant that occupancy or detection was higher with/at that covariate than without/not at it. For continuous variables a negative parameter estimate meant that detection or occupancy was decreasing as the covariate/s increased. For categorical variables a negative parameter estimate meant that occupancy or detection was lower with/at that covariate than without/not at it. The covariates in the top models were graphed to show how detection or occupancy was affected by that particular covariate.

Table 3.1. Detection models for amphibian larvae in restored wetlands along the Missouri River in Southeast Nebraska. Tadpole and covariate data was collected during April, May, and June of 2012 and 2013. Detection is on the right represented by p , and occupancy is on the left represented by Ψ . Detection was assessed first, using the global occupancy to hold occupancy constant. The global occupancy is the additive combination of all occupancy covariates (bend, wetland size, slope, aquatic vegetation, adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest). The top detection models held the covariates that were most likely to have affected adult anuran detection. The detection covariate/s in the top detection model was used to hold detection constant when the occupancy was assessed using a second set of models (Table 3.3). The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(\text{Global Covariates}) p(1)$
Time	$\Psi(\text{Global Covariates}) p(\text{time})$
Day (julian date)	$\Psi(\text{Global Covariates}) p(\text{julian})$
Slope	$\Psi(\text{Global Covariates}) p(\text{slope})$
Aquatic Vegetation	$\Psi(\text{Global Covariates}) p(\text{aquatic})$
Water Temperature	$\Psi(\text{Global Covariates}) p(\text{H2Otemp})$
Ephemeral Wetland	$\Psi(\text{Global Covariates}) p(\text{ephemeral})$
Water Temperature + Slope	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+slope})$
Global	$\Psi(\text{Global Covariates}) p(\text{time+julian+slope+aquatic+H2Otemp+ephemeral})$

Table 3.2. Detection models for adult anurans species in restored wetlands along the Missouri River in southeast Nebraska. Frog call and covariate data was collected during April, May, and June of 2012 and 2013. Detection is represented by p , and occupancy is represented by Ψ . Detection was assessed first, using the global occupancy to hold occupancy constant. The global occupancy is the additive combination of all occupancy covariates (bend, wetland size, slope, aquatic vegetation, adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest). The top detection models held the covariates that were most likely to have affected adult anuran detection. The detection covariate/s in the top detection model was used to hold detection constant when the occupancy was assessed using a second set of models (Table 2.2). The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(\text{Global Covariates}) p(.)$
Time	$\Psi(\text{Global Covariates}) p(\text{time})$
Wind (m/s)	$\Psi(\text{Global Covariates}) p(\text{wind})$
Day (julian date)	$\Psi(\text{Global Covariates}) p(\text{julian})$
Wetland Size	$\Psi(\text{Global Covariates}) p(\text{size})$
Moonshine	$\Psi(\text{Global Covariates}) p(\text{moonshine})$
Air Temperature	$\Psi(\text{Global Covariates}) p(\text{airtemp})$
Water Temperature	$\Psi(\text{Global Covariates}) p(\text{H2Otemp})$
Environmental Covariates 1	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind})$
Environmental Covariates + Day	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+julian})$
Environmental Covariates 2	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+moonshine})$
Global	$\Psi(\text{Global Covariates}) p(\text{time+wind+julian+size+moonshine+airtemp+H2Otemp})$

Table 3.3. Occupancy models for both adult and larval amphibians in restored wetlands along the Missouri River in Southeast Nebraska. Frog call, tadpole dip net, and covariate data was collected during April, May, and June of 2012 and 2013. Detection is on the right represented by p , and occupancy is on the left represented by Ψ . Occupancy was assessed after detection and used the detection covariate/s in the top detection model to hold detection constant. The top detection models held the covariates that were most likely to have affected adult anuran detection. The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(1) p(\text{Top Detection Covariate/s})$
Bend	$\Psi(\text{bend}) p(\text{Top Detection Covariate/s})$
Wetland Size	$\Psi(\text{size}) p(\text{Top Detection Covariate/s})$
Slope	$\Psi(\text{slope}) p(\text{Top Detection Covariate/s})$
Aquatic Vegetation	$\Psi(\text{aquatic}) p(\text{Top Detection Covariate/s})$
Adjacent Vegetation	$\Psi(\text{adjacent}) p(\text{Top Detection Covariate/s})$
Ephemeral Wetland	$\Psi(\text{ephemeral}) p(\text{Top Detection Covariate/s})$
Distance to Nearest Wetland	$\Psi(\text{distwet}) p(\text{Top Detection Covariate/s})$
Land Cover	$\Psi(\text{agr+forest}) p(\text{Top Detection Covariate/s})$
Wetland Vegetation	$\Psi(\text{aquatic+adjacent}) p(\text{Top Detection Covariate/s})$
Connectivity	$\Psi(\text{distwet+agr+forest}) p(\text{Top Detection Covariate/s})$
Global	$\Psi(\text{bend+size+slope+aquatic+adjacent+ephemeral+distwet+agr+forest}) p(\text{Top Detection Covariate/s})$

RESULTS

Tadpole Survey Detection

The number of wetlands holding water varied from month to month during 2012 and 2013. Annual variation in flood plain wetlands is typical; however the variation in 2012 and 2013 was extreme. In 2011 the Missouri River experienced historic flooding due to both higher than average snowpack melt and higher than average amounts of rainfall during the spring and summer months. This was followed by a historic drought in 2012 that continued into 2013. The drought was caused by record high temperatures combined with below average snowpack melt and precipitation. From mid-April until the beginning of May 2012 the water table was still high enough that average rainfall caused an increase in the number of wetlands holding water. However, from that point on the high temperatures continued to lower the number of wetlands that were holding water. Southeast Nebraska received average to above average rainfall in March and April of 2013, but it wasn't until extreme rainfall events in late May of 2013 that the soil was saturated enough for most of the wetlands to hold water. Thus, tadpole dip net surveys were conducted at 37 wetlands in April 2012, 28 wetlands in May 2012, 21 wetlands in June 2012, 21 wetlands in April 2013, 22 wetlands in May 2013, and 39 wetlands in June 2013.

There were 9 models in the detection model set for larval amphibians (Table 3.1). Ideally, I would have modeled all seven species, during all months (April, May, and June), and for both years, producing 42 different model sets. However, none of the species found had naïve occupancy high enough to be modeled for all three months and both years. Thus, the models were applied to 13 different sets of data and there were 13 different top models and confidence sets. The confidence sets contained 3 to 6 models (Appendix B). Null was the top model 8 of the 13 models and was in the confidence set 12 times (Table 3.4, Appendix B). This suggests that within the methodological constraints that the data was surveyed, the covariates measured are not affecting the detection of the larval amphibians. Below I address those covariates that were in the confidence set as well as individual species.

Slope

The model for slope was the top detection model three times and among the confidence sets 13 times (Table 3.4, Appendix B). The effect of slope on detection was mixed. Two of the top models had negative parameter estimates and so the detection probability decreased as the slope percent increased, and one had a positive parameter estimate which showed the opposite (Figure 3.1). The top model for the Toad species tadpoles did not have enough inference to produce standard errors for the model (Appendix B). Only three of the models in the confidence sets had negative parameter estimates, not including the model for Toad species tadpoles. This indicates that a steep slope is probably important for most species. However, two of the three that did have negative parameter estimates were the only set run for Blanchard's Cricket Frog tadpoles in June 2013, and the top model for Boreal Chorus Frog tadpoles in June 2013. It is possible that a shallow slope is more important for smaller tadpole species (Table 3.4, Appendix B).

Time

Time was the top model once and was among the detection confidence set nine times (Table 3.4, Appendix B). Six of the models in the confidence sets, including the top model for Boreal Chorus Frog tadpoles April 2012, had positive parameter estimates, while three had negative parameter estimates. Two of the three models with negative parameter estimates were the last model in the confidence set. There is not strong enough evidence to make a statement overall, but enough to suggest that detection has a positive relationship with time.

Aquatic Vegetation

Aquatic vegetation was the top model once and was among the detection confidence sets eight times (Table 3.4, Appendix B). In addition to the top model for Blanchard's Cricket Frog June 2013, five of the eight models among the confidence sets had negative parameter estimates. A negative parameter estimate indicates that the tadpoles were less likely to be detected when there wasn't aquatic vegetation at

a wetland (Figure 3.1, Appendix B). Again, there is not strong enough evidence to make a statement overall, but this does suggest tadpole detection has a negative relationship with aquatic vegetation.

Other Covariates

The models for ephemeral, water temperature, and water temperature plus slope were never top models but were still among the detection confidence sets. Ephemeral was among the confidence sets nine times with three models with positive parameter estimates, three with negative parameter estimates, and three with inference too low to obtain parameter estimates. Water temperature was among the confidence sets seven times and five of the models had negative parameter estimates. This suggests that as water temperature increased the detection probability decreased, but there is not strong enough evidence to make an overall statement. The model for water temperature plus slope was among the model sets twice. In both instances the water temperature had a negative parameter estimate, but the slope had opposite results. The models for day (julian date) and the global model were not even among the confidence sets (Appendix B).

Blanchard's Cricket Frog (*Acris blanchardi*)

There was sufficient data to assess detection models for Blanchard's Cricket Frog in June 2013. No Blanchard's Cricket Frogs were found in 2012. There were five models in the confidence set and the top model was aquatic vegetation with AICc weight 0.32 (Table 3.4, Appendix B: Table 6). The model of aquatic vegetation had a negative parameter estimate and showed that the probability of detection was higher when a wetland did not have aquatic vegetation (Figure 3.1). The model for whether or not a wetland was ephemeral showed the probability of detection to be higher when a wetland was not ephemeral.

Toad Species (*Anaxyrus spp.*)

There was sufficient data to assess detection models for Toad tadpoles in May 2013 alone. There were three models in the confidence set and the top model was slope with AICc weight 0.70 (Table

3.4, Appendix B). The parameter estimate was negative, but there was not enough inference to produce standard errors (Appendix B, Figure 3.1).

Smallmouth Salamanders (*Ambystoma texanum*)

There was sufficient data to assess detection models for Smallmouth Salamander in May 2013. No Smallmouth Salamander Larvae were found in 2012. There were five models in the confidence sets and the top model was slope with AICc weight 0.35 (Table 3.4, Appendix B: Table 7). The model for slope had a positive parameter estimate therefore, as the slope percentage increased the probability of detection increased as well (Figure 3.1, Appendix B). This is likely because the wetland where a majority of the larvae were found was a very deep hole directly adjacent to upland forest.

Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*)

There was sufficient data to assess detection models for Gray Treefrog tadpoles in June 2013 alone. There were six models in the confidence set and the top model was null with AICc weight 0.31 (Table 3.4, Appendix B: Table 9). Null is followed by time, ephemeral, water temperature, aquatic vegetation, and slope with AICc weights between 0.23 and 0.06 (Appendix B: Table 9). It is possible that all of these have an effect on detection but given that they fall after null and are so closely grouped it is unlikely that any of them have an effect.

Plains Leopard Frog (*Lithobates blairi*)

There was sufficient data to assess detection models for Plains Leopard Frog tadpoles in May and June 2013. Null is the top model for both May and June 2013 with AICc weights 0.73 and 0.42 (Table 3.4). There were five models in the confidence set for May 2013 and six models in the confidence set for June 2013. In May 2013 null holds the majority of the weight, but in June 2013 null is followed by aquatic vegetation, water temperature, ephemeral, time, and slope with AICc weights ranging from 0.29 to 0.04. It is possible that one or more of these affected detection, but given that they follow null it is unlikely (Table 3.4, Appendix B: Table 10-11).

American Bullfrog (*Lithobates catesbeianus*)

There was sufficient data to assess detection models for American Bullfrog tadpoles in all months of 2012. Due to drought conditions in 2012 that caused most of the wetland sites to dry up, very few Bullfrog tadpoles survived the winter. Therefore very few American Bullfrog tadpoles were found in 2013. All three months for 2012 had the null model as the top model with AICc weights 0.43, 0.75, and 0.55. There were six models in the confidence set for April 2012, four models in the confidence set for May 2012, and five models in the confidence set for June 2013 (Table 3.4, Appendix B: 1-3).

Boreal Chorus Frog (*Pseudacris maculata*)

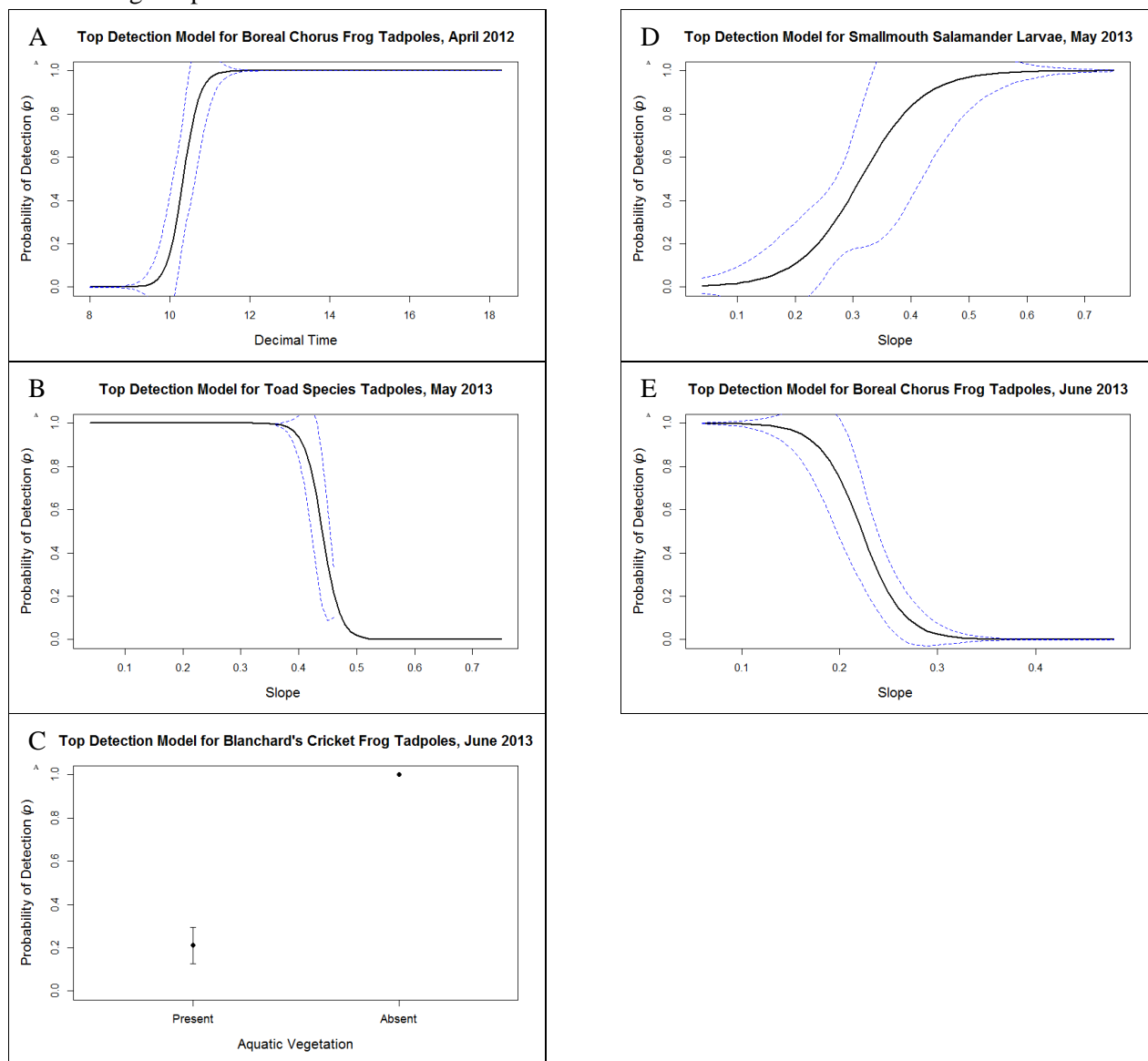
There was sufficient data to assess detection models for Boreal Chorus Frog tadpoles in April and June 2012 and May and June 2013. For June 2012 and May 2013 null was the top model with AICc weights 0.75 and 0.45 respectively. The confidence set for June 2012 had five models and the confidence set for May 2013 had four models (Table 3.4, Appendix B: Tables 4-5, and 12-13). For April 2012 the top model was time with AICc weight 0.42 and there were four models in the confidence set (Table 3.4, Appendix B). Slope was the top model for June 2013 with AICc weight 0.74, with only two models in the confidence set. The second was slope plus water temperature. (Table 3.4, Appendix B: Table 13). The parameter estimate for slope was negative and the detection probability decreased as the slope percentage increased (Figure 3.1).

Table 3.4. Top detection models for larval amphibian species in restored wetlands along the Missouri River in southeast Nebraska. A set of detection models was assessed for every species with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top detection models contain the measured covariates that affected tadpole dip net detection the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the detection models had a positive or negative relationship with detection. The detection estimate (\hat{p}) also includes the standard error.

Species	Month	Year	Model	AICcWt	$\hat{\beta}$	\hat{p}	95% CL
American Bullfrog	April	2012	$p(\cdot)$	0.43	NA	0.840 \pm 0.094	0.620-0.940
American Bullfrog	May	2012	$p(\cdot)$	0.75	NA	1.000 \pm 0.000+	0.000-1.000*
American Bullfrog	June	2012	$p(\cdot)$	0.55	NA	0.880 \pm 0.120	0.540-0.980
Boreal Chorus Frog	April	2012	$p(\text{time})$	0.42	+	1.000 \pm 0.000*	0.000-1.000*
Boreal Chorus Frog	June	2012	$p(\cdot)$	0.75	NA	0.390 \pm 0.870	0.260-0.860
Blanchard's Cricket Frog	June	2013	$p(\text{aquatic})$	0.32	-	1.000 \pm 0.000*	1.000-1.000
Smallmouth Salamander	May	2013	$p(\text{slope})$	0.35	+	0.980 \pm 0.024	0.850-1.000
Toad Species	May	2013	$p(\text{slope})$	0.7	-	1.000 \pm NA	NA
Gray Treefrog Complex	June	2013	$p(\cdot)$	0.31	NA	0.940 \pm 0.060	0.730-0.999
Plains Leopard Frog	May	2013	$p(\cdot)$	0.73	NA	0.840 \pm 0.120	0.560-0.960
Plains Leopard Frog	June	2013	$p(\cdot)$	0.42	NA	0.810 \pm 0.094	0.610-0.920
Boreal Chorus Frog	May	2013	$p(\cdot)$	0.45	NA	0.800 \pm 0.130	0.520-0.940
Boreal Chorus Frog	June	2013	$p(\text{slope})$	0.74	-	0.096 \pm 0.110	0.013-0.470

*Number is greater than zero but is smaller than 0.0005

Figure 3.1. Graphs of top detection model predictions for tadpole dip net surveys. The graphs are for A) Boreal Chorus Frog tadpoles in April 2012, B) Toad species tadpoles in May 2013, C) Blanchard's Cricket Frog tadpoles in June 2013, D) Smallmouth Salamander larvae in May 2013, and E) Boreal Chorus Frog Tadpoles in June 2013.



Tadpole Survey Occupancy

There were 12 models in the occupancy model set for adult anurans (Table 3.2). Ideally, these would have been run for all seven species, all months of April, May, and June, and both years 2012 and 2013, for a total of 42 potential model sets. However, none of the species found had naïve occupancy high enough to be run for all three months and both years. Thus, the models were applied to 13 different sets of data and there were 13 top models and confidence sets. The confidence sets contained 7 to 10 models (Appendix B). Overall, tadpole occupancy was affected by distance to the nearest wetland, bend, and aquatic vegetation. Null was the top model 2 of the 13 model runs and was in the confidence set 13 times (Table 3.5, Appendix B). Below I address those covariates that were in the confidence set as well as individual species.

Distance to the Nearest Wetland

The model for distance to the nearest wetland was the top model four times and was among the occupancy confidence sets 11 times. Eight of the 11 models within the confidence sets, including the four top models, had negative parameter estimates. Out of the three remaining models two had positive parameter estimates and one did not have enough data to produce a parameter estimate. The top models represent three different species and the negative parameter estimates indicate that the probability of occupancy decreases as the distance increases (Table 3.5, Appendix B, Figure 3.2).

Bend

The model for bend was the top occupancy model three times and among the confidence sets eight times. The top models represented three species, two of which were relative rare. These are the Smallmouth Salamander larvae and Toad species tadpoles. The Smallmouth Salamander is a species of concern in Nebraska and while Toad species aren't a concern, they were uncommon in the wetlands I surveyed (naïve occupancy > 0.10 in May of 2013 only). Both of these species were only found in Langdon Bend (Table 3.5, Appendix B, Figure 3.2).

Aquatic Vegetation

Aquatic Vegetation was the top model twice and was among the occupancy confidence sets thirteen times. The top models were for the American Bullfrog and the Boreal Chorus Frog in April of 2012 (Table 3.5, Appendix B). The April 2012 model for the American Bullfrog had a negative parameter estimate and so the probability of occupancy was highest when there was no aquatic vegetation, while the April 2012 model for the Boreal Chorus Frog had a positive parameter estimate and had a higher probability of occupancy when aquatic vegetation was present (Appendix B, Figure 3.2). While this seems confusing, ten of the thirteen aquatic vegetation models in the confidence sets had positive parameter estimates (Appendix B). This suggests that most species of amphibian larvae have a higher probability of occupancy when aquatic vegetation is present at a wetland.

Ephemeral

Ephemeral was a top occupancy model once and among the model confidence sets twelve times (Table 3.5, Appendix B). Of those twelve times, it had a positive parameter estimate six times and a negative parameter estimate six times. However, three of the six negative parameter estimates belonged to the only three model sets for the American Bullfrog. One of the other negative parameter estimates belonged to the only confidence set for the Smallmouth Salamander. The wetland that these were found in most often was a permanent wetland with an extremely steep slope. The top model was for the American Bullfrog in May 2012 and it had a negative parameter estimate. This means that the probability of occupancy was highest when the wetlands were not ephemeral (Appendix B, Figure 3.2). This suggests that for most species occupancy is higher when the wetland is ephemeral, but there is not enough data to make a statement overall.

Slope

Slope was the top model once and among the occupancy confidence sets 11 times. The top model was for the Plains Leopard Frog tadpoles in June 2013. The parameter estimate for this model was negative and the probability of occupancy decreased as the slope percent increased (Appendix B, figure

3.2). Out of the 11 slope models in the confidence sets, six had negative parameter estimates, four had positive parameter estimates, and one did not have enough inference to produce a parameter estimate (Appendix B). This suggests that wetlands with a shallow slope have higher occupancy, but there is not enough data to make a statement overall.

Other Covariates

The models of adjacent vegetation, size, land cover, aquatic vegetation plus adjacent vegetation, and distance to nearest wetland plus land cover were never top models. Adjacent vegetation was among the occupancy confidence sets three times, size was among the confidence sets 11 times, land cover was among the confidence sets six times, aquatic vegetation plus adjacent vegetation was among the confidence sets once, and distance to the nearest wetland plus land cover was among the confidence sets six times. All of these contain multiple variables and there is not enough data to make a statement overall. The only model that was never among the confidence sets was the global model (Appendix B).

Blanchard's Cricket Frog (*Acris blanchardi*)

There was sufficient data to assess occupancy models for Blanchard's Cricket Frog tadpoles in June 2013 alone. No Blanchard's Cricket Frog tadpoles were found in 2012. There were seven models in the confidence set and the top model was distance to the nearest wetland with AICc weight 0.28 (Table 3.5, Appendix B: Table 19). The parameter estimate was negative and therefore the probability of occupancy for the distance to the nearest wetland decreased slightly as the distance increased (Appendix B, Figure 3.2). Given the high number of models in the confidence set and the low AICc weight of the top model, it is possible that some of the other models affected the occupancy. The models in the confidence set are size, null, slope, ephemeral, aquatic vegetation, and distance to the nearest wetland plus land cover and the AICc weights range from 0.21 to 0.02.

Toad Species (*Anaxyrus spp.*)

There was sufficient data to assess occupancy models for Toad tadpoles in May 2013 alone. Only a few toad tadpoles were found in 2012. There were seven models in the confidence set and the top model was bend with AICc weight 0.41 (Appendix B: Table 21, Table 3.5). The model for bend only had a positive parameter estimate for Langdon Bend, showing that they only occurred in Langdon Bend (Appendix B: Table 21, Figure 3.2). The second model in the confidence set was slope with AICc weight 0.33. The rest of the models had AICc weights between 0.02 and 0.08. It is possible that slope may also affect occupancy for toad species, but it is unlikely that the rest have any affect (Appendix B: Table 21).

Smallmouth Salamanders (*Ambystoma texanum*)

There was sufficient data to assess occupancy models for Smallmouth Salamander larvae in May 2013. No Smallmouth Salamander larvae were found in 2012. There were seven models in the confidence set and Bend was the top model with AICc weight 0.45 (Appendix B: Table 20, Table 3.5). The model for Bend only had a positive parameter estimate for Langdon Bend, indicating that the larvae were only found in Langdon Bend (Appendix B: Table 20). The second model in the confidence set was the null model with AICc weight 0.25. The rest of the models had AICc weights between 0.04 and 0.07 (Appendix B: Table 20). Given the low AICc weight they are unlikely to affect occupancy.

Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*)

There was sufficient data to assess occupancy models for Gray Treefrog tadpoles in June 2013 alone. Gray Treefrog tadpoles were only found in one wetland in 2012. There were eight models in the confidence set and the top model was distance to nearest wetland with AICc weight 0.35. The parameter estimate was negative and the probability of occupancy decreased as the distance increased (Table 3.5, Appendix B: Table 22, Figure 3.2). The second model in the confidence set was slope with AICc weight 0.29, followed by null with AICc weight 0.12. The model for slope had a negative parameter estimate which indicates that the occupancy decreased as the slope percent increased. The rest of the models had

AICc weight between 0.02 and 0.08. Given the low weight and the fact that they follow the null model it is unlikely these model affected occupancy (Appendix B: Table 22).

Plains Leopard Frog (*Lithobates blairi*)

There was sufficient data to assess occupancy models for Plains Leopard Frog tadpoles in May and June 2013. The confidence set for May 2013 had seven models and the top model was null with AICc weight 0.34 (Table 3.5, Appendix B: Table 23). The rest of the models had AICc weights from 0.25 to 0.03, and given that they fall after the null model it is unlikely that they affect occupancy. The confidence set for June 2013 had six models and the top model was slope with AICc weight 0.58 (Appendix B: Table 24, Table 3.5). The parameter estimate for the slope model was negative and thus the probability of occupancy decreased as the percent slope increased (Appendix B: Table 24, Figure 3.2). The second model is bend with AICc weight 0.23. The parameter estimate for Hamburg Bend is positive while the other two bends are negative, meaning that Plains Leopard Frog tadpoles were more likely to be found at Hamburg Bend (Appendix B: Table 24). The last four models had AICc weight between 0.02 and 0.06 and it is unlikely they had any effect on occupancy.

American Bullfrog (*Lithobates catesbeianus*)

There was sufficient data to assess occupancy models for American Bullfrog tadpoles in all three months in 2012. The confidence set for April 2012 had ten models and the top model was aquatic vegetation with AICc weight 0.23 (Appendix B: Table 14, Table 3.5). The parameter estimate was negative and so the probability of occupancy was slightly higher when aquatic vegetation was absent in a wetland (Figure 3.2). The second model is the null model with AICc weight 0.13 and the rest of the models have AICc weights between that and 0.04. It is unlikely that any of these models, including the top model, had a strong effect on occupancy (Appendix B: Table 14). The confidence set for May 2013 had seven models and the top model was ephemeral with AICc weight of 0.29 (Appendix B: Table 15, Table 3.5). The parameter estimate for this model is negative indicating that the probability of occupancy was higher in wetlands that were not ephemeral (Appendix B: Table 15, Figure 3.2). This follows with

the life history of American Bullfrog, because their tadpoles overwinter and would die in a wetland that doesn't hold water all year long. The second model in this confidence set was also the null model with AICc weight 0.26 and the rest of the model range between this and 0.03 (Appendix B: Table 15). As with the models in the April confidence set it is unlikely that any of these models had a strong effect on occupancy. The confidence set for April 2012 had eight models and the top model was the null model with 0.26. This again suggests that none of these models had an effect on occupancy (Appendix B: Table 16). The American Bullfrog is a generalist and a successful invasive species. In 2012 they were found in comparatively more wetlands than other species and it is likely that they have few requirements for a wetland other than the presence of water.

Boreal Chorus Frog (*Pseudacris maculata*)

There was sufficient data to assess occupancy models for the Boreal Chorus Frog tadpoles in April and June of 2012 and May and June of 2013. The confidence set for April 2012 had seven models and the top model was aquatic vegetation with AICc weight 0.50 (Table 3.5, Appendix B: Table 17). The parameter estimate was positive and therefore the probability of occupancy was higher when aquatic vegetation was present at a wetland (Appendix B: Table 17, Figure 3.2). The second model was size with AICc weight 0.17 and the other five models in the confidence set ranged between that and 0.03. It is unlikely that any of these had much of an effect on occupancy. The second model was null with AICc weight 0.20 and it is unlikely that any of the models following this had an effect on occupancy. The confidence set for June 2012 had seven models and bend was the top model with AICc weight 0.38 (Table 3.5, Appendix B: Table 18). The parameter estimate for Hamburg was positive and the other two were negative, this and the graph of the model predictions shows that the Boreal Chorus Frog tadpoles were only present at Hamburg Bend in June 2012 (Appendix B: Table 18, Figure 3.2). Distance to the nearest wetland was the top model for both May and June 2013 with AICc weights 0.37 and 0.55 (Table 3.5). Both had confidence sets with seven models (In both instances the parameter estimate was negative and the probability of occupancy decreased as the distance increased (Appendix B: Table 19, Figure 3.2).

The model for land cover was the second model for May 2013 and had an AICc weight of 0.25. It is possible that this model also affected occupancy, but the rest of the model had AICc weights less than 0.08 and it is unlikely that they had an effect. The parameter estimate for agriculture was positive and negative for forest indicating that tadpoles were more likely to be present at a wetland surrounded by agriculture than one surrounded by forest (Appendix B: Table 25). The second model for June 2013 was null with AICc weight 0.16. It is unlikely that any of the models that followed had an effect on occupancy (Appendix B: Table 26).

Adult Amphibian Occupancy vs. Larval Amphibian Occupancy

There was sufficient data to assess the larval amphibian models for 13 of the 42 potential species, month, and year combinations. In contrast, the frog call surveys provided sufficient data to assess 28 of the 48 potential combinations (Chapter 2). The different monitoring methods not only produced differing amounts of data, the frog call and tadpole dip net surveys also had results that were different. The occupancy models that were the top models most often for the frog call surveys were aquatic vegetation and slope, with seven and six appearances respectively. These models were present in the top models for the tadpole dip net surveys, but the model that was top model most often was distance to the nearest wetland with four appearances. This model was only the top model once for the frog call surveys (Table 3.6).

When the top models from the same species, month, and year for both methods were compared, there were only three model sets that had the same top model (Table 3.6). These were for the Boreal Chorus Frog in April 2012, Woodhouse's Toad in May 2013, and Plains Leopard Frog in June 2013. For the Boreal Chorus Frog in April 2012, both methods had Aquatic Vegetation as the top model. For both models the parameter estimate was positive and the probability of occupancy was highest when aquatic vegetation was present (Table 3.6, Appendix A and B, Figure 3.2). However, the occupancy estimate for the frog call surveys was much higher. For the Woodhouse's Toad in May 2013 both the adult and tadpole amphibians had bend as the top model. The adult amphibians were found in both Hamburg Bend

and Langdon Bend, but the tadpoles were only found in Langdon Bend (Figure 3.2). This was also the only comparison, out of the 11 coinciding species, month, and year combinations, that had a higher occupancy estimate for the tadpoles (Table 3.6). The tadpoles for both toad species were combined into one category, due to identification difficulty at that stage. Therefore, the higher tadpole occupancy estimate could be caused by this combination. While this was the only combination with a higher occupancy estimate in the tadpoles, Plains Leopard Frogs in May 2013 had the same occupancy estimate with slightly different standard error (Table 3.6). For the Plains Leopard Frog in June 2013, both had slope as the top model. In both cases the parameter estimate was negative and the probability of occupancy decreased as the slope percent increased (Table 3.6, Figure 3.2). Again, the adult anurans had a higher occupancy estimate than the larval amphibians (Table 3.6).

Table 3.5. Top occupancy models for larval amphibian species in restored wetlands along the Missouri River in southeast Nebraska. A set of occupancy models was assessed for every species with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top occupancy models contain the measured covariates that affected tadpole occupancy the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the occupancy models had a positive or negative relationship with occupancy. For $\hat{\beta}$ H represents Hamburg Bend, K represents Kansas Bend, and L represents Langdon Bend. The occupancy estimate ($\hat{\Psi}$) also includes the standard error. For the last column CL refers to confidence limits.

Species	Month	Year	Model	AICcWt	$\hat{\beta}$	$\hat{\Psi}$	95% CL
American Bullfrog	April	2012	$\Psi(\text{aquatic})$	0.23	-	0.310 \pm 0.083	0.200-0.460
American Bullfrog	May	2012	$\Psi(\text{ephemeral})$	0.29	-	0.003 \pm 0.130	0.000-1.000*
American Bullfrog	June	2012	$\Psi(.)$	0.26	NA	0.240 \pm 0.094	0.120-0.430
Boreal Chorus Frog	April	2012	$\Psi(\text{aquatic})$	0.50	+	0.000 \pm 0.079*	0.000-1.000*
Boreal Chorus Frog	June	2012	$\Psi(\text{bend})$	0.38	H (+), K (-), L (-)	0.000 \pm 0.000*	0.000-1.000*
Blanchard's Cricket Frog	June	2013	$\Psi(\text{distwet})$	0.28	-	0.006 \pm 0.016	0.000-1.000*
Smallmouth Salamander	May	2013	$\Psi(\text{bend})$	0.45	H (-), K (-), L (+)	0.680 \pm 25.00	0.000-1.000*
Toad Species	May	2013	$\Psi(\text{bend})$	0.41	H (-), K (-), L (+)	0.260 \pm 13000	0.000-1.000*
Gray Treefrog Complex	June	2013	$\Psi(\text{distwet})$	0.35	-	0.032 \pm 0.044	0.003-0.260
Plains Leopard Frog	May	2013	$\Psi(.)$	0.34	NA	0.330 \pm 0.100	0.180-0.510
Plains Leopard Frog	June	2013	$\Psi(\text{slope})$	0.58	-	0.130 \pm 0.085	0.042-0.340
Boreal Chorus Frog	May	2013	$\Psi(\text{distwet})$	0.37	-	0.046 \pm 0.058	0.006-0.290
Boreal Chorus Frog	June	2013	$\Psi(\text{distwet})$	0.55	-	0.024 \pm 0.047	0.001-0.410

*Number is greater than zero but is smaller than 0.0005

Table 3.6. Top occupancy models for both adult and larval amphibian species in restored wetlands along the Missouri River in southeast Nebraska. A set of occupancy models was assessed for every species with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top occupancy models contain the measured covariates that affected occupancy the most. The occupancy estimate ($\hat{\Psi}$) also includes the standard error. For the last column CL refers to confidence limits.

Species	Month	Year	Adult Model	AICcWt	$\hat{\Psi}$	Tadpole Model	AICcWt	$\hat{\Psi}$
Blanchard's Cricket Frog	April	2012	$\Psi(\text{distwet}+\text{agr}+\text{forest})$	0.67	0.340 \pm 0.230			
Blanchard's Cricket Frog	May	2012	$\Psi(\text{bend})$	0.28	0.999 \pm 0.240			
Blanchard's Cricket Frog	June	2012	$\Psi(\text{aquatic})$	0.33	0.690 \pm 0.390			
Woodhouse's Toad	April	2012	$\Psi(\text{bend})$	0.33	0.000 \pm 0.000*			
Gray Treefrog Complex	April	2012	$\Psi(\text{slope})$	0.85	0.000 \pm 0.000*			
Gray Treefrog Complex	June	2012	$\Psi(\text{ephemeral})$	0.76	0.380 \pm 0.290			
Plains Leopard Frog	April	2012	$\Psi(.)$	0.68	0.830 \pm 0.100			
Plains Leopard Frog	May	2012	$\Psi(\text{distwet})$	0.31	0.630 \pm 0.150			
Plains Leopard Frog	June	2012	$\Psi(\text{ephemeral})$	0.47	1.000 \pm 0.530			
American Bullfrog	April	2012	$\Psi(.)$	0.26	0.999 \pm 0.180	$\Psi(\text{aquatic})$	0.23	0.310 \pm 0.083
American Bullfrog	May	2012	$\Psi(\text{bend})$	0.74	1.000 \pm 0.003	$\Psi(\text{ephemeral})$	0.29	0.003 \pm 0.130
American Bullfrog	June	2012	$\Psi(\text{slope})$	0.39	0.840 \pm 0.280	$\Psi(.)$	0.26	0.240 \pm 0.094
Boreal Chorus Frog	April	2012	$\Psi(\text{aquatic})$	0.43	0.940\pm0.780	$\Psi(\text{aquatic})$	0.50	0.000\pm0.079*
Boreal Chorus Frog	June	2012				$\Psi(\text{bend})$	0.38	0.000 \pm 0.000*
Blanchard's Cricket Frog	May	2013	$\Psi(\text{slope})$	0.49	0.999 \pm 0.014			
Blanchard's Cricket Frog	June	2013	$\Psi(\text{aquatic})$	0.70	0.820 \pm 0.220	$\Psi(\text{distwet})$	0.28	0.006 \pm 0.016
Woodhouse's Toad	May	2013	$\Psi(\text{bend})$	0.33	0.120\pm4.550	$\Psi(\text{bend})$ Toad species	0.41	0.680\pm25.00
Woodhouse's Toad	June	2013	$\Psi(\text{size})$	0.23	0.860 \pm 0.800			
Great Plains Toad	June	2013	$\Psi(\text{slope})$	0.64	0.006 \pm 0.041			
Gray Treefrog Complex	May	2013	$\Psi(\text{aquatic})$	0.62	0.029 \pm 0.510			
Gray Treefrog Complex	June	2013	$\Psi(\text{aquatic})$	0.51	0.780 \pm 0.095	$\Psi(\text{distwet})$	0.35	0.260 \pm 13000
Plains Leopard Frog	April	2013	$\Psi(.)$	0.29	0.330 \pm 0.100			
Plains Leopard Frog	May	2013	$\Psi(\text{aquatic})$	0.79	0.032 \pm 0.580	$\Psi(.)$	0.34	0.032 \pm 0.044
Plains Leopard Frog	June	2013	$\Psi(\text{slope})$	0.30	0.710\pm0.160	$\Psi(\text{slope})$	0.58	0.330\pm0.100
American Bullfrog	May	2013	$\Psi(\text{distwet}+\text{agr}+\text{forest})$	0.20	0.430 \pm 1.250			
American Bullfrog	June	2013	$\Psi(\text{slope})$	0.93	0.999 \pm 0.012			
Boreal Chorus Frog	April	2013	$\Psi(.)$	0.22	0.650 \pm 0.130			
Boreal Chorus Frog	May	2013	$\Psi(.)$	0.43	0.560 \pm 0.140	$\Psi(\text{distwet})$	0.37	0.130 \pm 0.085
Boreal Chorus Frog	June	2013	$\Psi(\text{aquatic})$	0.45	0.270 \pm 0.130	$\Psi(\text{distwet})$	0.55	0.046 \pm 0.058
Smallmouth Salamander	May	2013				$\Psi(\text{bend})$	0.45	0.024 \pm 0.047

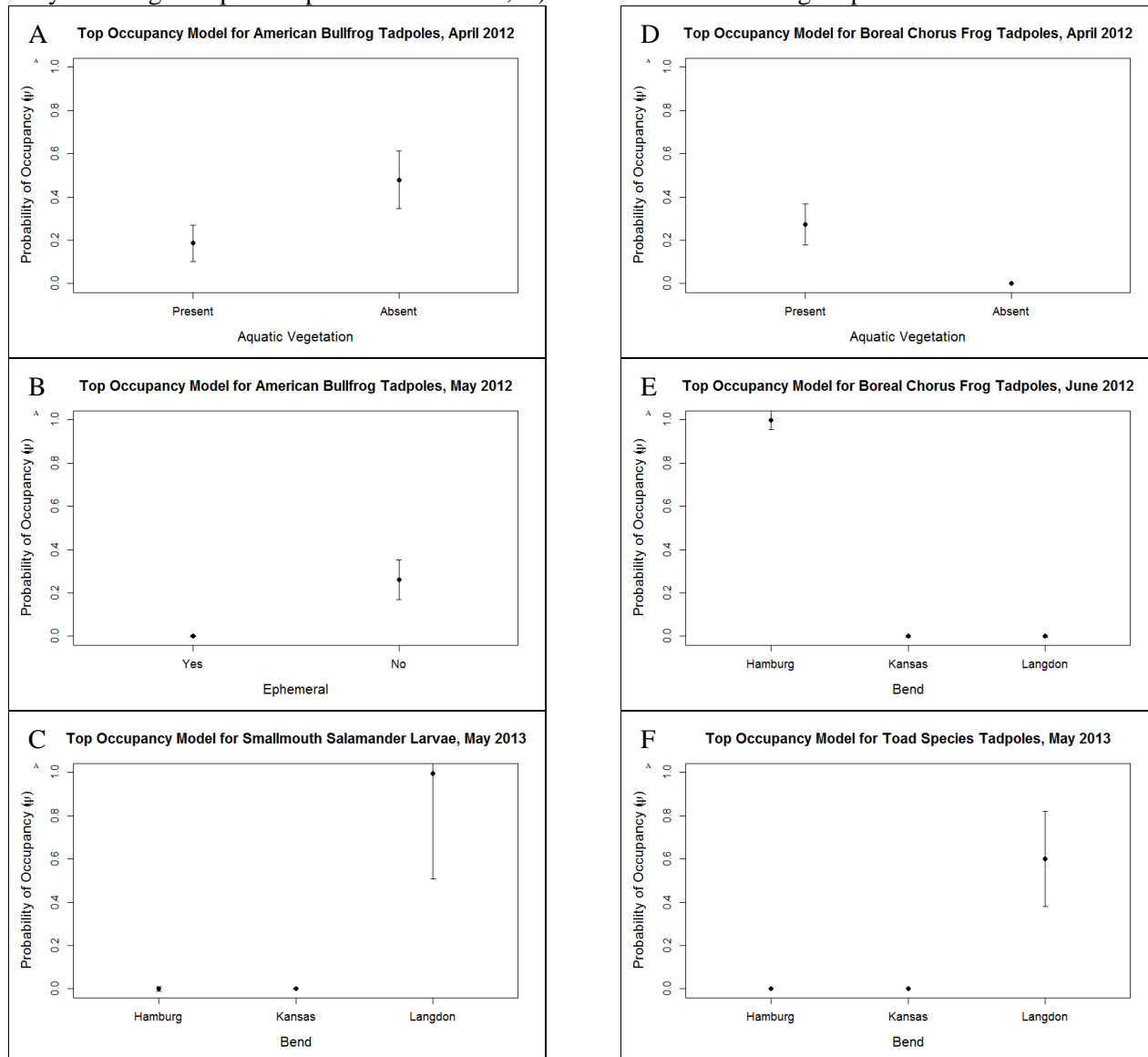
*Number is greater than zero but is smaller than 0.0005

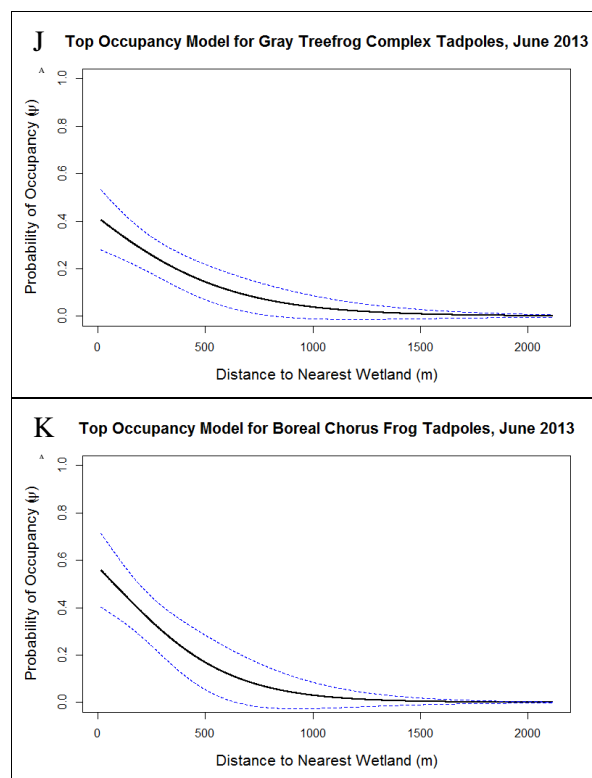
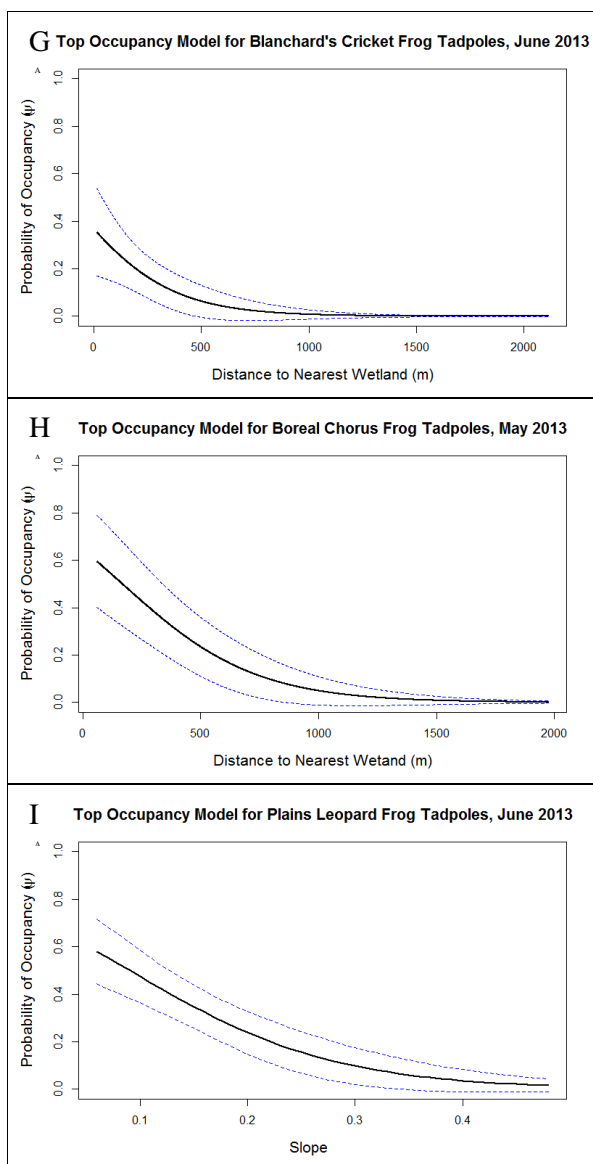
Table 3.7. Equations for the top slope and distance to nearest wetland models for both the adult and larval amphibian occupancy. Where y is occupancy and x is a distance or percent slope.

Species	Life Stage	Month	Year	Covariate	Equation
Gray Treefrog Complex	Adult	April	2012	Slope	$y = -173.60x + 25.50$
American Bullfrog	Adult	June	2012	Slope	$y = 53.66x - 8.26$
Blanchard's Cricket Frog	Adult	May	2013	Slope	$y = 19.30x - 2.30$
Great Plains Toad	Adult	June	2013	Slope	$y = -45.98x + 6.87$
Plains Leopard Frog	Adult	June	2013	Slope	$y = -9.60x + 3.39$
American Bullfrog	Adult	June	2013	Slope	$y = 46.06x - 5.78$
Plains Leopard Frog	Tadpole	June	2013	Slope	$y = -10.59x + 0.96$
Plains Leopard Frog	Adult	May	2012	Distance to Nearest Wetland	$y = -0.0019x + 2.29$
Blanchard's Cricket Frog	Tadpole	June	2013	Distance to Nearest Wetland	$y = -0.004x + -0.54$
Gray Treefrog Complex	Tadpole	June	2013	Distance to Nearest Wetland	$y = -0.003x - 0.34$
Boreal Chorus Frog	Tadpole	June	2013	Distance to Nearest Wetland	$y = -0.004x + 0.29$
Boreal Chorus Frog	Tadpole	May	2013	Distance to Nearest Wetland	$y = -0.004x + 0.60$

*Number is greater than zero but is smaller than 0.0005

Figure 3.2. Graphs of top occupancy model predictions for tadpole dip net surveys. The graphs are for A) American Bullfrog tadpoles in April 2012, B) American Bullfrog tadpoles in May 2012, C) Smallmouth Salamander larvae in May 2013, D) Boreal Chorus Frog tadpoles in April 2012, E) Boreal Chorus Frog tadpoles in June 2012, F) Toad species tadpoles in May 2013, G) Blanchard's Cricket Frog tadpoles in June 2013, H) Boreal Chorus Frog tadpoles in May 2013, I) Plains Leopard Frog tadpoles in June 2013, J) Gray Treefrog Complex tadpoles in June 2013, K) and Boreal Chorus Frog tadpoles in June 2013.





DISCUSSION

To determine what a successful wetland is and create the wetland restoration guidelines it is essential to consider the needs of all life stages and as many amphibian species as possible. If this could be done using only frog call surveys or tadpole dip net surveys, this would be ideal. While conducting both ensures that all life stages are considered, it is time consuming. Adult and larval amphibians produced differing results and none of the species assessed responded in the same way to covariates potentially affecting detection and occupancy of Missouri River wetland. Under the same circumstances, the frog call surveys produced 28 model sets and the tadpole dip net surveys produced only 13. Only three of the top models had the same model for both the adult and larval amphibians. While the frog call surveys produced enough data, the differences in the top models for tadpole dip net surveys suggest that important covariates could be overlooked. Additionally, occupancy estimates based on chorus' alone may overestimate effective occupancy (occupancy of wetlands contributing to population viability) and lead to the spurious identification of covariates. Tadpole surveys provide better data because of their link to reproduction and the lack of spurious data. They can also be collected during the day, when most biologist and managers are typically working. This is a safer work environment than conducting frog call surveys late at night. If one method is used, it should be tadpole dip net surveys. However, if the surveys are conducted during a drought year or a flood year there will probably not be enough data to perform an occupancy analysis for most tadpole species. Therefore, conducting both methods is the best way to produce the most accurate wetland restoration guidelines.

There were covariates that had more support from the frog call surveys and others that had more support from the tadpole dip net surveys. The covariate most frequently in model confidence sets and top model for frog occupancy was aquatic vegetation. Overall, the adult anurans were more likely to be present at wetlands with aquatic vegetation. This is also a top model for larval amphibians twice, and was present in every confidence set. The same overall trend was seen in the tadpoles as well. Aquatic vegetation provides protection from predators for both adults and larvae amphibians, is used to anchor

eggs, and for some species it is used as a place to call from. Thus, it is not surprising that its presence is an important part of a successful wetland. However, more research should be done to specify what kind of aquatic vegetation is best and how this varies regionally.

Slope was the second most frequent top model for the adult amphibians and was the top model for larval amphibians once. However, it was among the larval amphibian confidence sets 11 times. For both life stages the overall trend was a decrease in the probability of occupancy as the slope percent increased. A more specific idea of what slope to use can be found by using the equations made for the species that had slope as a top model (Table 3.7). It is as easy as picking a representative species with slope as a top model, such as the Gray Treefrog complex, the Great Plains Toad, or the Plains Leopard frog, and then entering the desired occupancy for y and solving for the percent slope (x) (Table 3.7). However, most species had higher occupancies when the percent slope is less than 0.20 (Chapter 2).

The model for ephemeral was the top model twice for the adults and once for the larval amphibians. For the adults it was clear that the probability of occupancy was highest when a wetland was ephemeral, but it was not as clear for the tadpoles. It is possible that these results were skewed by the drought in 2012 and the beginning of 2013. For example, if the adults bred in both ephemeral and some permanent, but most of the ephemeral wetlands dried up before the tadpoles could metamorphose this would make it seem like they preferred permanent wetlands. In addition, the American Bullfrog adults and tadpoles often had opposing results to the majority of the other species. This is because the tadpoles over winter and need a permanent body of water to survive. Given that they are considered an invasive species and are not native to Nebraska, it probably would be best to leave them out when building the wetland restoration guidelines. With this in mind, it is easier to conclude that the overall probability of occupancy is higher when a wetland is ephemeral.

The model for distance to the nearest wetland was the top model four times for the larval amphibians, but only once for the adults. This is where including the tadpole surveys becomes important.

Clearly this is an important variable in maintaining a viable amphibian population and therefore is part of what defines a successful wetland. For most of the species in both life stages the probability of occupancy decreased as the distance to the nearest wetland increased. Amphibians differ in size and dispersal ability, but they all tend to exist as metapopulations especially in a dynamic ecosystem like the Missouri River Flood Plains. Therefore, having other wetlands within their dispersal range not only aids in genetic distribution, but also provides habitat in situations like drought. The results show that it would be best to have the next wetland within 500 m (Figure 3.2). The equations for distance to the nearest wetland can provide a more specific distance, depending on the occupancy goal and the species the equation was made for (Table 3.7). The process is the same for the distance to the nearest wetland equations as the process for the slope equations, enter the desired occupancy for y and solve for the distance or x . All of the tadpole species that had distance to the nearest wetland as the top-ranked model were smaller species. It is possible that the distance to the nearest wetland is important to the larger species as well, but because most wetlands have other wetlands within their dispersal distances it did not show up as a top model. However, a successful wetland provides habitat for a diverse range of species and thus it is important that another wetland is within 500 meters.

Therefore, the combined frog call and tadpole data suggest that a successful wetland contains aquatic vegetation, has a shallow slope (less than 0.20), is ephemeral, and has at least one other wetland nearby (within 500 m). These may be simple and general guideline, but they do give a starting point for creating successful wetlands on a flood plain. Amphibians have been used to inform wetland restoration in many different ways. Examples include using amphibian distribution to inform buffer zones around wetlands, looking at how adjacent land use effects amphibian species richness, and analyzing behavioral responses and survival to determine whether local wetland sites should be designed to support metapopulations or patchy populations (Rittenhouse and Semlitsch 2007; Houlahan and Findlay 2003; Pentranka and Holbrook 2006). Occupancy analysis has often been used to compare different hypothesis about what affects amphibian distribution and detection (Mazerolle et al. 2005; Muths et al. 2005; Weir et

al. 2005; Bailey et al. 2004). However, amphibian occupancy analysis has rarely been used to specifically inform wetland restoration. Balas et al. used amphibian occupancy to determine the importance of season wetlands in conservation grasslands and Hellman conducted a very similar analysis of the Nebraska wetland sites for the first two years of the overall amphibian monitoring project. She found that shallow slope was important for both adult and tadpole amphibian occupancy (Balas et al. 2012; Hellman 2013). My analysis has produced additional wetland variables that define a successful wetland, shown that tadpole dip net surveys provide better data than frog call surveys, and has also provided equations for slope and distance to the nearest wetland to aid in wetland restoration. There is still a disparity between what is legally required to create or restore a wetland and their ability to replace the lost structure and functions (Streever 1999). Using amphibian occupancy to inform wetland restoration provides a way to determine if restored wetlands are providing some of the lost function and structure of the original wetlands. The definition of a successful wetland gained from this study provides both information that will lead to more successfully restored wetlands, and a framework for more specific wetland restoration guidelines. Thus, continued monitoring in an adaptive management framework is needed to build upon the wetland restoration guidelines.

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CHAPTER 4: USING AMPHIBIAN CO-OCCURANCE TO INFORM WETLAND RESTORATION

INTRODUCTION

Wetlands play a significant role in ecosystem functions. They contribute to nutrient cycling, are essential to the hydrological cycle, and provide habitat for many fish and wildlife species (Mitsch and Gosselink 2007). Unfortunately, much of this critical habitat has been lost to channelization or has been degraded. Along the Missouri River the main cause for wetland habitat loss was the Bank Stabilization and Navigation Project implemented in 1912. Its passage led to the channelization of the Missouri River and the loss of over 200,000 ha of Missouri River habitat (U.S. Army Corp of Engineers Kansas City and Omaha Districts 2003).

The importance of the Missouri River for navigation has been steadily decreasing since 1977, especially above Kansas City (MRRP pamphlet 2009). Concomitant with this decrease has been an increase in the recognition of the importance of wetlands and wetland habitats (Mitsch and Gosselink 2007, Moll and Moll 2004, Zedler and Kercher, 2005). Wetlands provide a broad range of ecosystem services, including regulating the hydrological cycle, contributing to nutrient cycling, and acting as refuge for many fish and wildlife species. In addition to those ecosystem services, wetlands, and undiked floodplains, can greatly reduce flooding and flood impacts (Bakacsi et al. 2011). Climate variability is expected to increase in the relatively near future, and this variability is expected to include more extreme weather events (IPCC 2003). The recognition of the importance of wetlands has guided U.S. government policy, and led to legislation such as the Clean Water Act of 1986 and the Water Resources Development Act of 1986.

After the enactment of the Clean Water Act of 1986 there has been effort by the United States Army Corps of Engineers (USACE) to enforce the policy of “no net loss.” From 1996 until 2005 there was an estimated net gain of 10,000 ha/yr of wetland and associated uplands habitat. However, it is

unclear how successful these created, restored, enhanced, or preserved wetlands are in terms of restoring function and habitat. There is little information on what ecosystem functions were lost and what functions were gained (Mitsch and Gosselink 2007). Although restoration techniques continue to improve, there are still no overall guidelines for building a functional wetland outside of engineered hydrological changes. Because of ongoing restoration efforts, increasing threats to biodiversity and to wetlands, and lack of ecologically-based performance guidelines, there needs to be concerted and quantifiable efforts to assess the success of restored riverine wetland systems, so that general requirements can be put into effect.

The Water Resources Development Act of 1986 led to the creation of the Missouri River Fish and Wildlife Mitigation Project (MRFWMP). The MRFWMP is dedicated to preserving existing habitat and restoring lost habitat that supports native vegetation, fish, and wildlife species within the floodplain. The requirements of the MRFWMP include monitoring restored habitat (U.S. Army Corp of Engineers 2006). In 2008, the MRFWMP created an opportunity to monitor amphibian populations as part of the overall monitoring effort. An amphibian sub-committee was created, and a monitoring protocol designed and implemented in Nebraska, Iowa, Kansas, and Missouri. Amphibian monitoring was conducted in order to assess the success of previously restored wetlands and to help create wetland restoration guidelines.

Amphibians were chosen for monitoring because they are globally declining (Glascon *et al.* 2005; Pentranka and Holbrook 2006; Collins and Crump 2009; Crump 2010), they integrate terrestrial and aquatic environments (Collins and Crump 2009; Vitt and Caldwell 2009; Collins and Halliday 2005) and because they are good indicators of wetland restoration success (Bowers *et al.* 1998; Collins and Halliday 2005; Welsh and Ollivier 1998). Specifically, their biphasic life cycle and permeable skin make them uniquely sensitive to conditions in the water and the surrounding terrestrial environment. Wetlands are complex ecosystems that are hard to define and are made of both aquatic and terrestrial components. They are easily altered or degraded by changes in either. Thus, amphibians, whose biphasic life cycle and

permeable skin makes them dependent on these intricate and essential ecosystems, are ideal indicators of their success.

Amphibian monitoring has a relatively long record, but historically amphibian monitoring focused on call surveys, because they are easy, rapid, and most species give predictable vocalizations. However, call surveys are flawed in that they fail to account for detection differences, which vary by season, day, time of day, location and among species. Vocalization loudness is often not predictive of abundance. In 2002, MacKenzie et al. presented a method for amphibian monitoring that statistically accounts for varying detection rates (MacKenzie et al. 2002). This method, occupancy modeling, consists of conducting two or more surveys at each site (in this case wetland) within a specified period of time. It is assumed that the system is closed and that the population (and therefore the probability of detection) does not change during the specified period of time. If a species is found at least once during the time period it is present at that site. However, if it is not found it is either present and undetected or not present. If one, reasonably, assumes that the detection and occupancy are not constant across sites, then it is possible to model the probability of detection and occupancy as a function of a measured covariate.

Generally, single species models are used in occupancy modeling. However, single species models can lead to numerous supported models, with significant covariates of presence varying by species in confusing ways (Rheme, Powell, and Allen 2011), and most wetlands are occupied by more than one species. Because the goal of wetland restoration is to create functioning wetlands, rather than a series of habitat patches for individual species, I conducted occupancy models for co-occurring groups of species, assuming that co-occurring groups of species within restored wetlands is related to wetland function. I also conducted individual species models, because a wetland supporting only a single species may still be functionally important, but those results are reported in Chapter 2. Therefore, the environmental variables present at the co-occupied wetlands were used to define wetland success. This definition was then used to inform wetland restoration for flood plains wetlands.

METHODS

Study Sites

The sites used in this study consist of 55 restored wetlands distributed among three wetland complexes (river bends) between Nebraska City, NE and Nemaha, NE. Hamburg Bend is located 8 miles southeast of Nebraska City, Nebraska in Otoe County. This is the northern most wetland complex and it consists of ~638 ha of former agricultural land that was purchased by the USACE between 1993 and 2004. Hamburg Bend is adjacent to the right descending bank of the Missouri River between river miles 552 to 556. Due to the alterations of the Missouri River Bank Stabilization and Navigation Project (BSNP), the side channels were closed and this allowed the land to accrete. In response a side channel was constructed as well as several back water areas (U.S. Army Corp of Engineers 2006).

Kansas Bend is located 3 miles east of Peru, Nebraska in Nemaha County. This wetland complex is composed of ~427 ha of former agricultural land and is separated into two areas by privately owned farmland. Kansas Bend was purchased by the Corps between 1993 and 1999 and is adjacent to the right descending bank of the Missouri River between river miles 544 to 547. Due to the activities of the BSNP several side channels were closed using dikes and revetments. Two side channels have been reopened.

Langdon Bend is located 3 miles south of Brownville, Nebraska, in Nemaha County. This is the southernmost wetland complex and it is comprised of ~675 of former agricultural land that was purchased by the USACE between 1994 and 2003. Langdon Bend is the area adjacent to the right descending bank of the Missouri River between river miles 520 and 532. Due to the Missouri River Bank Stabilization and Navigation Project the chute at Langdon Bend was closed and the side channel was cut off. The old channel could not be opened due to its proximity on the upstream end to the Cooper Nuclear Power Plant. Thus, the newly constructed channel is connected to the river at the outlet, but stops before meeting the river at the upstream end. The Corps in conjunction with the Nebraska Game and Parks Commission designed the wetland complex that was built on the west side of the levee. There are about 89 ha of wetlands that were constructed starting in July 2008 and ending May 2009.

Frog Call Surveys

Occupancy data for the adult amphibians was collected using frog call surveys. In 2012 and 2013 each wetland was visited twice within a two week period during April, May, and June. Surveys were started a half an hour after sunset and were concluded no later than 1 am, because most temperate anuran species peak calling hours end around midnight (Dodd 2010). To avoid disturbing the chorus, surveys were conducted 10 m from the shoreline after a 2 min acclimation period. We listened and recorded with a hand held recording device (Olympus DM-10) for five min and marked the presence of any species heard calling. The recordings were later analyzed by a second observer to decrease listener bias (Lotz and Allen 2007).

Covariates

Both survey specific covariates and site specific covariates were collected for the frog call surveys. The survey specific covariates time of day, day of year, air temperature, water temperature, wind speed (m/s), water pH, relative humidity, were collected for every frog call survey. The survey specific covariates moon phase (percent) and cloud cover (percent) were only collected for the frog call surveys. Moon phase percent was determined from the United States Naval Meteorology and Oceanography Command website (<http://aa.usno.navy.mil/data/docs/MoonFraction.php>). Percent cloud cover was visually estimated. Moon phase and percent cloud cover were combined into a covariate called moonshine by first subtracting the percent cloud cover from one and then multiplying this by the moon phase percent.

Tadpole dip net surveys were conducted along with the frog call surveys. However, due to lower tadpole naïve occupancy it was not possible to run co-occurrence model sets for the tadpole data. Three of the site specific covariates, aquatic vegetation, adjacent vegetation, and the shallow water slope, were collected during tadpole dip-net surveys during the daytime. For each sweep, in addition to the presence of tadpoles, I determined whether the net was drawn through herbaceous, woody, or open water and a water height measurement was taken at 0.5 m and 1 m from the shoreline. Using all 20 net sweeps I then

calculated the percentage of herbaceous, woody, and open water habitat along the 200 m surveyed. In addition, a visual survey of the entire wetland was done and aquatic vegetation was marked as present or absent. Aquatic vegetation was determined as present if it was present in the visual survey or if the percent of herbaceous and woody vegetation was $\geq 10\%$. A visual survey was also used for adjacent vegetation. I determined the percent grass, herbs and forbs, trees and shrubs, or bare ground within 1 m of the waterline. Shallow water slope was calculated using the water height measurement at 1 m and dividing by 100 cm. Two covariates, whether a wetland is ephemeral or not and wetland size, were determined at the beginning of the monitoring activity in 2009. Wetland size was recorded as small (≤ 2 ha), medium (2-5 ha), and large (> 5 ha). Land cover and the distance to the nearest wetland were calculated using ArcGIS. Land cover was determined by creating a 1000 m buffer around each wetland and determining the percentage of grass, forest, and agriculture within that buffer, and assigning landcover type to the dominant (%) landcover class. The distance to the nearest wetland was determined from digital landcover maps, and calculated from centroid to centroid.

Analysis

An occupancy modeling analysis was conducted using the package unmarked in program R (R Development Core Team 2008; Fiske and Chandler 2011). Occupancy modeling represents a major improvement to amphibian monitoring techniques because it accounts for differences in detection, which may vary across time, across wetlands, and across species. As manifest in Program Presence (Hines 2006) and in the package unmarked in program R, I used an occupancy model structured multi-model inference to select models that best fit the observed data. Like all implementations of multi-model inference, model selection is critical to sound inference. Model sets were created *a priori* and were run as single species - single season models, using month as season. To run single species models for more than one species a one was used to represent when both species were present and a zero was used for any other situation. Therefore, detection (p) is only the detection of the species as a group. This method does not account for varying detection of species. This will affect the detection estimates and thus produce

conservative occupancy estimates. However, the goal of this analysis is to define what a successful wetland is by determining what parameters affect detection and how and not the amphibian occupancy estimates. The co-occurrence models by MacKenzie, Bailey, and Nichols were not used because I wanted to use models that were similar to the models used in the single species analysis (Chapter 2; MacKenzie, Bailey, and Nichols 2004). With their co-occurrence models the model parameters increase exponentially with the number of species used and it is unlikely that the data collected is enough to produce results from such a model set (MacKenzie, Bailey, and Nichols 2004). My method is far simpler both in process and output, while still producing results that are useful to the goals of the project.

The species combinations used were chosen either because they had the largest naïve occupancy, or because they were an interesting and potentially important combination. Some species combinations were both. A separate model set was created for adult detection (12 models) and adult occupancy (12 models) (Table 4.1 and Table 4.2). Some of the models in each model set correlated, but they were all included because they have different management implications. All chosen combinations were modeled for all months and years that had naïve occupancy (number of wetlands that a species was present at least once/total number of wetlands) higher than 0.10. This cutoff was chosen because species with low detection have uncertain occupancy and most model sets below this naïve occupancy would not run in program R (Hellman 2013). There are fewer model sets from the year 2012 due to drought which caused many wetlands to dry and fewer anurans to call. Detection models were conducted separately from occupancy models. The detection models were conducted first using the global covariates to hold occupancy constant (Table 4.1). Then, the top detection model for each data set was used to hold detection constant while running the occupancy models (Table 4.2). Due to drought in 2012 and the beginning of 2013, many of the wetlands dried and adult occupancy was affected. This increased the number of models and covariates in the global occupancy model that wouldn't converge. I removed variables from the global model if convergence was not achieved and any models that did not converge were removed from the model sets.

To determine what variables affect detection and occupancy, a confidence set accounting for 95% of overall model weight was considered (Burnham and Anderson 2002). The 95% confidence set was calculated by adding up the models AICc weight until it was greater than or equal to 0.95. A large confidence set was used to ensure that no important parameters were excluded. As previously mentioned, my research is part of a multi-state monitoring project that is set up to function in an adaptive management framework. If continued, an inclusive confidence set allows for more confident model set refinement. Within the confidence set the AICc weight was used to determine the model's fit. Parameter estimates were used to determine what affect a covariate/s had on detection or occupancy. In this case, a positive parameter estimate for a continuous variable like slope or distance to the nearest wetland meant that the detection or occupancy increased as the covariate/s increased. A positive parameter estimate for a categorical variable like aquatic vegetation, bend, or size meant that occupancy or detection was higher with/at that covariate than without/not at it. For continuous variables a negative parameter estimate meant that detection or occupancy was decreasing as the covariate/s increased. For categorical variables a negative parameter estimate meant that occupancy or detection was lower with/at that covariate than without/not at it. The covariates in the top models were graphed to show how detection or occupancy was affected by that particular covariate.

Table 4.1. Detection models for adult anurans species in restored wetlands along the Missouri River in southeast Nebraska. Frog call and covariate data was collected during April, May, and June of 2012 and 2013. Detection is represented by p , and occupancy is represented by Ψ . Detection was assessed first, using the global covariates to hold occupancy constant. The global covariates are bend, wetland size, slope, aquatic vegetation, adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest. The top detection models held the covariates that were most likely to have affected adult anuran detection. The covariate/s in top detection model was used to hold detection constant when the occupancy was assessed. The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(\text{Global Covariates}) p(.)$
Time	$\Psi(\text{Global Covariates}) p(\text{time})$
Wind (m/s)	$\Psi(\text{Global Covariates}) p(\text{wind})$
Day (julian date)	$\Psi(\text{Global Covariates}) p(\text{julian})$
Wetland Size	$\Psi(\text{Global Covariates}) p(\text{size})$
Moonshine	$\Psi(\text{Global Covariates}) p(\text{moonshine})$
Air Temperature	$\Psi(\text{Global Covariates}) p(\text{airtemp})$
Water Temperature	$\Psi(\text{Global Covariates}) p(\text{H2Otemp})$
Environmental Covariates 1	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind})$
Environmental Covariates + Day	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+julian})$
Environmental Covariates 2	$\Psi(\text{Global Covariates}) p(\text{H2Otemp+airtemp+wind+moonshine})$
Global	$\Psi(\text{Global Covariates}) p(\text{time+wind+julian+size+moonshine+airtemp+H2Otemp})$

Table 4.2. Occupancy models for both adult anurans in restored wetlands along the Missouri River in Southeast Nebraska. Frog call and covariate data was collected during April, May, and June of 2012 and 2013. Detection is on the right represented by p , and occupancy is on the left represented by Ψ . Occupancy was assessed after detection and used the covariate/s from the top detection model (from each separate model set) to hold detection constant. The top detection models held the covariates that were most likely to have affected adult anuran detection. The top occupancy models provided the covariates that helped define a successful wetland and were used to build the wetland restoration guidelines.

Name	Model
Null	$\Psi(.) p(\text{Top Detection Covariate/s})$
Bend	$\Psi(\text{bend}) p(\text{Top Detection Covariate/s})$
Wetland Size	$\Psi(\text{size}) p(\text{Top Detection Covariate/s})$
Slope	$\Psi(\text{slope}) p(\text{Top Detection Covariate/s})$
Aquatic Vegetation	$\Psi(\text{aquatic}) p(\text{Top Detection Covariate/s})$
Adjacent Vegetation	$\Psi(\text{adjacent}) p(\text{Top Detection Covariate/s})$
Ephemeral Wetland	$\Psi(\text{ephemeral}) p(\text{Top Detection Covariate/s})$
Distance to Nearest Wetland	$\Psi(\text{distwet}) p(\text{Top Detection Covariate/s})$
Land Cover	$\Psi(\text{agr+forest}) p(\text{Top Detection Covariate/s})$
Wetland Vegetation	$\Psi(\text{aquatic+adjacent}) p(\text{Top Detection Covariate/s})$
Connectivity	$\Psi(\text{distwet+agr+forest}) p(\text{Top Detection Covariate/s})$
Global	$\Psi(\text{bend+size+slope+aquatic+adjacent+ephemeral+distwet+agr+forest}) p(\text{Top Detection Covariate/s})$

RESULTS

Detection

The number of wetlands holding water varied from month to month during 2012 and 2013. Annual variation in flood plain wetlands is typical; however the variation in 2012 and 2013 was extreme. In 2011 the Missouri River experienced historic flooding due to both higher than average snowpack melt and higher than average amounts of rainfall during the spring and summer months. This was followed by a historic drought in 2012 that continued into 2013. The drought was caused by record high temperatures combined with below average snowpack melt in the Rocky Mountains, and precipitation. From mid-April until the beginning of May 2012 the water table was still high enough that average rainfall caused an increase in the number of wetlands holding water. However, from that point on the high temperatures continued to decrease the number of wetlands that were holding water. Southeast Nebraska received average to above average rainfall in March and April of 2013, but it wasn't until the extreme rainfall events in late May of 2013 that the soil was saturated enough for most of the wetlands to hold water. Thus, frog call suveys were conducted at 26 wetlands in April 2012, 30 wetlands in May 2012, 21 wetlands in June 2012, 22 wetlands in April 2013, 20 wetlands in May 2013, and 43 wetlands in June 2013.

There were 12 models used in to assess detection for adult anurans (Table 4.1). Ideally, these would have been run for all seven species combinations, all months of April, May, and June, and both years 2012 and 2013, for a total of 42 potential model sets. None of the species combinations had sufficient data to run for all months and both years. Thus, the models were run 14 times and there were 14 top models and confidence sets. The confidence sets contained between 1 to 6 models (Appendix C). Overall, detection was affected by, time, air temperature, moonshine, and environmental covariates 1 (the combination of wind+air temperature+water temperature+moonshine). However, null was the top model seven of the 14 model runs and was in the confidence set 13 times (Table 4.3, Appendix C). This suggests that within the methodological constraints that the data was surveyed, the covariates measured

are not affecting the detection of the adult anurans, or statistical power or effect size were low. Below I address those covariates that were in the confidence sets.

Time

The model for time was the top model for detection twice and was among the confidence sets nine times (Table 4.3, Appendix C). For eight out of the nine total models in the confidence sets, including both top models, the parameter estimate was positive. A positive parameter estimate means that the probability of detection increased as the time increased (Appendix C, Figure 4.1). Time was converted to decimal time and this may have made it difficult to see the true trend in the probability of detection, due to an inability to show the time line from sunset until sunrise. In decimal time midnight is represented by zero and R will not model a sequence that goes from a higher number to a lower number. Therefore, while we did not sample at all times of the day, a reasonable biological explanation for time effecting detection is that the frogs started calling around 20.0, or 8 PM, increased calling until around midnight, and then decreased calling until they stopped around sunrise, as one would expect.

Air Temperature

The model for air temperature was the top model twice and was among the detection confidence sets 11 times (Table 4.3, Appendix C). For ten of the 11 models in the confidence sets, including both of the top models, the parameter estimate was positive. As the air temperature increased, the probability of detection increased (Appendix C, Figure 4.1). This concurs with current knowledge of anuran biology and behavior. The frequency, volume, call tempo, and even pitch of a frog call will increase as the weather warms. Thus, we were more likely to detect them when the air temperature was higher.

Moonshine

The model for moonshine was the top model for detection twice and was among the confidence sets eight times (Table 4.3, Appendix C). For five of the eight models in the confidence sets, including

the two top models, the parameter estimate was negative. A negative parameter estimate indicates that the probability of detection decreases as the percent moonshine increases (Appendix C, Figure 4.1).

Environmental Covariates 2

The model for environmental covariates 2 (water temperature + air temperature + wind + moonshine) was only among the detection confidence sets once, but it was the top model with 0.96 AICc weight (Appendix C, Table 4.3). Detection was affected by the interaction of these four covariates. The model is additive and so it couldn't be graphed. However, the parameter estimates were -33.878 ± 14.436 for moonshine, 3.704 ± 1.725 for water temperature, -0.966 ± 0.528 for air temperature, and 1.727 ± 2.964 for wind. The parameter estimates for moonshine and air temperature are negative and the parameters estimates for water temperature and wind are positive. Therefore, the probability of detection was highest when the percent moonshine and air temperature was low, and the water temperature and wind speed was high.

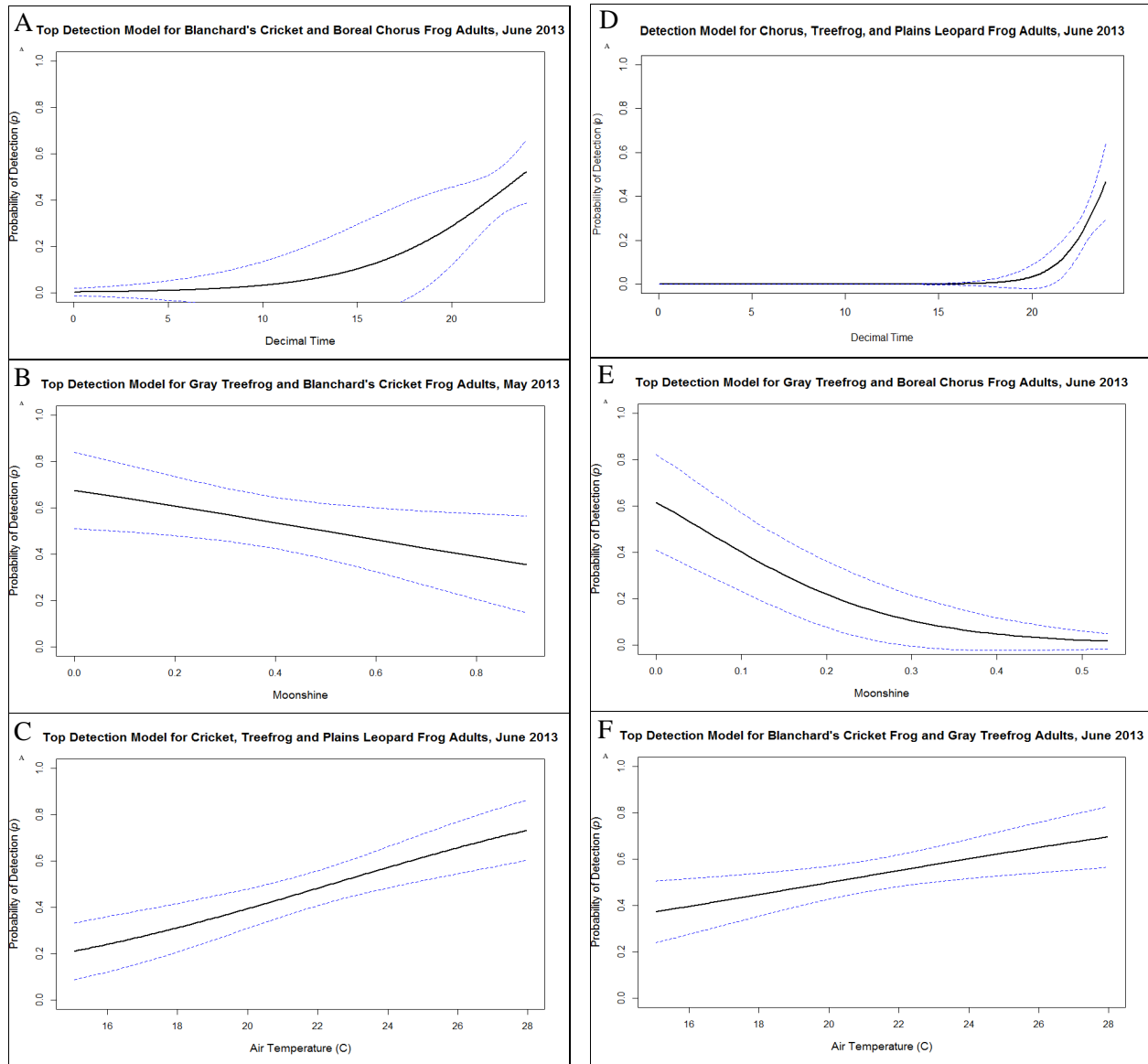
Other Models

The model for water temperature was never the top model, but was among the detection confidence sets nine times. For seven of the nine models the parameter estimate was positive and the probability of detection increased as the water temperature increased (Appendix C). Wind was among the confidence sets seven times, but was never the top model. Out of the seven models in the confidence sets, four had a negative parameter estimate and the probability of detection decreased with the increase in wind speed (Appendix C). It is difficult to draw conclusions, given the low placement of the models and the almost even split of positive and negative parameter estimate. However, we do know that it is harder to hear frog calls when the wind speed is high, and thus detection would be lower with high wind speeds. The model for size was among the confidence sets twice, but was never a top model. For both models the parameter estimate for small wetlands was positive and the parameter estimates for medium and large were negative. This indicates that the frogs were more likely to be detected at small wetlands (Appendix C).

Table 4.3. Top detection models for co-occurring adult anurans in restored wetlands along the Missouri River in southeast Nebraska. A set of detection models was assessed for biologically important species combinations with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top detection models contain the measured covariates that affected frog call detection the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the detection models had a positive or negative relationship with detection. The detection estimate (\hat{p}) also includes the standard error.

Species	Month	Year	Model	AICcWt	$\hat{\beta}$	\hat{p}	95% CL
Cricket Frog + Treefrog	June	2012	$p(.)$	0.56	NA	0.410±0.270	0.095-0.815
Cricket Frog + Treefrog	May	2013	$p(\text{moonshine})$	0.53	-	0.510±0.120	0.325-0.688
Cricket Frog + Treefrog	June	2013	$p(\text{airtemp})$	0.38	+	0.000±0.000*	0.000-1.000*
Cricket Frog + Chorus Frog	May	2013	$p(.)$	0.97	NA	0.500±0.300	0.119-0.881
Cricket Frog + Chorus Frog	June	2013	$p(\text{time})$	0.85	+	0.051±0.140	0.001-0.860
W. Toad + Leopard Frog	May	2013	$p(.)$	0.45	NA	0.530±0.160	0.290-0.763
W. Toad + Leopard Frog	June	2013	$p(\text{H2O+air+wind+moon})$	0.96	W(+), A(-), W(+), M(-)	0.220±0.072	0.128-0.362
Treefrog + Chorus Frog	May	2013	$p(.)$	0.48	NA	0.650±0.130	0.414-0.825
Treefrog + Chorus Frog	June	2013	$p(\text{moonshine})$	0.56	-	0.140±0.120	0.028-0.470
Cricket +Treefrog + Leopard Frog	May	2013	$p(.)$	0.91	NA	0.700±0.140	0.427-0.875
Cricket +Treefrog + Leopard Frog	June	2013	$p(\text{airtemp})$	0.30	+	0.460±0.076	0.341-0.586
Chorus +Treefrog + Leopard Frog	May	2013	$p(.)$	0.57	NA	0.590±0.240	0.226-0.878
Chorus +Treefrog + Leopard Frog	June	2013	$p(\text{time})$	0.58	+	0.000±0.000*	0.000-0.487*
Toad+Cricket+Treefrog+ Leopard	May	2013	$p(.)$	0.72	NA	0.290±NA	NA

Figure 4.1. Graphs of the probability of detection for the top detection models of co-occurring adult amphibians. The graphs are for A) Blanchard's Cricket and Boreal Chorus Frog adults in June 2013, B) Gray Treefrog and Blanchard's Cricket Frog adults in May 2013, C) Blanchard's Cricket, Gray Treefrog, and Plains Leopard Frog adults in June 2013, D) Boreal Chorus, Gray Treefrog, and Plains Leopard Frog adults in June 2013, E) Gray Treefrog and Boreal Chorus Frog adults in June 2013, and F) Blanchard's Cricket and Gray Treefrog adults in June 2013.



Occupancy

There were 12 models in the occupancy model set for adult anurans (Table 4.1). Ideally, these would have been conducted for all seven species combinations, all months of April, May, and June, and both years 2012 and 2013, for a total of 42 potential model sets. None of the species combinations had sufficient data to run for all months and both years. Thus, the models were run 14 times and there were 14 top models and confidence sets. The confidence sets contained between 1 to 8 models (Appendix C). Overall, occupancy was affected by, aquatic vegetation, slope, distance to the nearest wetland, land cover, bend, and ephemeral. Null was the top model 1 of the 14 model runs, but was in the confidence set 12 times (Table 4.4, Appendix C). Below I address those covariates that were in the confidence sets as well as individual species.

Aquatic Vegetation

The model for aquatic vegetation was among the occupancy confidence sets 12 times and was a top model four times. All twelve models in the confidence sets, including the four top models, had positive parameter estimates. This indicated that the probability of occupancy was higher when aquatic vegetation was present (Table 4.4, Appendix C, Figure 4.2).

Slope

The slope model was among the occupancy confidence sets 11 times and was the top model three times. For six of the 11 models among the confidence sets, including the three top models, the parameter estimate was negative. Therefore, the probability of occupancy decreased as the slope percentage increased (Table 4.4, Appendix C, Figure 4.2).

Distance to the Nearest Wetland

The model for distance to the nearest wetland was among the occupancy confidence sets 11 times and was the top model three times. For eight of the 11 models in the confidence sets, including the three

top models, the parameter estimate was negative. Thus, the probability of occupancy decreased as the distance to the next wetland increased (Table 4.4, Appendix C, Figure 4.2).

Land Cover

The model for land cover (agriculture + forest) was among the occupancy confidence sets six times and was the top model once (Appendix C, Table 4.4). The top model and three of the other land cover models did not produce standard errors. However, all of the models but one had a positive parameter estimate for agriculture and a negative parameter estimate for forest. This suggests that occupancy was higher when wetlands were surrounded by agriculture.

Bend

The model for bend was among the occupancy confidence sets seven times and was the top model once. Bend was the top model for Woodhouse's Toad (*Anaxyrus woodhousii*) and Plains Leopard Frog (*Lithobates blairi*) in May 2013 and it had positive parameter estimates for Hamburg Bend and Langdon Bend. This indicates that both of these species were present at Hamburg and Langdon Bend, but not Kansas Bend. For the other six models the anurans were present at Hamburg Bend five times, Kansas Bend once, and Langdon Bend once. However, for one of the models with just a positive parameter at Hamburg Bend, there was not enough data to produce standard errors. This indicates that more species occurred in wetlands together at Hamburg Bend than the other bends (Table 4.4, Appendix C, Figure 4.2).

Ephemeral

The model for ephemeral was among the occupancy confidence sets 11 times and was the top model once. For eight of the 11 models in the confidence sets, including the top model, the parameter estimate was positive. Therefore, the anurans were more likely to be present at ephemeral wetlands (Table 4.4, Appendix C, Figure 4.2).

Other Models

The model for distance to the nearest wetland plus land cover was among the occupancy confidence sets twice, but was never a top model. Given its rarity in the confidence sets and that it never had an AICc weight above 0.09, it is unlikely that this model affected occupancy. Size was among the confidence sets once and was never the top model. It was the third model for Blanchard's Cricket Frog and Boreal Chorus Frog in May 2013. However, it only had 0.02 AICc weight and is unlikely to affect occupancy. The model for adjacent vegetation was among the confidence sets once with AICc weight 0.05. With such a low AICc weight and its marginal presence among the confidence sets, it is unlikely that it affected occupancy. The model for adjacent vegetation plus aquatic vegetation was among the confidence sets twice and never held more than 0.07 AICc weight. With such a low AICc weight and marginal presence among the confidence sets, it is unlikely that it affected occupancy. The global model was the only model that was never among the confidence sets (Appendix C).

Blanchard's Cricket Frog (*Acris blanchardi*) + Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*)

There were seven models in the confidence set for June 2012. However, the top model was ephemeral with AICc weight 0.72 (Table 4.4, Appendix C). Ephemeral had a positive parameter estimate and so the probability of occupancy was higher when a wetland was ephemeral than when it was not (Table 4.4, Appendix C, Figure 4.2).

There were six models in the confidence set for May 2013 and only one model in the confidence set for June 2013. (Appendix C). The top model for both was aquatic vegetation with AICc weight 0.28 for May 2013, and 0.96 for June 2013 (Table 4.4). The parameter estimate for both models was positive and so the probability of occupancy was highest when aquatic vegetation was present (Appendix C, Figure 4.2). The model for the distance to nearest wetland was the second model for May 2013 with AICc weight 0.24, which was followed by the null model with an AICc weight of 0.23. The rest of the models hold AICc weights below 0.08. Therefore it is probable that the model for distance to the nearest

wetland also affected the occupancy in May 2013. It had a negative parameter estimate and so the probability of occupancy decreased as the distance to the next wetland increased (Table 4.5, Appendix C).

Table 4.5. Blanchard's Cricket Frog and Gray Treefrog model equation for distance to the nearest wetland. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
May	2013	Distance to Nearest Wetland	$\Psi = 1.600 + -0.002x$

Blanchard's Cricket Frog (*Acris blanchardi*) + Boreal Chorus Frog (*Pseudacris maculata*)

There were five models in the confidence set for May 2013 (Appendix C). However, the top model was land cover with AICc weight 0.81 (Table 4.4). This did not produce standard errors, but the parameter estimate was positive for agriculture and negative for forest (Appendix C)

There were two model in the confidence set for June 2013 and the top model was distance to the nearest wetland with AICc weight 0.83 (Table 4.4, Appendix C). The parameter estimate for this model was negative and therefore the probability of occupancy decreased as the distance to the next wetland increased (Table 4.6, Appendix C, Figure 4.2).

It should be noted that while these are the two smallest frogs heard calling at these sites, they do not have similar natural history. The most significant difference being that the Boreal Chorus Frogs starts calling earlier, leading to little call overlap. However, the top model with 0.83 AICc weight was distance to the nearest wetland. Given that these are the two smallest species with the shortest dispersal ability this is something that is clearly important in determining their occupancy of a wetland.

Table 4.6. Blanchard's Cricket Frog and Boreal Chorus Frog model equation for the distance to the nearest wetland. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
June	2013	Distance to Nearest Wetland	$\Psi = 2.090 + -0.009x$

Woodhouse's Toad (*Anaxyrus woodhousii*) + Plains Leopard Frog (*Lithobates blairi*)

There were five models in the confidence set for May 2013 (Appendix C). The top model was bend with AICc weight 0.33, followed closely by ephemeral with AICc weight 0.24, aquatic vegetation with AICc weight 0.15, slope with AICc weight 0.12, and null with AICc weight 0.11 (Table 4.4, Appendix C). Given the low AICc weight of the top model and how close the AICc weights of the others are to it and each other, it is probable that they also affected occupancy. The model for bend had a positive parameter estimate for both Hamburg and Langdon Bend and a negative parameter estimate for Kansas Bend. Therefore, these two species were not found together at Kansas Bend, but were present together at Hamburg and Langdon Bend (Appendix C, Figure 4.2). The ephemeral and aquatic models had positive parameter estimates and higher probability of occupancy when the wetland was ephemeral and had aquatic vegetation respectively (Appendix C). The model for slope had a negative parameter estimate and therefore the probability of occupancy decreased as the percent slope increased (Appendix C).

There were four models in the confidence set for June 2013 (Appendix C). The top model for June was null with AICc weight 0.5 (Table 4.4). It is possible that the other models could have affected occupancy, but given that the top model is the null model, it is unlikely (Appendix C).

Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*) + Boreal Chorus Frog (*Pseudacris maculata*)

There were six models in the confidence set for May 2013 (Appendix C). The top model was distance to nearest wetland with AICc weight 0.30, followed by aquatic vegetation with AICc weight 0.24 and null with AICc weight 0.24 (Table 4.4, Appendix C). The model for distance to the nearest wetland had a negative parameter estimate and therefore the probability of occupancy decreased as the distance to the nearest wetland increased (Table 4.7, Appendix C, Figure 4.2). Given the low AICc weight of the top model, it is probable that the model for aquatic vegetation also affected occupancy. The next model was null and the models after this all had AICc weights 0.06 and under and were unlikely to have affected

occupancy. The model for aquatic vegetation had a positive parameter estimate and therefore the probability of occupancy was highest when aquatic vegetation was present (Appendix C).

There were eight models in the confidence set for June 2013 (Appendix C). The top model was aquatic vegetation with AICc weight 0.32, followed by null with AICc weight 0.16 (Table 4.4, Appendix C). Given that the second model is the null model, it is unlikely that any of the others affected occupancy. The model for aquatic vegetation had a positive parameter estimate and thus the probability of occupancy was highest when aquatic vegetation was present (Appendix C, Figure 4.2).

Table 4.7. Gray Treefrog and Boreal Chorus Frog model equation for distance to the nearest wetland. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
May	2013	Distance to Nearest Wetland	$\Psi = 1.410 + -0.003x$

Blanchard's Cricket Frog (*Acris blanchardi*) + Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*) + Plains Leopard Frog (*Lithobates blairi*)

There were seven models in the confidence set for May 2013 (Appendix C). The top model was aquatic vegetation with AICc weight 0.26, followed by the distance to the nearest wetland with AICc weight 0.24, and null with AICc weight 0.19 (Table 4.4, Appendix C). All of the models after null were unlikely to have affected occupancy, but the model for distance to the nearest wetland had an AICc weight close to the top model and was above the null model. The model for aquatic vegetation had a positive parameter estimate and therefore the probability of occupancy was highest when aquatic vegetation was present (Appendix C, Figure 4.2). The second model, distance to the nearest wetland, had a negative parameter estimate and the probability of occupancy decreased as the distance increased (Table 4.8, Appendix C).

There were six models in the confidence set for June 2013 (Appendix C). The top model was slope with AICc weight 0.39, followed by aquatic vegetation with AICc weight 0.35, and null with AICc

weight 0.08 (Table 4.4, Appendix C). All of the models after null had very low AICc weights and were unlikely to have affected occupancy. The model for slope has a negative parameter estimate and so the probability of occupancy decreased as the slope percentage increased (Table 4.8, Appendix C, Figure 4.2). The model for aquatic vegetation had a positive parameter estimate and thus the probability of occupancy was highest when aquatic vegetation was present (Appendix C).

Table 4.8. Blanchard's Cricket Frog, Gray Treefrog, and Boreal Chorus Frog model equations for slope and distance to the nearest wetland. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
May	2013	Distance to Nearest Wetland	$\Psi = 1.610 + -0.002x$
June	2013	Slope	$\Psi = 2.530 + -11.240x$

Boreal Chorus Frog (*Pseudacris maculata*) + Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*) + Plains Leopard Frog (*Lithobates blairi*)

There were six models in the confidence set for May 2013 (Appendix B). The top model was distance to the nearest wetland with AICc weight 0.30, followed by aquatic vegetation with AICc weight 0.24 and null with AICc weight 0.24 (Table 4.4, Appendix C). All of the models after null were unlikely to have affected occupancy, but the model for aquatic vegetation had an AICc weight close to the top model and was equal to the null model. The model for distance to the nearest wetland had a negative parameter estimate and so the probability of occupancy decreased as the distance increased (Table 4.9, Appendix C, Figure 4.2). The aquatic vegetation model had a positive parameter estimate therefore, the probability of occupancy was highest when aquatic vegetation was present (Appendix C).

There were seven models in the confidence set for June 2013 (Appendix C). The top model was slope with AICc weight 0.34, followed by null with AICc weight 0.20 (Table 4.4). The model for slope had a negative parameter estimate and therefore the probability of occupancy decreased as the percent slope increased (Table 4.9, Figure 4.2). All of the models after null were unlikely to have affected occupancy.

Table 4.9. Boreal Chorus Frog, Gray Treefrog, and Plains Leopard Frog model equations for slope and distance to the nearest wetland. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
May	2013	Distance to Nearest Wetland	$\Psi = 1.680 + -0.003x$
June	2013	Slope	$\Psi = 15.800 + -54.000x$

Woodhouse's Toad (*Anaxyrus woodhousii*) + Blanchard's Cricket Frog (*Acris blanchardi*) + Gray Treefrog Complex (*Hyla chrysoscelis*, *Hyla versicolor*) + Plains Leopard Frog (*Lithobates blairi*)

There were six models in the confidence set for May 2013 (Appendix C). The top model was slope with AICc weight 0.45, followed by null with AICc weight 0.14 (Table 4.4, Appendix C). The model of slope had a negative parameter estimate and therefore the probability of occupancy decreased as the slope percentage increased (Table 4.10, Figure 4.2). Given that the second model is null and it has a low AICc weight, it is unlikely that any of the other covariates affected occupancy (Appendix C).

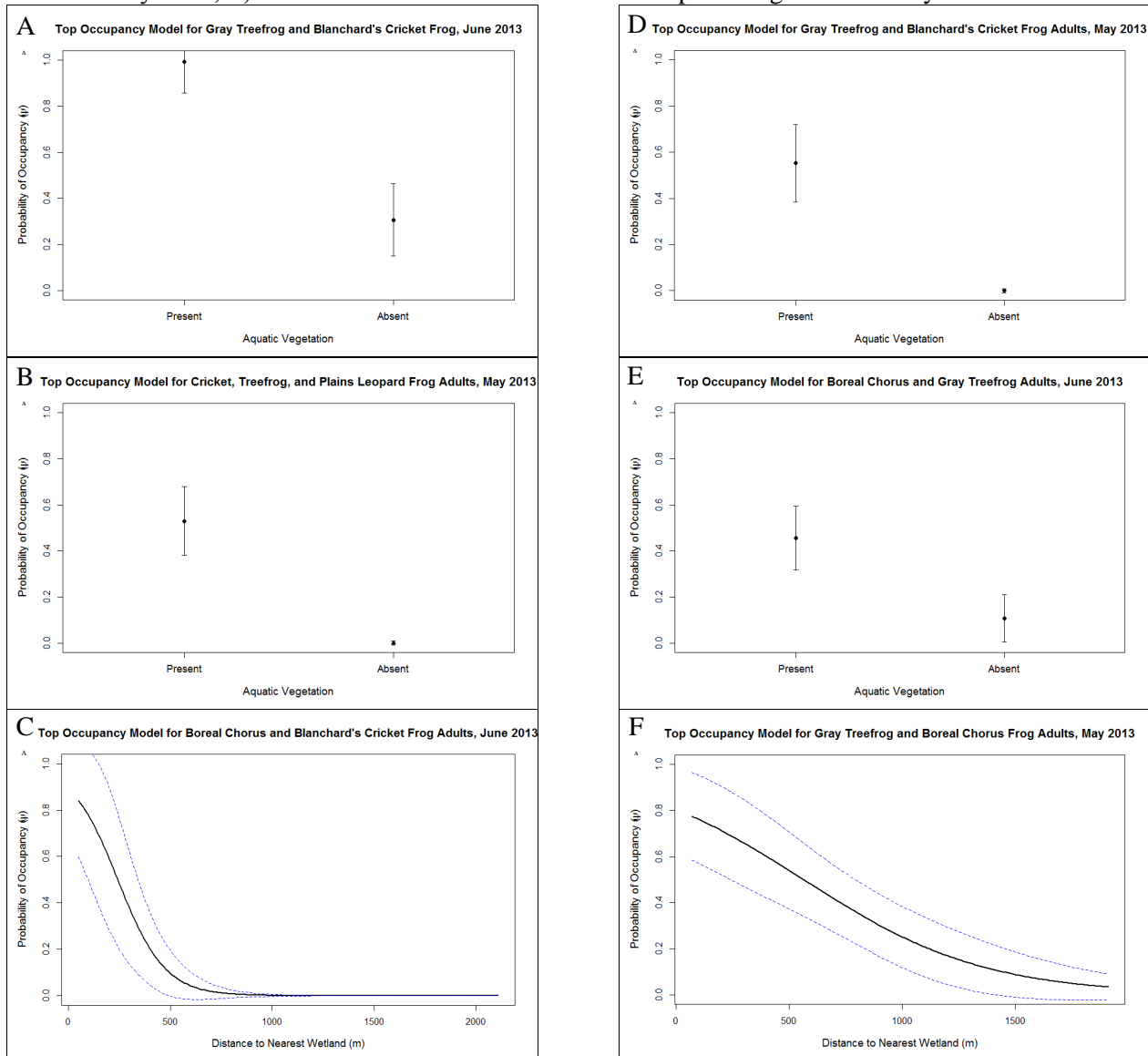
Table 4.10. Woodhouse's Toad, Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog model equation for slope. Where $\Psi = B_0 + B_1x$, Ψ is occupancy, B_1 is the parameter estimate, x is a covariate amount, and B_0 is the intercept.

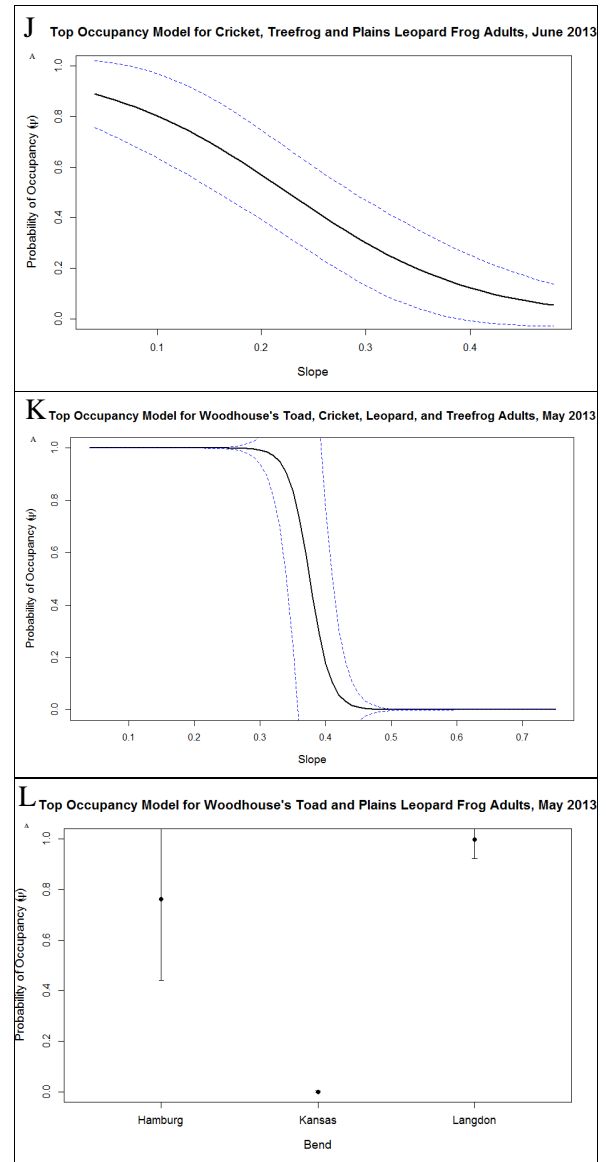
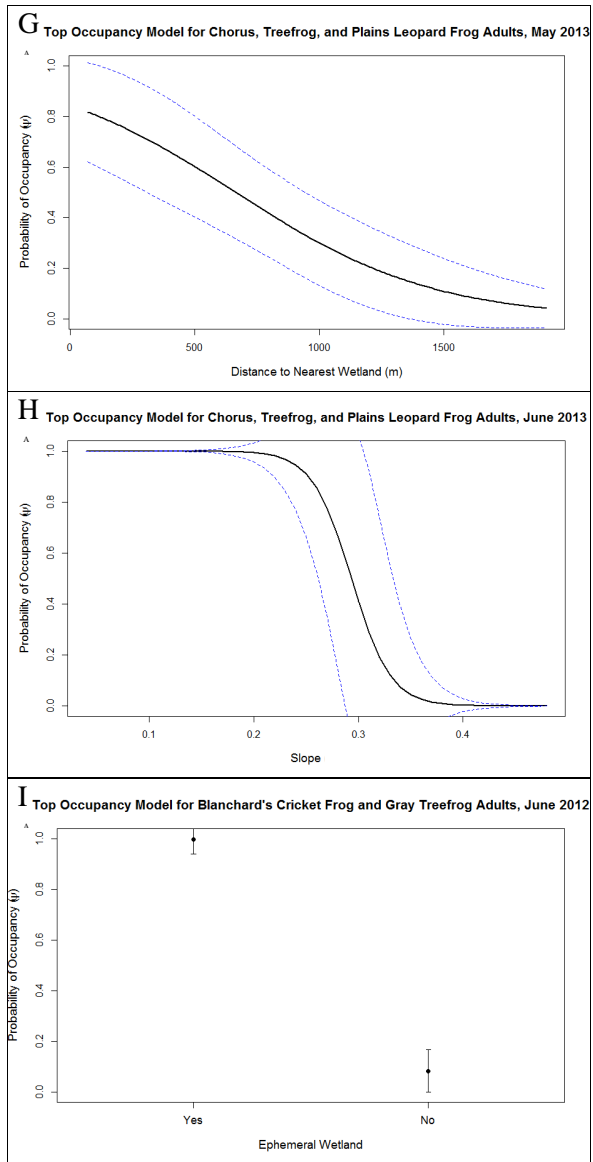
Month	Year	Model	Equation ($\Psi = B_0 + B_1x$)
May	2013	Slope	$\Psi = 23.800 + -63.300x$

Table 4.4. Top occupancy models for co-occurring adult anuran species in restored wetlands along the Missouri River in southeast Nebraska. A set of occupancy models was assessed for every biologically important species combination with naïve occupancy ≥ 0.10 during each month of April, May, and June for both 2012 and 2013. The top occupancy models contain the measured covariates that affected frog call occupancy the most. The beta parameter estimate ($\hat{\beta}$) shows whether the covariates in the occupancy models had a positive or negative relationship with occupancy. For $\hat{\beta}$ H represents Hamburg Bend, K represents Kansas Bend, and L represents Langdon Bend, A represents agriculture, and F represents forest. The occupancy estimate ($\hat{\Psi}$) also includes the standard error. For the last column CL refers to confidence limits.

Species	Month	Year	Covariate/s	AICcWt	$\hat{\beta}$	$\hat{\Psi}$	95% CL
Cricket Frog + Treefrog	June	2012	$\Psi(\text{ephemeral})$	0.72	+	0.880 \pm 1.740	0.000-1.000*
Cricket Frog + Treefrog	May	2013	$\Psi(\text{aquatic})$	0.28	+	0.015 \pm 0.330	0.000-1.000*
Cricket Frog + Treefrog	June	2013	$\Psi(\text{aquatic})$	0.96	+	0.890 \pm 0.960	0.000-1.000*
Cricket Frog + Chorus Frog	May	2013	$\Psi(\text{agr+forest})$	0.81	A(+), F(-)	NA	NA
Cricket Frog + Chorus Frog	June	2013	$\Psi(\text{distwet})$	0.83	-	0.001 \pm 0.003	0.000-0.172*
W. Toad + Leopard Frog	May	2013	$\Psi(\text{bend})$	0.33	H(+), K(-), L(+)	0.120 \pm 4.550	0.000-1.000*
W. Toad + Leopard Frog	June	2013	$\Psi(.)$	0.50	NA	0.300 \pm 0.092	0.172-0.467
Treefrog + Chorus Frog	May	2013	$\Psi(\text{distwet})$	0.30	-	0.260 \pm 0.130	0.097-0.520
Treefrog + Chorus Frog	June	2013	$\Psi(\text{aquatic})$	0.32	+	0.240 \pm 0.120	0.100-0.474
Cricket +Treefrog + Leopard Frog	May	2013	$\Psi(\text{aquatic})$	0.26	+	0.800 \pm 0.190	0.374-0.966
Cricket +Treefrog + Leopard Frog	June	2013	$\Psi(\text{slope})$	0.39	-	0.400 \pm 0.170	0.172-0.686
Chorus +Treefrog + Leopard Frog	May	2013	$\Psi(\text{distwet})$	0.30	-	0.380 \pm 0.170	0.159-0.672
Chorus +Treefrog + Leopard Frog	June	2013	$\Psi(\text{slope})$	0.34	-	0.860 \pm 0.330	0.068-0.998
Toad+Cricket+Treefrog+ Leopard	May	2013	$\Psi(\text{slope})$	0.45	-	0.230 \pm 0.700	0.000-0.995*

Figure 4.2. Graphs of the probability of occupancy for the top occupancy models of co-occurring adult amphibians. The graphs are for A) Gray Treefrog and Blanchard's Cricket Frog adults in June 2013, B) Blanchard's Cricket, Gray Treefrog, and Plains Leopard Frog adults in May 2013, C) Boreal Chorus and Blanchard's Cricket Frog adults in June 2013, D) Gray Treefrog and Blanchard's Cricket Frog adults in May 2013, E) Boreal Chorus and Gray Treefrog adults in June 2013, F) Gray Treefrog and Boreal Chorus Frog adults in May 2013, G) Boreal Chorus, Gray Treefrog, and Plains Leopard Frog adults in May 2013, H) Boreal Chorus, Gray Treefrog, and Plains Leopard Frog adults in June 2013, I) Blanchard's Cricket and Gray Treefrog adults in June 2012, J) Blanchard's Cricket, Gray Treefrog, and Plains Leopard Frog adults in June 2013, K) Woodhouse's Toad, Blanchard's Cricket, Plains Leopard, and Gray Treefrog adults in May 2013, L) and Woodhouse's Toad and Plains Leopard Frog adults in May 2013.





DISCUSSION

To determine what a successful wetland is and create the wetland restoration guidelines it is essential to consider the needs of as many amphibian species as possible. While this could be done with the overall results of many individual species, it is more useful and efficient to use co-occurring species. In fact, it has been previously shown (Chapter 3) that with single species occupancy analysis both frog call surveys and tadpole dip net surveys are needed to build wetland guidelines that provide for all life stages. The main difference between the single species analysis for frog call surveys and tadpole dip net surveys was the prevalence of the model distance to the nearest wetland. This model was the most supported occupancy model for the tadpoles, but was of uncertain consideration for the adults. However, this model is well supported in the analysis of co-occurring species. Therefore, not only is the analysis more efficient, but because only frog call surveys are needed the data collection is less time consuming.

Aquatic vegetation was the model that was the top model most often. For every combination it had a positive parameter estimate and the probability of occupancy was highest when aquatic vegetation was present. Aquatic vegetation provides protection from predators for both adults and larvae amphibians, it is used to anchor eggs, and for some species it is used as a place to call from. Thus, it is not surprising that its presence is an important part of a successful wetland. However, more research should be done to specify what kind of aquatic vegetation is best and how this varies regionally.

The model for slope was tied with distance to the nearest wetland for the second most frequent top model. For all the top models and most of the models in the confidence set the probability of occupancy decreased as the percent slope increased. A more specific idea of what slope percentage to use can be found by using the equations made from the parameter estimates (Table 4.5-4.10). It is as easy as picking a representative species combination with slope as a top model, such as Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard frog in June 2013, and then entering the desired occupancy for y and

solving for the percent slope (x). However, assessment of the graphs of the top occupancy models reveals that the species combinations have higher occupancies when the percent slope is less than 0.30.

For all of the top models and most of the distance to the nearest wetland models in the confidence set the probability of occupancy decreased as the distance to the nearest wetland increased. Amphibians differ in size and dispersal ability, but they all tend to exist as metapopulations, especially in a dynamic ecosystem like the Missouri River Flood Plains. Therefore, having other wetlands within their dispersal range not only aids in genetic distribution, but also provides habitat in situations like drought. The models for distance to the nearest wetland can provide a better idea of how close other wetlands should be (Table 4.5-4.10). The process is the same as with slope enter the desired occupancy for y and solve for the distance or x. However, occupancy is highest when the next wetland is within 500 m (Figure 4.2). All of the species combinations that have distance to the nearest wetland as the top model include the Boreal Chorus Frog, one of the smallest species. The most severe drop off is for the combination of the two smallest species, the Boreal Chorus Frog and the Blanchard's Cricket Frog in June 2013. It is possible that this is important to the larger species as well, but because most wetlands have other wetlands within their larger dispersal distances it did not show up as a top model. However, a successful wetland provides habitat for a diverse range of species and thus it is important that other wetlands are close enough that all species can travel to them.

The model for ephemeral was the top model only once, but it was among the confidence sets 11 times. For most of those the probability of occupancy was highest when a wetland was ephemeral. It is possible that these results were skewed by the drought in 2012 and the beginning of 2013. If more ephemeral wetlands were holding water it could have been a top model more often. Certain variables could be more important during a drought or a flood, than in a normal year. To adequately determine if this is true data would need to be gathered for decades and include several floods, droughts, and normal seasons.

Therefore, a successful wetland contains aquatic vegetation, has a shallow slope (less than 0.30), is ephemeral, and has at least one other wetland nearby (within 500 m). These may be simple and general guideline, but they do give a starting point for creating successful wetlands on a flood plain. Amphibians have been used to inform wetland restoration in many different ways. Examples include using amphibian distribution to inform buffer zones around wetlands, looking at how adjacent land use effects amphibian species richness, and analyzing behavioral responses and survival to determine whether local wetland sites should be designed to support metapopulations or patchy populations (Rittenhouse and Semlitsch 2007; Houlahan and Findlay 2003; Pentranka and Holbrook 2006). Occupancy analysis has often been used to compare different hypothesis about what affects amphibian distribution and detection (Mazerolle et al. 2005; Muths et al. 2005; Weir et al. 2005; Bailey et al. 2004). However, amphibian occupancy analysis has rarely been used to specifically inform wetland restoration. Balas et al. used amphibian occupancy to determine the importance of season wetlands in conservation grasslands and Hellman conducted a similar single species analysis of the Nebraska wetland sites for the first two years of the overall amphibian monitoring project. She found that shallow slope was important for both adult and tadpole amphibian occupancy (Balas et al. 2012; Hellman 2013). My analysis has produced additional wetland variables that define a successful wetland, used a novel co-occurrence method that produced a similar definition of a successful wetland as the combined results of the single species frog call and tadpole dip net survey analyses without the variation between species, months, years, and survey techniques (Chapter 3), and has also provided equations for slope and distance to the nearest wetland that apply to more than one species at a time and therefore are more useful in informing wetland restoration. There is still a disparity between what is legally required to create or restore a wetland and their ability to replace the lost structure and functions (Streever 1999). Using an occupancy modeling analysis of co-occurring species to inform wetland restoration provides a way to determine if restored wetlands are providing some of the lost function and structure of the original wetlands. The definition of a successful wetland gained from this study provides both information that will lead to more successfully restored

wetlands, and a framework for more specific wetland restoration guidelines. Thus, continued monitoring in an adaptive management framework is needed to build upon the wetland restoration guidelines.

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CHAPTER 5: USING FUNCTIONAL CONNECTIVITY AND ANURAN DISPERSAL DISTANCES TO INFORM WETLAND RESTORATION

INTRODUCTION

Wetlands play a significant role in ecosystem function. They contribute to nutrient cycling, are essential to the hydrological cycle, and provide habitat for plants, fish and wildlife species (Mitsch and Gosselink 2007). Unfortunately, much of this critical habitat has been altered or destroyed (Kryzick 1998 and Leja 1998). Along the Missouri River the main cause for wetland habitat loss was the Bank Stabilization and Navigation Project implemented in 1912. It was responsible for the channelization of the Missouri River and led to the loss of over 500,000 acres of Missouri River habitat (U.S. Army Corp of Engineers Kansas City and Omaha Districts 2003). These alterations have blocked wetlands from annual flooding that used to maintain variable, but connected wetland complexes.

The importance of the Missouri River for navigation has been steadily decreasing since 1977, especially above Kansas City (MRRP pamphlet 2009). Concomitant with this decrease has been an increase in the recognition of the importance of wetland and wetland habitats (Mitsch and Gosselink 2007, Moll and Moll 2004, Zedler and Kercher, 2005). In addition to the many other ecosystem services wetlands and undiked floodplains can reduce flooding and flood impacts (Bakacsi et al. 2011). Climate variability is expected to increase in the relatively near future, and this variability is expected to include more extreme weather events (IPCC 2003). The recognition of the importance of wetlands has guided U.S. government policy, and led to legislation such as the Clean Water Act of 1986 and the Water Resources Development Act of 1986 (U.S. Army Corp of Engineers 2006).

After the enactment of the Clean Water Act of 1986 there has been effort by the United States Army Corps of Engineers (USACE) to enforce the policy of “no net loss.” From 1996 until 2005 there was an estimated net gain of 10,000 ha/yr of wetland and associated uplands habitat in the United States. However, it is unclear how successful these created, restored, enhanced, or preserved wetlands are in

terms of restoring function and habitat. There is little information on what ecosystem functions were lost and what functions were gained (Mitsch and Gosselink 2007). Although restoration techniques continue to improve, there are still no overall guidelines for building a functional wetland outside of engineered hydrological changes. Because of ongoing restoration efforts, increasing threats to biodiversity and to wetlands, and lack of ecologically-based performance guidelines, there needs to be concerted and quantifiable efforts to assess the success of restored riverine wetland systems, so that general requirements can be put into effect.

The Water Resources Development Act of 1986 led to the creation of the Missouri River Fish and Wildlife Mitigation Project (MRFWMP). The MRFWMP is dedicated to preserving existing habitat and restoring lost habitat that supports native vegetation, fish, and wildlife species within the flood plain. The requirements of the MRFWMP include monitoring restored habitat. This project is part of those monitoring efforts, but it also includes linking monitoring with hypothesis testing in an adaptive framework. The MRFWNP monitoring effort was expanded in 2007 to include reptiles and amphibians. In 2008 a multi-institutional project funded by the USACE was initiated in four states (Kansas, Missouri, Iowa, and Nebraska). The main goal of the overall project is to assess the success of previously restored wetlands and to create wetland restoration guidelines for future use.

While restoring successful individual wetlands is important, flood plain wetlands are highly variable in space and time, and because of this restoration should be conducted for complexes rather than individual wetlands. The channelization, damming, and flood control structures implemented on the Missouri River since 1912 have disconnected the wetlands from the annual flooding that maintained the variable but connected wetland complexes. This, combined with the increasing climate variability, suggests that creating resilient wetland complexes is essential to wetland restoration success on a flood plain. One way to determine the resilience of the wetland complexes is to analyze the functional connectivity. Functional connectivity can be used to determine the connectivity of any habitat patches.

In this case, functional connectivity is how connected the individual wetlands are to one another, which is dependent on the amphibians' dispersal capability. Functional connectivity is used to inform the placement of habitat corridors, indicate where to focus management effort, acquire new habitat for conservation, and help determine the best design for preserves or wetland complexes (Dodd 2010; Calabrese and Fagan 2004).

Within the five years (2009-2013) that this study was conducted there was a historic flood and a historic drought, which provided extreme conditions to analyze wetland connectivity under variable conditions. To assess connectivity under drought and flood conditions, I conducted a functional connectivity analysis using anuran dispersal distances. Anuran dispersal distances were chosen to assess wetland success because they are declining globally (Glascon et al. 2005; Pentranka and Holbrook 2006; Collins and Crump 2009; Crump 2010), they are important parts of the terrestrial and aquatic environments (Collins and Crump 2009; Vitt and Caldwell 2009; Collins and Halliday 2005), and they are good indicators of wetland restoration success (Bowers et al. 1998; Collins and Halliday 2005; Welsh and Ollivier 1998). Specifically, their biphasic life cycle and permeable skin make them uniquely sensitive to conditions in the water and the surrounding terrestrial environment. Wetlands are complex ecosystems that are hard to define and are made of both aquatic and terrestrial components. They are easily altered or degraded by changes in either. Thus, anurans, whose biphasic life cycle and permeable skin makes them dependent on these intricate and essential ecosystems, are ideal indicators of their success. Additionally, many anuran species function as metapopulations and need connectivity to survive disturbances like drought. Thus, if the restored wetlands are functionally connected they are more likely to contribute to ecosystem function and are more likely to maintain viable populations of anurans despite weather and climate variability.

I analyzed the functional connectivity of three restored wetland complexes that are located adjacent to Missouri River in Nebraska from Nebraska City to Nemaha. I determined functional connectivity in drought conditions and more 'normal' conditions for three different anuran dispersal

distances, short range dispersers (500 m), medium range dispersers (1500 m), and long range dispersers (3000 m). My objectives were to determine if the wetland complexes are functionally connected in both drought and normal conditions for all species, if the complexes are not functionally connected for all situation to then determine why they are not, identify critical wetlands for protection, use this information to inform wetland restoration, and to provide a useful methodology that busy managers can use to inform wetland restoration in the future. Because anurans are good indicators of wetland success, the information gained from this analysis adds to the definition of wetland success and aids wetland restoration guidelines.

METHODS

Study Sites

The sites used in this study consist of 55 restored wetlands distributed among three wetland complexes (river bends) between Nebraska City, NE and Nemaha, NE. Hamburg Bend is located 8 miles southeast of Nebraska City, Nebraska in Otoe County. It is ~638 ha of former agricultural land that was purchased by the USACE between 1993 and 2004. It is adjacent to the right descending bank of the Missouri River between river miles 552 to 556. Due to the alterations of the Missouri River Bank Stabilization and Navigation Project (BSNP), the side channels where closed and this allowed the land to accrete. In response a side channel was constructed as well as several back water areas (U.S. Army Corp of Engineers 2006).

Kansas Bend is located 3 miles east of Peru, Nebraska in Nemaha County. It is composed of ~427 ha of former agricultural land and is separated into two areas by privately owned farmland. It was purchased by the Corps between 1993 and 1999 and is adjacent to the right descending bank of the Missouri River between river miles 544 to 547. Due to the activities of the BSNP several side channels were closed using dikes and revetments. Two side channels have been reopened.

Langdon Bend is located 3 miles south of Brownville, Nebraska, in Nemaha County. It is comprised of ~675 of former agricultural land that was purchased by the USACE between 1994 and 2003. It is the area adjacent to the right descending bank of the Missouri River between river miles 520 and 532. Due to the Missouri River Bank Stabilization and Navigation Project the chute at Langdon Bend was closed and the side channel was cut off. The old channel could not be opened due to its proximity on the upstream end to the Cooper Nuclear Power Plant. Thus, the newly constructed channel is connected to the river at the outlet, but stops before meeting the river at the upstream end. The Corps in conjunction with the Nebraska Game and Parks Commission designed the wetland complex that was built on the west side of the levee. There are about 89 ha of wetlands that were constructed starting in July 2008 and ending May 2009.

Field Methods

Frog call surveys were conducted twice at every wetland holding water during a two week period during the months of April, May, and June in 2012 and 2013. Tadpole surveys were conducted in the same manner during the second week of the month. A severe drought in 2012 affected the number of wetlands holding water. This drought continued into 2013 and the number of wetland did not increase until the copious amount of rain in May finally saturated the soil enough to fill a majority of the wetlands in June 2013. Thus, the lowest number of wetlands surveyed was 20 wetlands during the first two weeks of May 2013 and the highest number was 43 during the first two weeks of June 2013. The wetlands present during the first two weeks of May 2013 were used to represent a drought situation and the wetlands present during the first two weeks of June 2013 were used to represent a moderate or normal situation.

Analysis

A majority of the wetland polygons were created by walking around each wetland with a hand held GPS unit (Trimble XH) with sub-meter accuracy, though highly variable water levels and wetland boundaries that shifted following floods meant realized accuracy was less than this. The polygons

representing the drought period and those representing a moderate year were merged separately in ArcGIS. The wetlands present at Hamburg, Kansas, and Langdon bend for both the drought and moderate were also merged to create six different layers. The area for each wetland was calculated and added as a field to each layer. Each of the merged layers was converted from degrees to UTM so the units would be meters.

The Conefor ArcGIS extension was used to transform each of the eight merged layers into node and distance inputs for Conefor, a landscape connectivity software program (Saura and Torné 2009). The inputs for all eight situations were conducted with a 500 m, 1500 m, and 3000 m dispersal distance. These were used to represent three different anuran dispersal capabilities. The 500 m dispersal distance represents small native species like the Blanchard's Cricket Frog (*Acris blanchardi*) that have short dispersal capabilities. The 1500 m dispersal range represents medium dispersers like the Plains Leopard Frog (*Lithobates blairi*), and the 3000 m dispersal range represents long range dispersers such as the Woodhouse's Toad (*Anyraxus woodhousii*) (Smith and Green 2005).

Conefor provides many different connectivity indices to analyze the overall connectivity and the importance of individual patches (nodes), including options for both binary (connected or not connected) and probabilistic analysis. The best performing binary indices is the integral index of connectivity (IIC), which considers both graph theory and habitat availability (Pascual-Hortal and Saura 2006, Saura and Pascual-Hortal 2007). Habitat availability allows for the inclusion of the patch area (or another patch attribute), to account for the connectivity that occurs within the patch itself. For this study the binary indices number of links (NL), the number of compounds (NC), and the Equivalent Connectivity (EC (IIC)) were used to determine overall functional connectivity. The EC (IIC) maintains all the desirable properties of the IIC, but has several advantages. It avoids the low metric values that can be obtained by the IIC when the overall area is much larger than the patches, it has the same units as the node attributes, it doesn't require specification of the overall area, and the relative variation after a spatial change can be directly compared with the variation in total habitat area (Saura et al. 2011a, 2011b).

$$A) IIC = \frac{\sum_{i=1}^n \sum_{j=1}^n \frac{a_i a_j}{1 + nl_{ij}}}{A_L^2} \quad B) EC(IIC) = \sqrt{\sum_{i=1}^n \sum_{j=1}^n \frac{a_i a_j}{1 + nl_{ij}}}$$

Equation A) is for the integral index of connectivity (IIC). Equation B) is for the Equivalent Connectivity of the integral index of connectivity (EC (IIC)). For both equations n is the total number of nodes on the landscape, a_i and a_j are the attributes of nodes i and j , nl_{ij} is the number of links in the shortest path between patches i and j , and A_L is the maximum landscape attribute.

The node (wetland) importance to functional connectivity was determined by the dIIC, which is calculated by adding together three different indices, dIICintra, dIICflux, and dIICconnector. The index dIICintra measures the intrapatch connectivity, or the habitat that a patch provides within itself. The dIICflux measures how well the patch is connected within the landscape, but does not account for how important it is for maintaining connectivity. The dIICconnector measures how important a patch is as a connector or stepping stone (Saura and Rubio 2010).

Maps of the functional connectivity were created using gps points and the link files created by Conefor. These were used to make excel files of from and to nodes that were uploaded into ArcGIS. While points were used in the maps to represent the wetlands, the area of the wetland polygons was included in the actual analysis. The polygons weren't used in the maps because many of the wetlands were too small or too narrow to show up clearly at a complex level. In addition, mapping the connections for many of the long, narrow wetlands accurately was difficult. The common method is to use a centroid to centroid link, but anurans are unlikely to move along wetlands in this manner. Therefore, I used points to more simply visualize the results.

RESULTS

Overall, the lowest number of components (NC) was three and therefore the three bends were not connected to one another for any of the dispersal distances in drought or normal conditions (Table 5.1, Figure 5.1). The highest functional connectivity for all three wetlands combined was for the long-range dispersers (3000 m) in normal conditions. The lowest overall connectivity was for the short-range dispersers (500 m) during drought conditions (Table 5.1).

Table 5.1. Overall functional connectivity for three wetland complexes in Southeast Nebraska during drought conditions and normal conditions, and for short-range dispersers (500 m), medium-range dispersers (1500 m), and long-range dispersers (3000 m).

Condition	Distance (m)	NL	NC	EC(IIC)
Drought	500	9	10	70609
Drought	1500	22	3	85533
Drought	3000	35	3	89141
Normal	500	50	12	108999
Normal	1500	155	3	140738
Normal	3000	236	3	147214

When the wetland complexes were analyzed individually, the wetland complex with the highest functional connectivity for all distances and both conditions was Langdon Bend. It had about twice the EC(IIC) number of Hamburg Bend and about three times the EC(IIC) number of Kansas in drought conditions. In normal conditions Langdon bend still had about double the EC(IIC) of Hamburg, but it only had about one and a third the EC(IIC) of Kansas (Table 5.2-5.4). During the drought Hamburg Bend had higher functional connectivity than Kansas Bend (Table 5.2 and 5.3). However, Kansas Bend gained 13 wetlands in the normal conditions, while Hamburg Bend only gained 4 wetlands (Figure 5.2-5.7). Thus, during the normal wetland conditions Kansas Bend had higher functional connectivity than Hamburg Bend (Table 5.2 and 5.3).

Hamburg Bend functional connectivity was higher for normal conditions than for the drought conditions for all distances. The functional connectivity also increased as the dispersal distance increased

for both conditions (Table 5. 2). There was only one pair of connected wetlands and six components (groups of wetlands connected to each other but not to other groups) for the drought conditions at 500m. In normal conditions at 500 m there were 6 links, but there were still 6 components (Table 5.2, Figure 5.2). For all other situations the wetlands were connected, in varying degrees, into one component (Table 5.2, Figure 5.2-5.4).

Table 5.2. Functional connectivity of Hamburg Bend during drought conditions and normal conditions, and for short-range dispersers (500 m), medium-range dispersers (1500 m), and long-range dispersers (3000 m).

Condition	Distance (m)	NL	NC	EC(IIC)
Drought	500	1	6	29253
Drought	1500	7	1	36861
Drought	3000	11	1	37976
Normal	500	6	6	33769
Normal	1500	17	1	48973
Normal	3000	29	1	53885

The functional connectivity for Kansas Bend was also higher during the normal wetland conditions than for drought conditions. The functional connectivity increased as the dispersal distance increased for both conditions. In fact, Kansas is the only bend that has functional connectivity that is higher for normal conditions at 500 m than for drought conditions at 3000 m (Table 5.3). For drought conditions at 500m, drought conditions at 1500 m, and normal conditions at 500 m, the wetlands were not connected into one component. The rest were connected into one component, but the number of links increased as the distance increased (Table 5.3, Figures 5.5-5.7).

Table 5.3. Functional connectivity of Kansas Bend during drought conditions and normal conditions, and for short-range dispersers (500 m), medium-range dispersers (1500 m), and long-range dispersers (3000 m).

Condition	Distance (m)	NL	NC	EC(IIC)
Drought	500	4	2	15970
Drought	1500	8	2	17059
Drought	3000	14	1	18350
Normal	500	36	3	68160
Normal	1500	107	1	79717
Normal	3000	162	1	82770

Langdon Bend functional connectivity was higher during normal conditions than for drought conditions for all dispersal distances. The functional connectivity also increased as the dispersal distance increased for both conditions (Table 5.4). The wetlands were not functionally connected into one component for drought and normal conditions at 500 m (Table 5.4, Figure 5.8). For both the drought and normal conditions at 3000 m all the wetlands were connected (Figure 5.10). The rest were connected into one component, but the number of links increased as the distance increased (Table 5.4, Figures 5.8-5.10).

Table 5.4. Functional connectivity of Langdon Bend during drought conditions and normal conditions, and for short-range dispersers (500 m), medium-range dispersers (1500 m), and long-range dispersers (3000 m).

Condition	Distance (m)	NL	NC	EC(IIC)
Drought	500	4	2	62248
Drought	1500	7	1	75274
Drought	3000	10	1	78531
Normal	500	8	3	78068
Normal	1500	31	1	105138
Normal	3000	45	1	109168

There were a few wetlands in each bend that continuously had the highest dIIC and therefore are important to functional connectivity. Other wetlands played important roles at different dispersal distances, and occasionally had higher dIICflux or dIICconnector values, but there were a few that were consistently the most important. The wetland in Hamburg Bend most important to connectivity for all dispersal distances and conditions was H18/59 (Figure 5.11, Appendix D). The importance of this wetland is due to both its position and size. It is a long, narrow wetland that provides the only habitat between the north and south end of the wetland complex. For Kansas Bend the wetland most important to connectivity during the drought condition was K24 (Figure 5.12, Appendix D). This wetland is long, thin, and is connected to the river. The wetland most important to connectivity during normal conditions is K07 (Figure 5.12, Appendix D). This is a large, but shallow ephemeral wetland. The wetland most important to connectivity of Langdon Bend for all dispersal distances and conditions was L5370 (Figure 5.13, Appendix D). This wetland is very large, long and is occasionally connected to the river itself. More detailed results of wetland importance can be found in Appendix D.

Figure 5.1. The functional connectivity of the wetland complexes Hamburg Bend, Kansas Bend, and Langdon Bend at the 3000m dispersal range during normal conditions. The wetlands range between Nebraska City, NE and Nemaha, NE along the Missouri River. The orange dots represent the wetlands of Hamburg Bend, the yellow dots represent the wetlands of Kansas Bend, and the red dots represent Langdon Bend. The connections are in magenta.

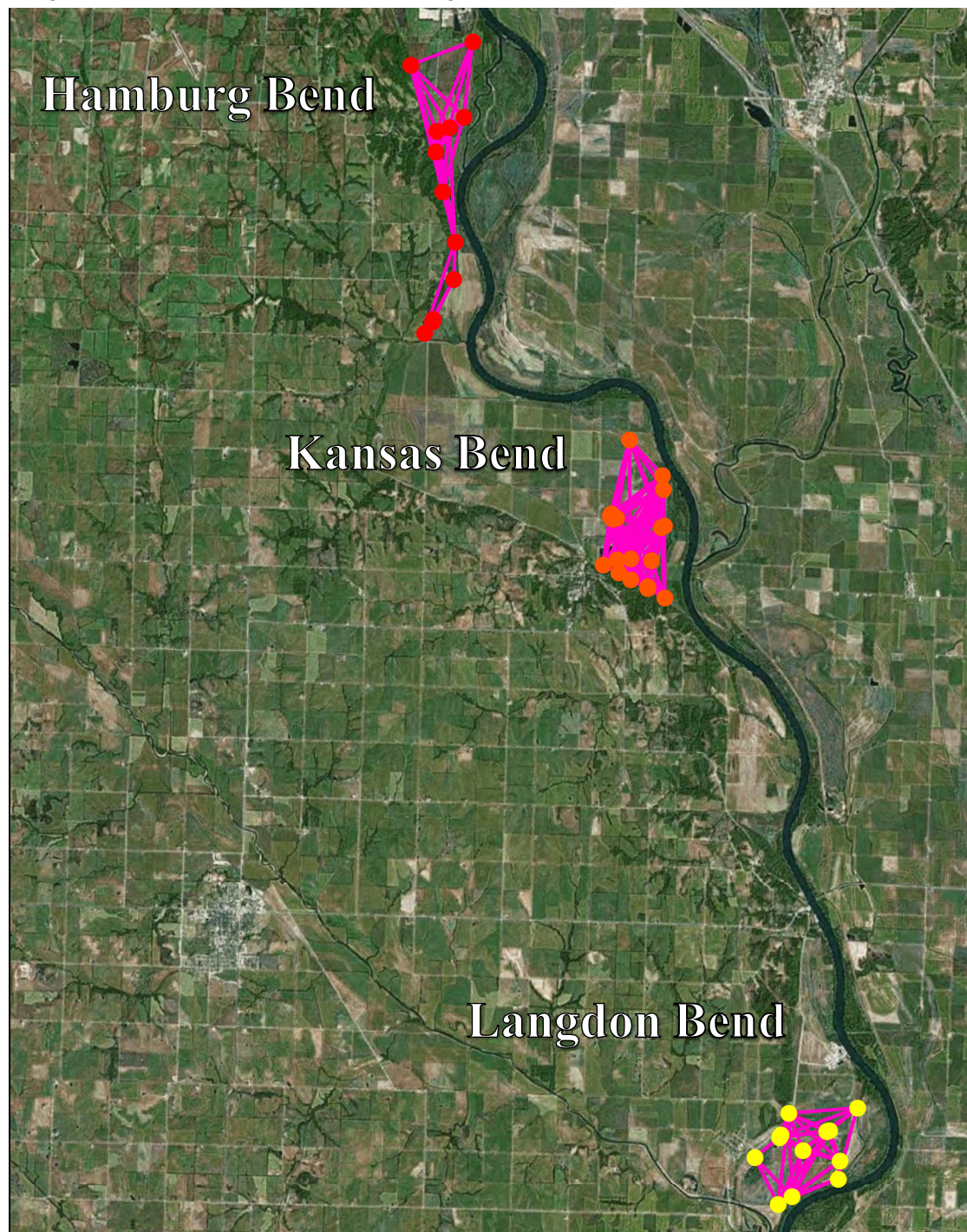


Figure 5.2. Map of the functional connectivity of Hamburg Bend during drought and normal conditions for short-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by red dots and the links are represented by green lines.

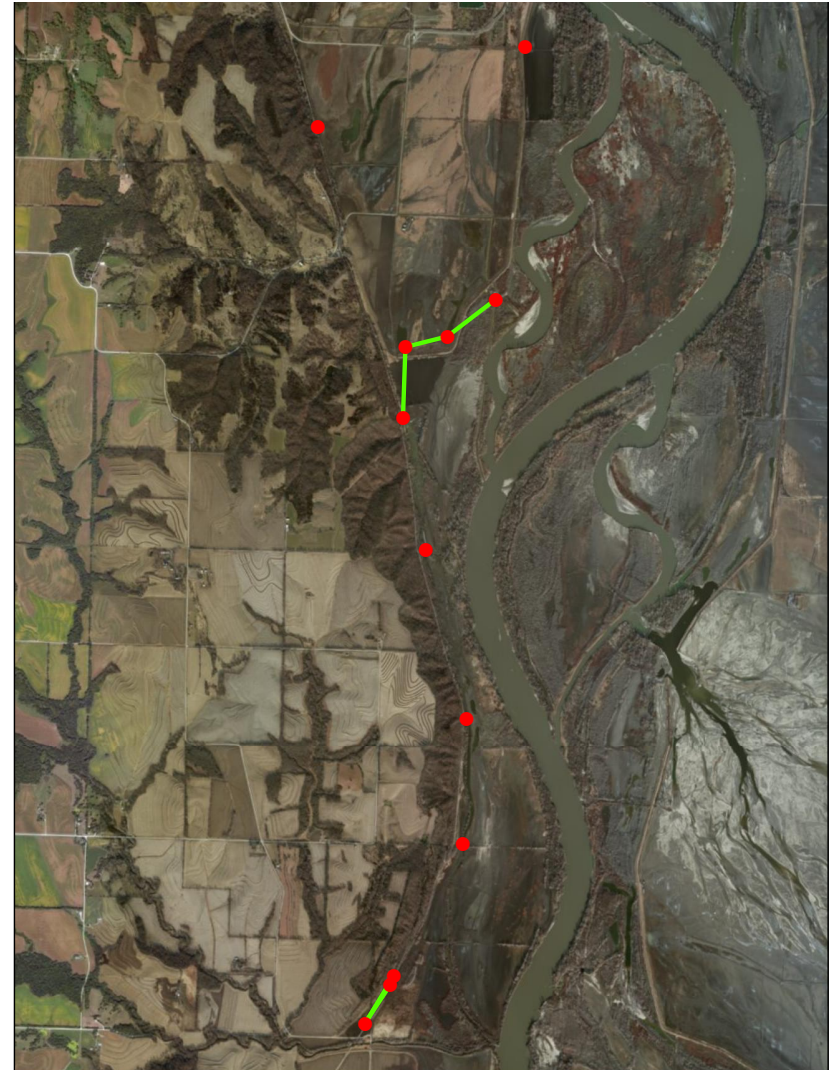


Figure 5.3. Map of the functional connectivity of Hamburg Bend during drought and normal conditions for short-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by red dots and the links are represented by light blue lines.

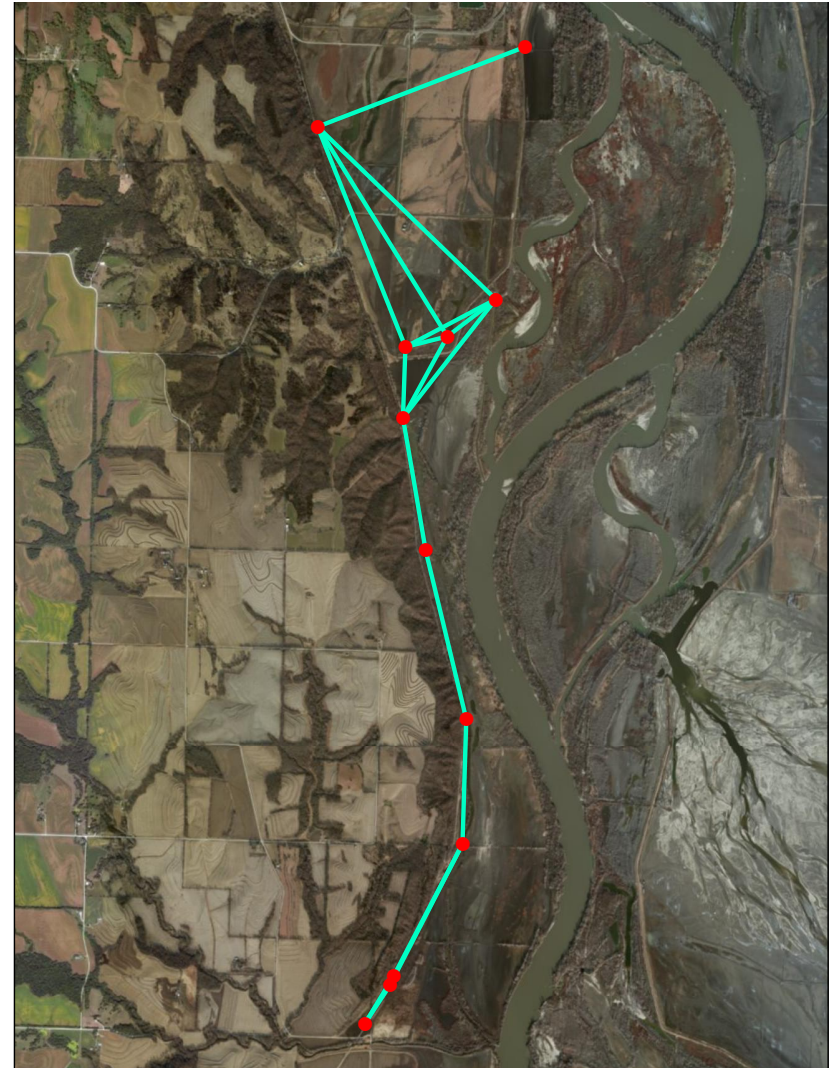
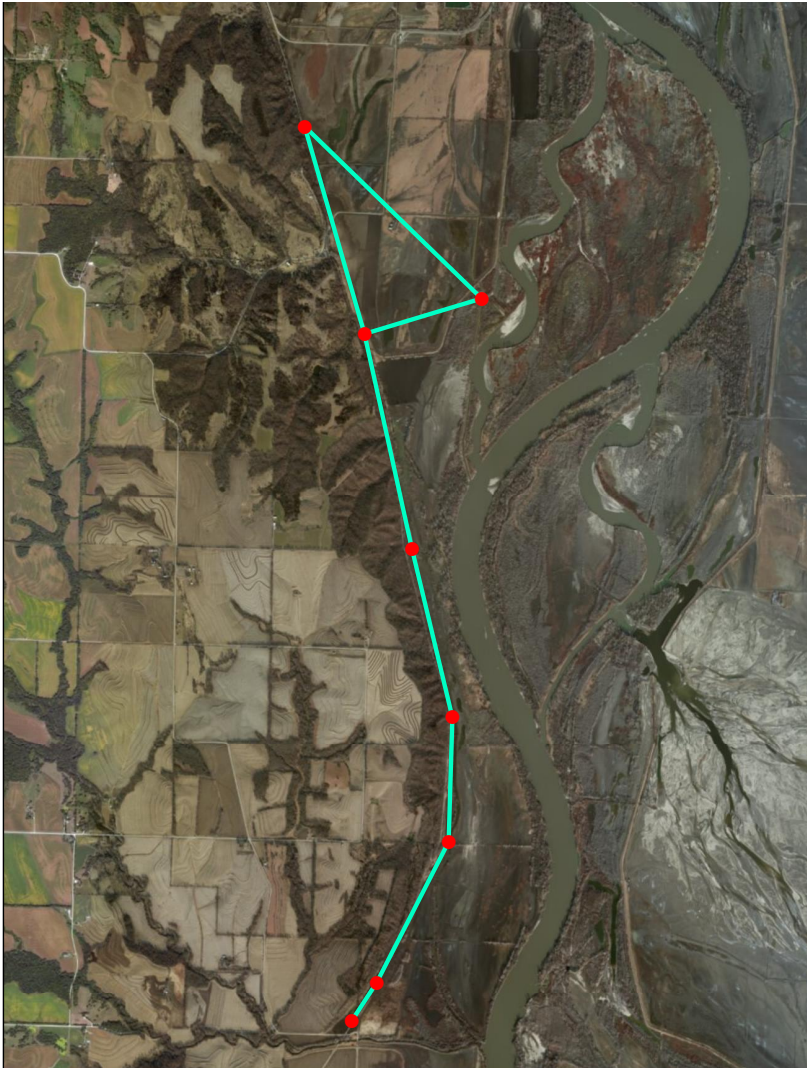


Figure 5.4. Map of the functional connectivity of Hamburg Bend during drought and normal conditions for long-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by red dots and the links are represented by magenta lines.

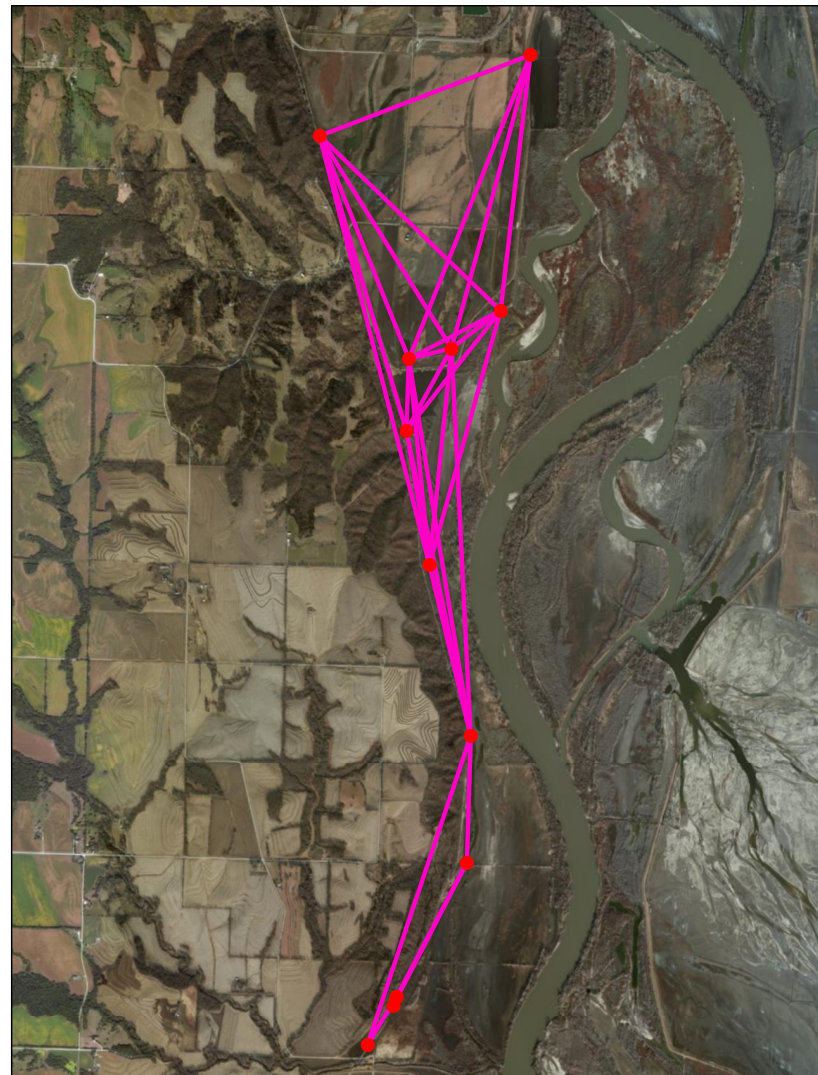
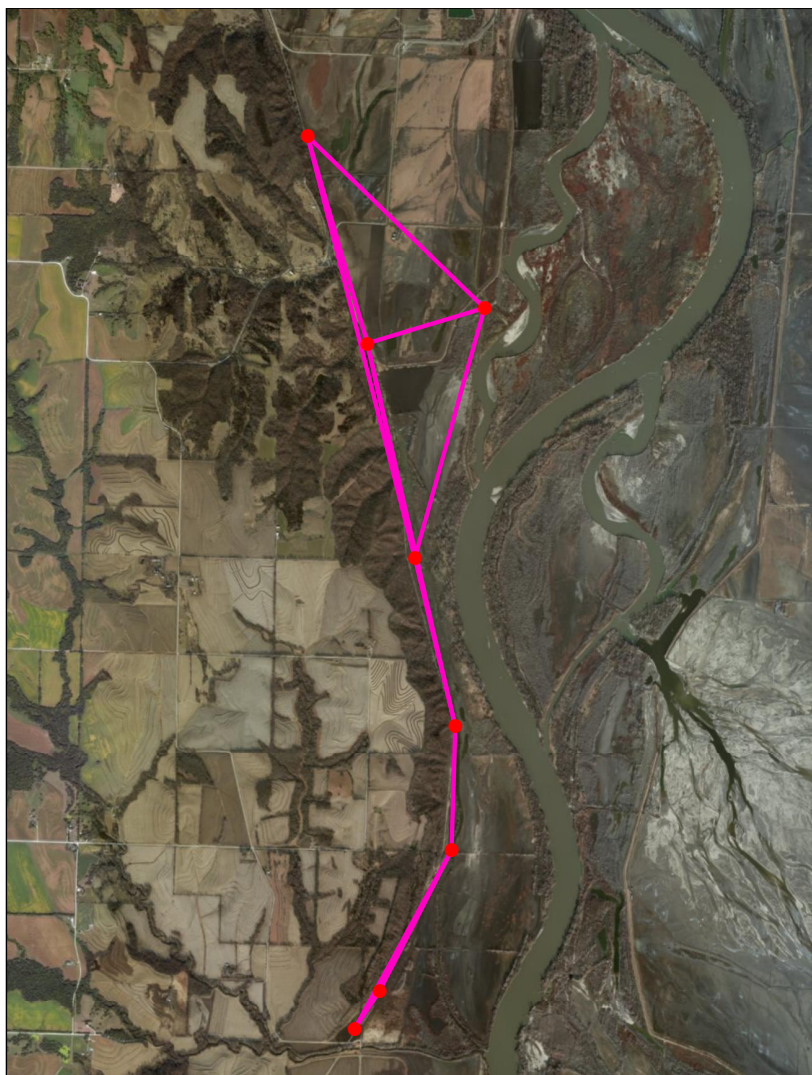


Figure 5.5. Map of the functional connectivity of Kansas Bend during drought and normal conditions for short-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by orange dots and the links are represented by green lines.

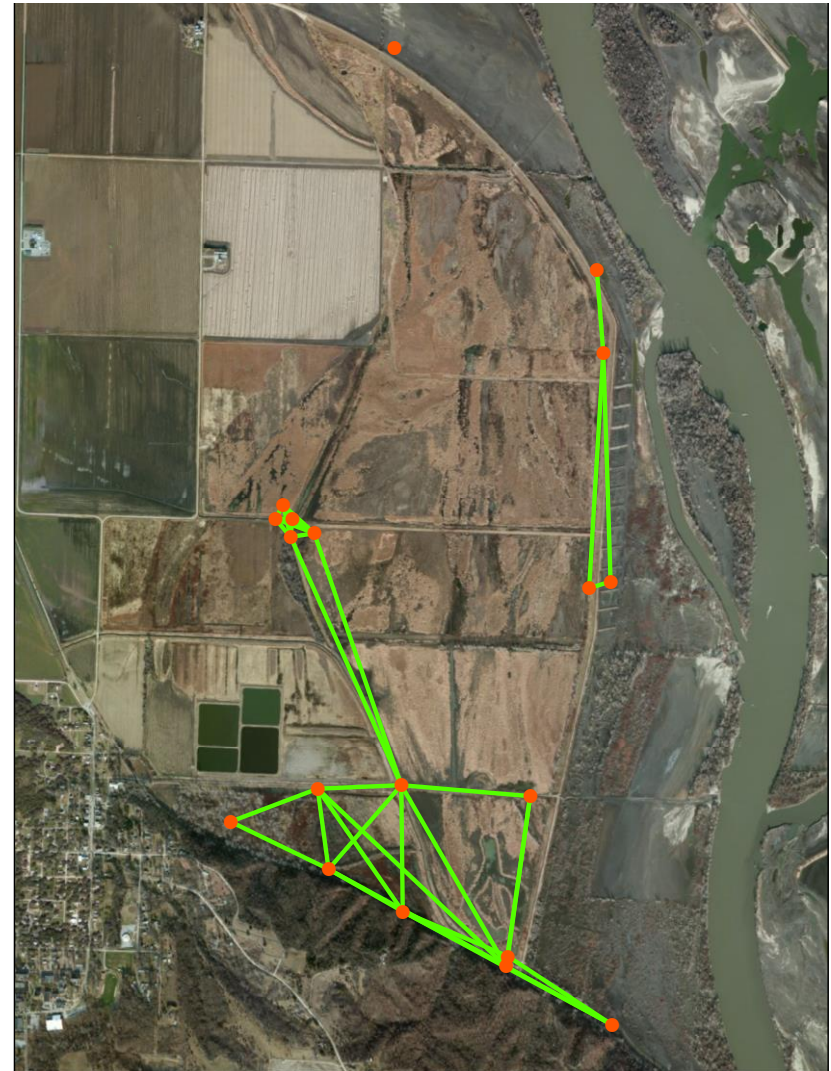


Figure 5.6. Map of the functional connectivity of Kansas Bend during drought and normal conditions for short-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by orange dots and the links are represented by light blue lines.

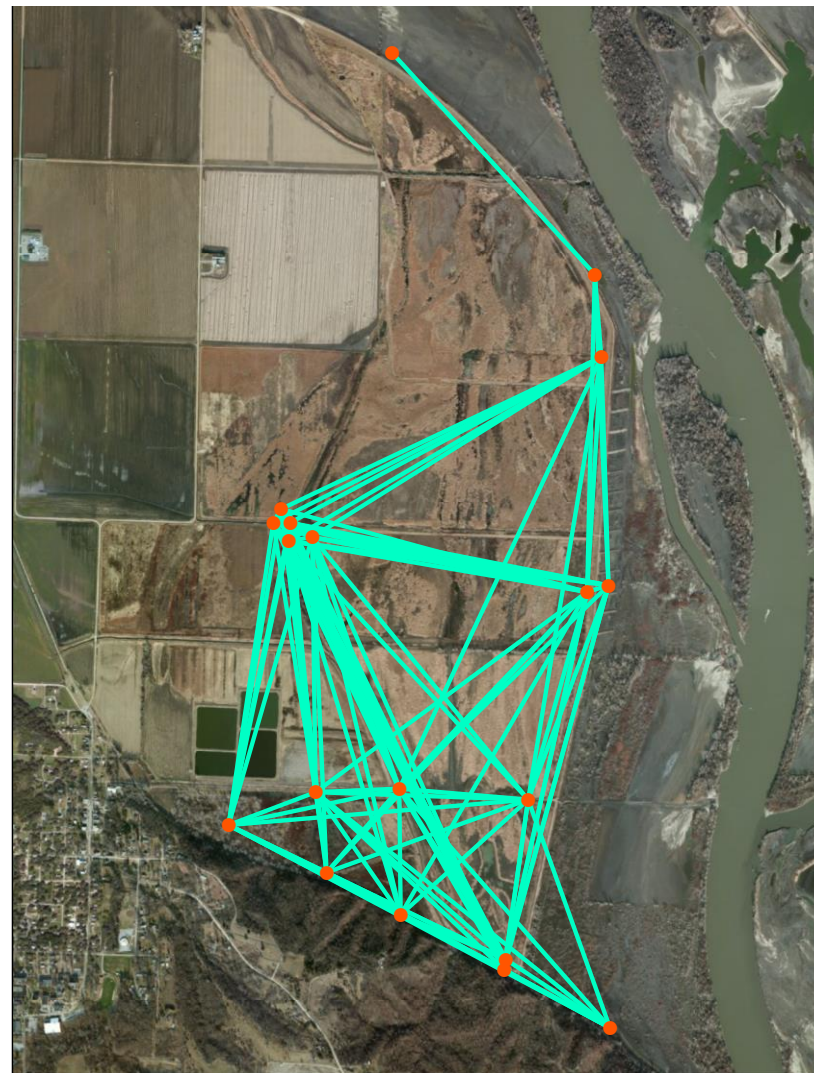


Figure 5.7. Map of the functional connectivity of Kansas Bend during drought and normal conditions for long-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by orange dots and the links are represented by magenta line.

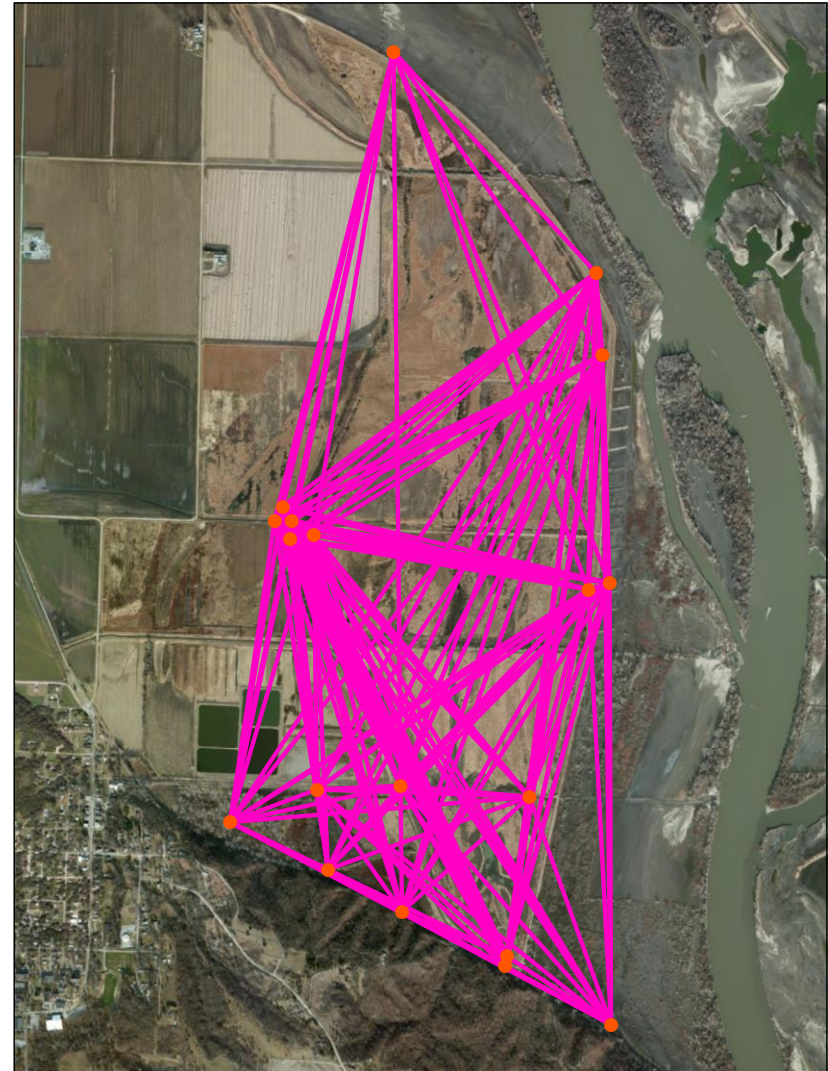
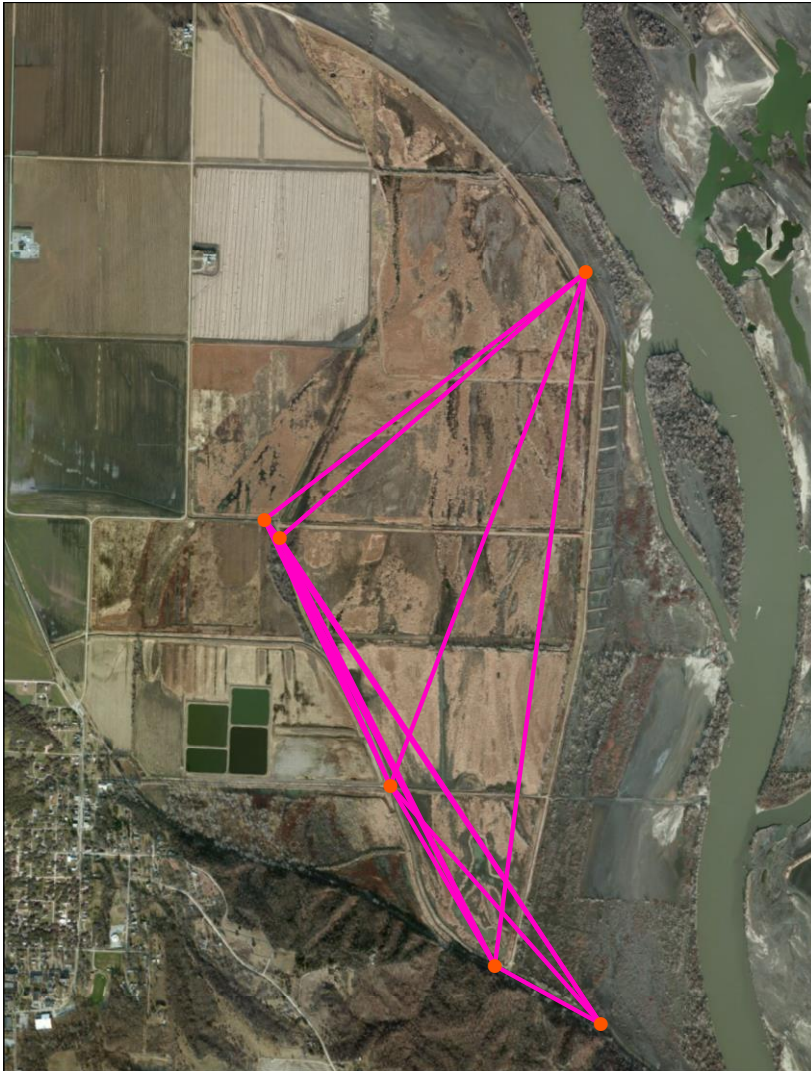


Figure 5.8. Map of the functional connectivity of Langdon Bend during drought and normal conditions for short-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by yellow dots and the links are represented by green lines.

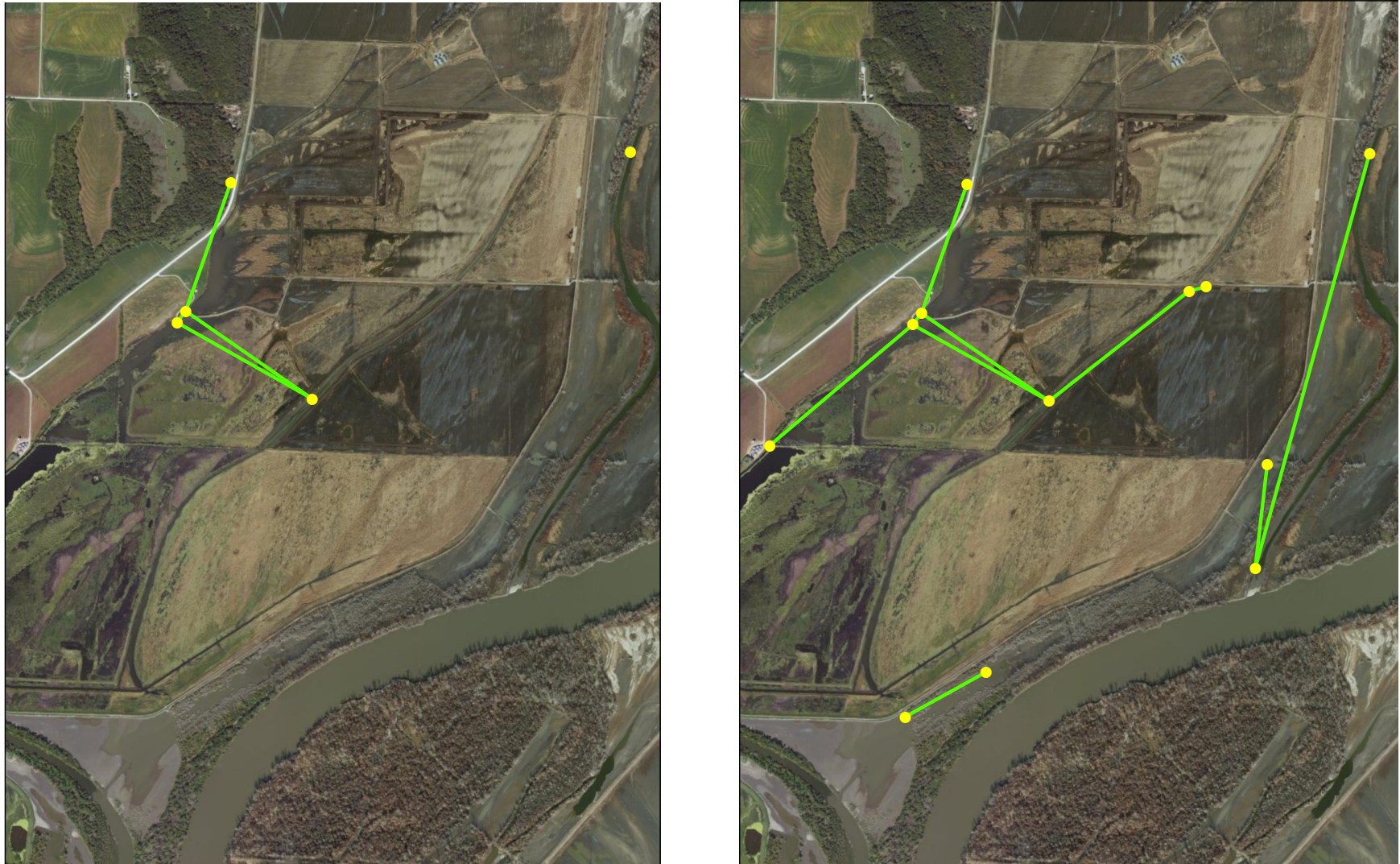


Figure 5.9. Map of the functional connectivity of Langdon Bend during drought and normal conditions for medium-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by yellow dots and the links are represented by light blue lines.

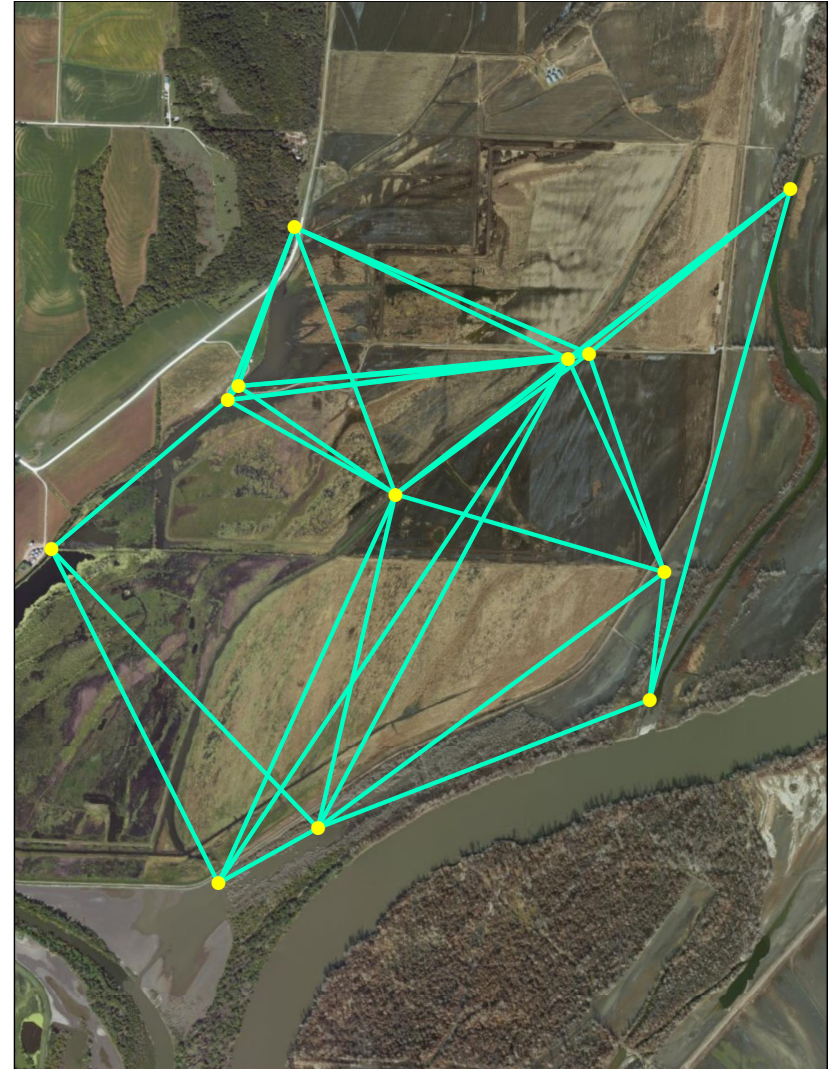
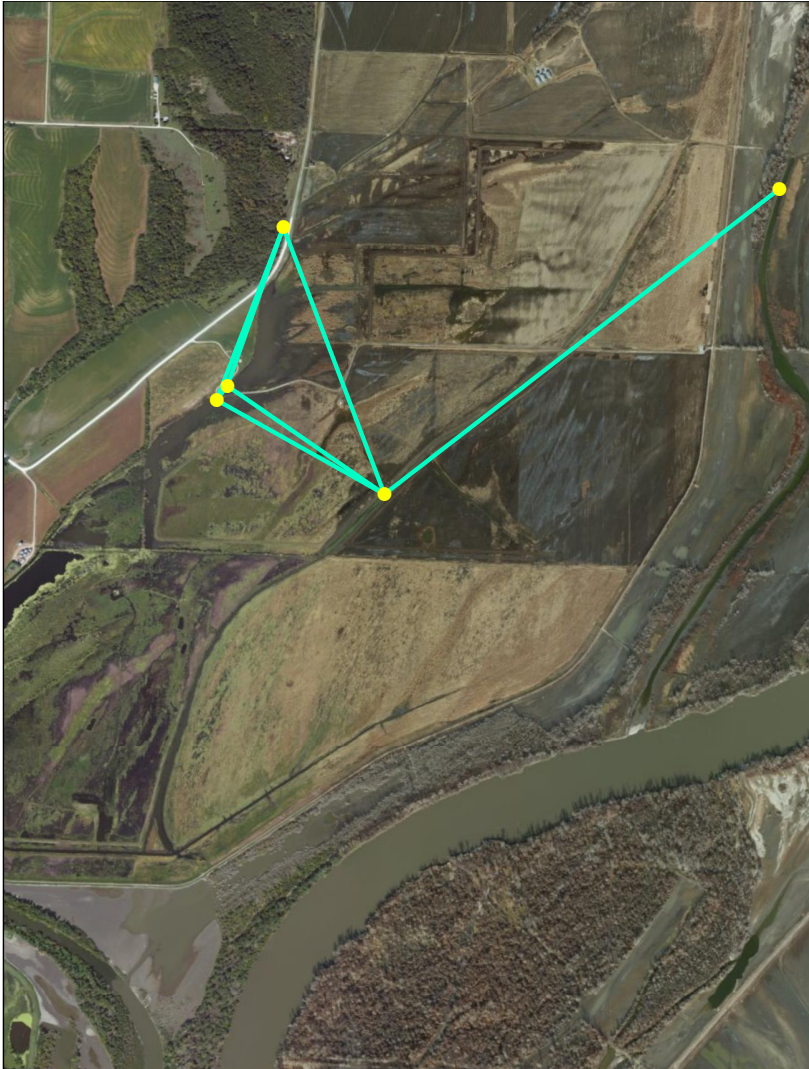


Figure 5.10. Map of the functional connectivity of Langdon Bend during drought and normal conditions for long-range dispersers. The functional connectivity for drought conditions is on the left and the functional connectivity of normal conditions is on the right. Wetlands are represented by yellow dots and the links are represented by magenta lines.

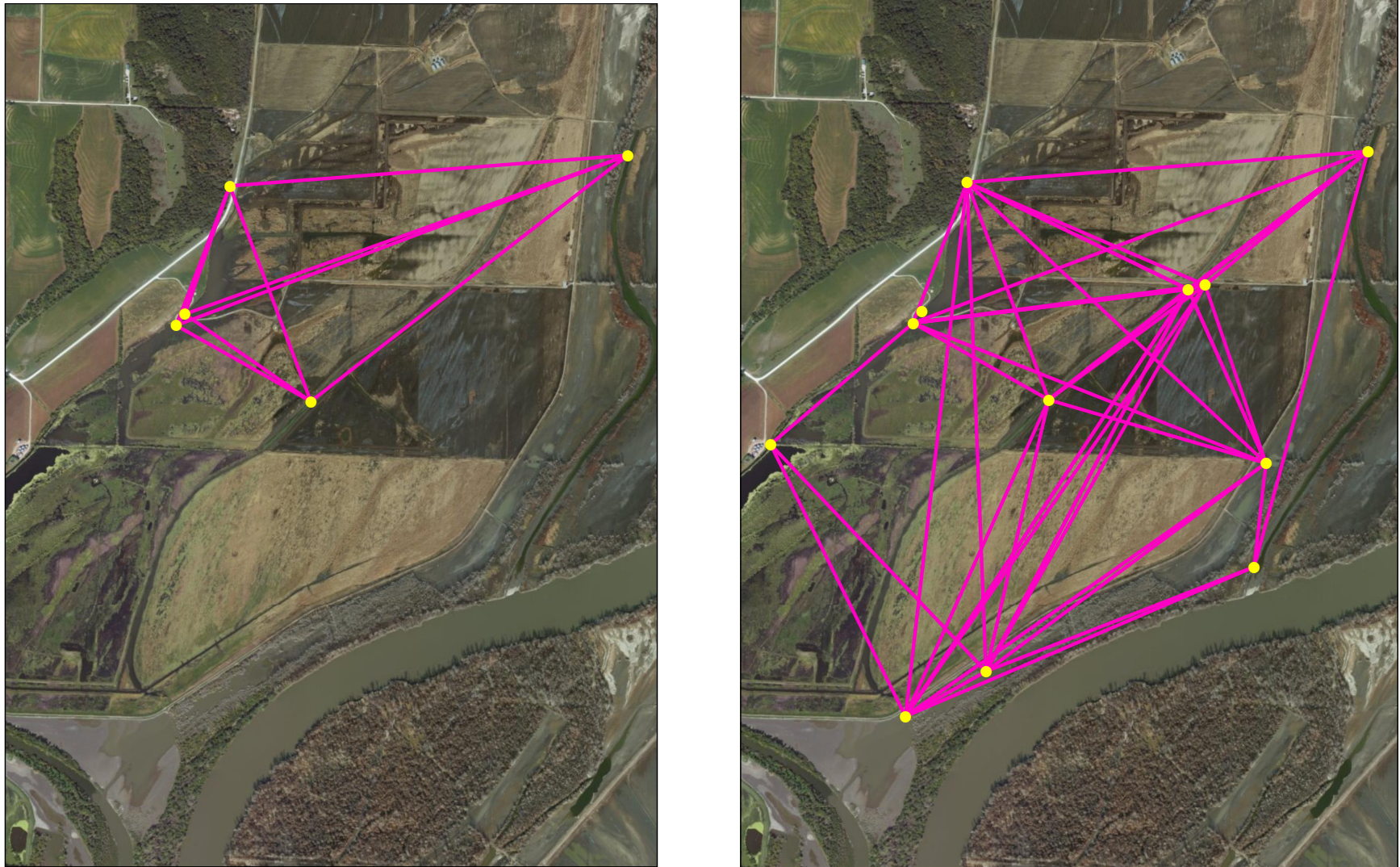


Figure 5.11. Functional connectivity of Hamburg Bend during normal conditions for long-range dispersers. The wetland, H59/61, that consistently had the highest dIIC is circled in white. While two points are shown, it was analyzed as one wetland. Two points were used at each end to more accurately visualize functional connectivity for the long, narrow wetland. Both its size and position contribute to its high dIIC value.

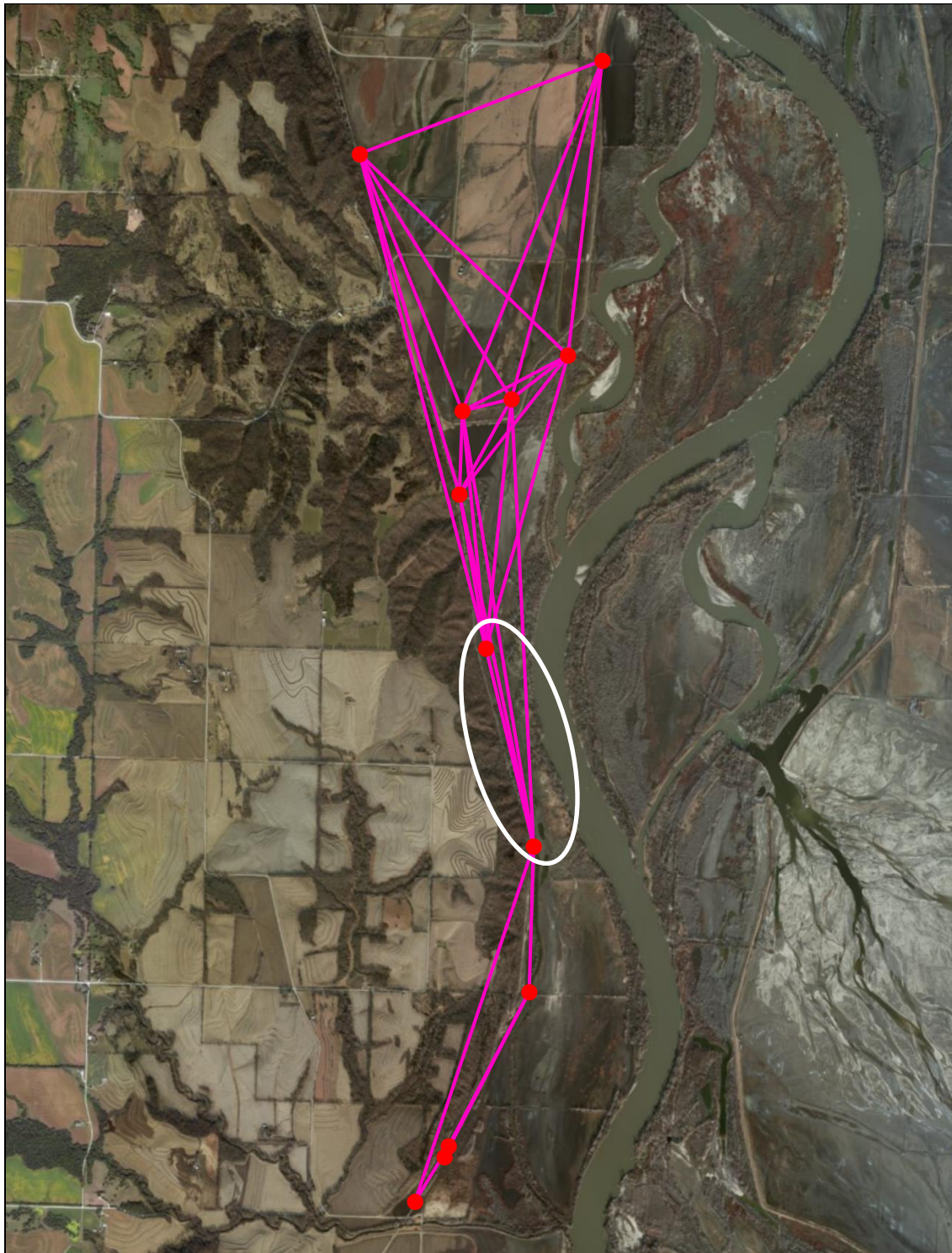


Figure 5.12. Functional connectivity of Kansas Bend during drought (left) and normal (right) conditions for long-range dispersers. The wetlands that had the highest dIIC, K24 and K0, are circled in white. In drought conditions K24 had the highest dIIC. In normal conditions K07 had the highest dIIC.

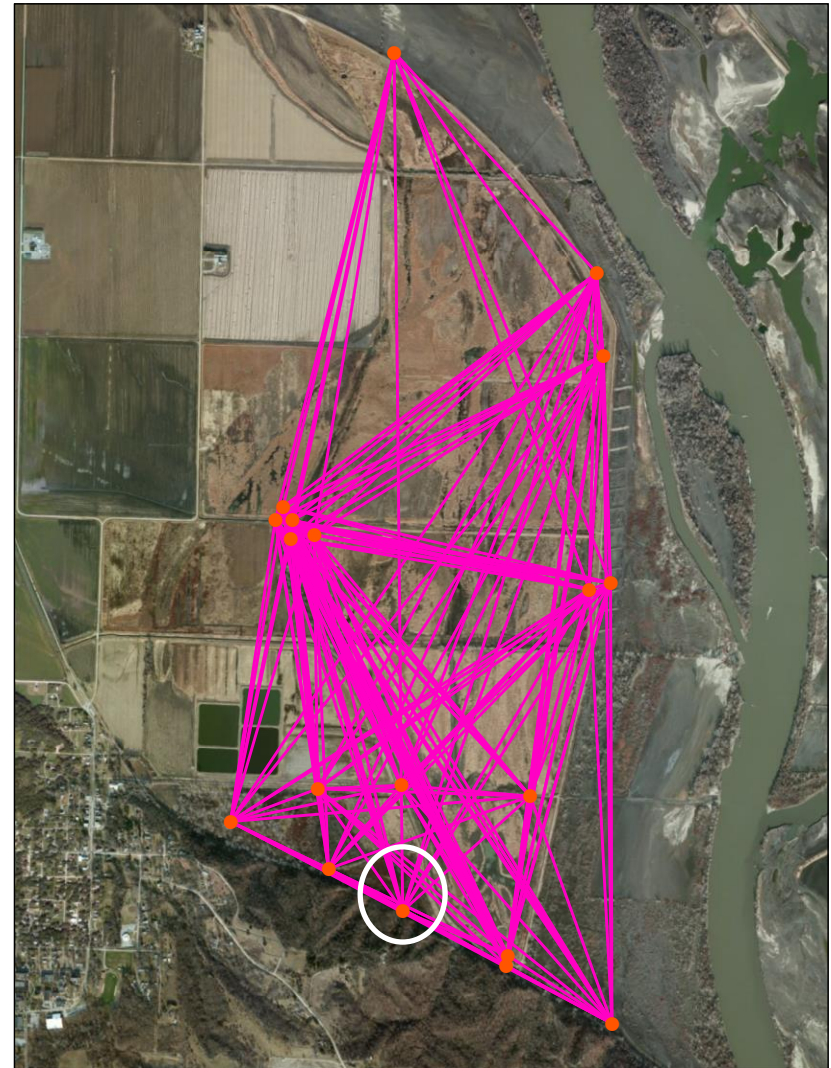
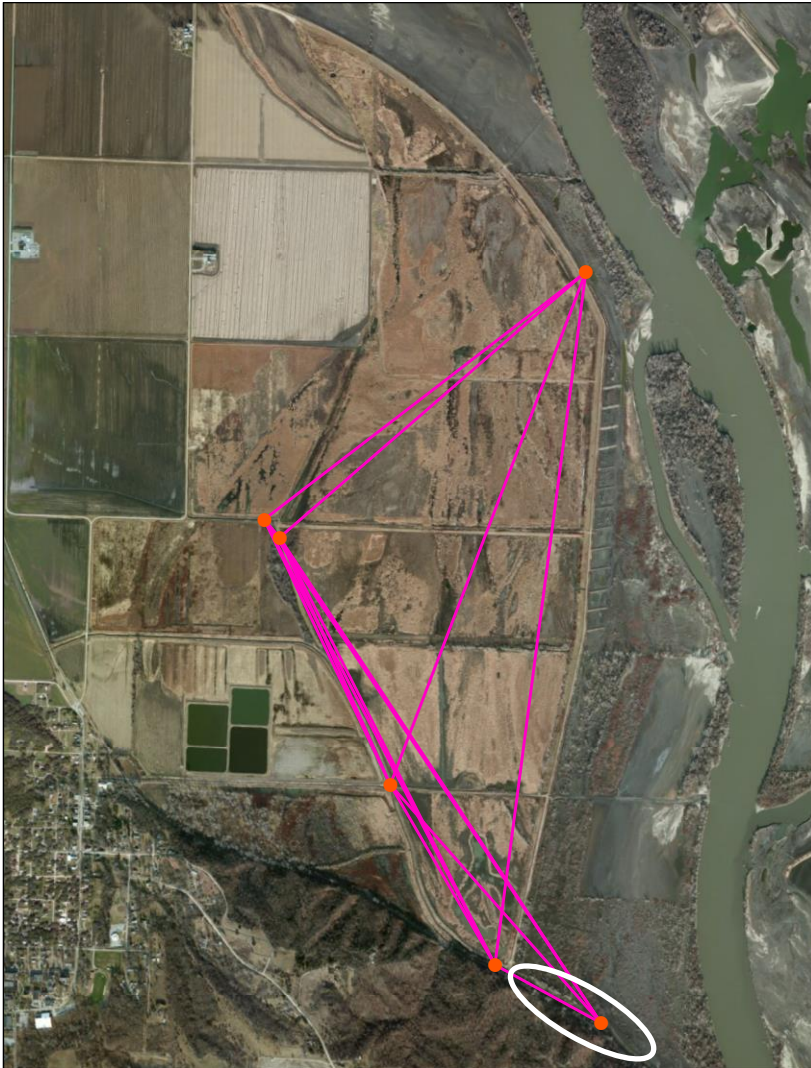
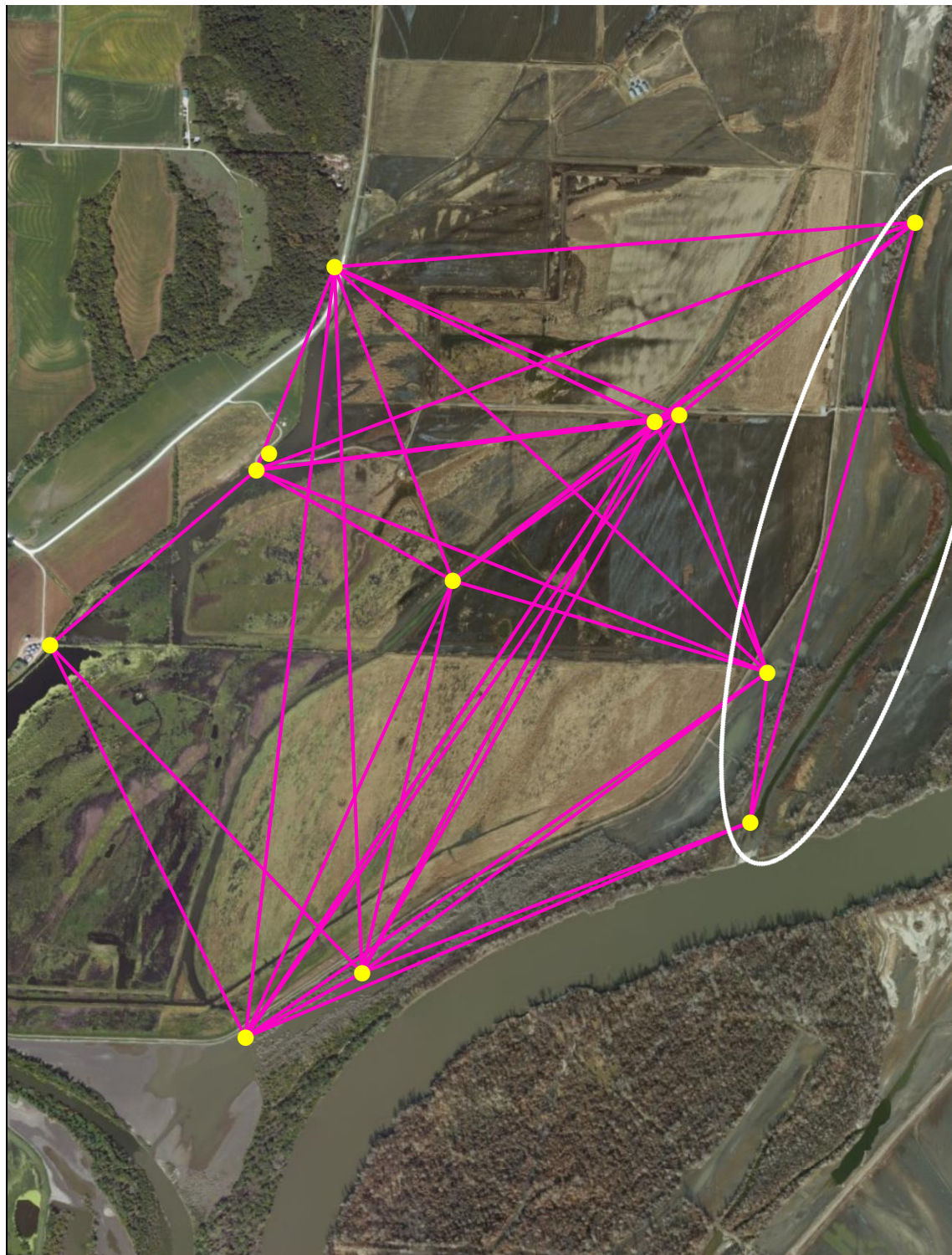


Figure 5.13. Functional connectivity of Kansas Bend during normal conditions for long-range dispersers. The wetland, L53/70, that consistently had the highest dIIC is circled in white. While two points are shown, it was analyzed as one wetland. Two points were used at each end to more accurately visualize functional connectivity for the long, narrow wetland.



DISCUSSION

To reflect the complexity of wetland restorations in a highly variable environment inhabited with organisms with very different dispersal abilities, it is necessary to assess functional connectivity of river bends with on-going restoration efforts. Drought decreased the functional connectivity of all the restored wetland complexes, as expected. However, even with normal water conditions the complexes did not provide complete functional connectivity within bend complexes, for the anurans with the shortest dispersal distance. This is especially true for Hamburg Bend. This is probably because the wetlands have been created linearly. Kansas Bend was a little more clustered and less linear, but had some outliers. Langdon Bend had the most compact wetland complex and had the highest functional connectivity for all conditions and dispersal distances.

In drought conditions Hamburg Bend had about twice the functional connectivity of Kansas Bend. This is because Hamburg Bend has many long, deep, and thin tributaries and ditches that provide habitat in drought conditions, while Kansas Bend had fewer with smaller area. These may not be ideal wetlands, but they provide habitat that provides refugia during drought. It is my personal observation that some of these less successful wetlands function differently in drought conditions. When a lot of the water evaporates their slope becomes shallower, they are more likely to have aquatic vegetation, and most of the fish die. Even when they do function differently, anurans are more likely to just use them as refuge and not use them to breed. In 2012, the worst drought year, there were fewer species heard calling, and those species that were calling were doing so at fewer wetlands. For the tadpoles in 2012, the situation was even more dismal. Only two species, one of which was the invasive American Bullfrog, had a naïve occupancy (number of wetlands where a species is present at least once/total number of wetlands) higher than 0.10, while in 2013 six species did, not including the American Bullfrog. This suggests that anurans decrease their reproduction in drought years, using the less ideal habitat to survive until conditions are more favorable. In the normal conditions Kansas Bend has about one and a half to two times the functional connectivity of Hamburg Bend. However, in normal conditions, Hamburg Bend had four more

wetlands and Kansas Bend gained 13. The creation of a large amount of smaller, shallower, ephemeral wetlands in normal years greatly increased the functional connectivity. Wetlands that have aquatic vegetation, shallow slopes, are ephemeral, and have at least one other wetland nearby are more likely to have successfully breeding anurans. Clearly, the refugia wetlands are less than ideal. However, ephemeral wetlands that produce the most successful reproduction are the first to dry up in drought conditions. Therefore, the less ideal tributaries, ditches, and ground fed, permanent wetlands are necessary to provide refuge for the anurans in drought conditions.

This was reiterated by the type of wetlands that were most important to functional connectivity in drought conditions. All of the wetlands that were most important to functional connectivity in drought conditions were large, thin, and had steep slopes. In fact, for Hamburg and Langdon Bend the same wetland that was most important to functional connectivity during drought conditions was most important to functional connectivity in normal conditions as well.

Wetlands need to be close enough to one another that short-range dispersers can move from a successful wetland into a deeper, refuge when necessary. This level of connectivity is present in Langdon Bend. The compact nature and variety of wetlands provided the best functional connectivity in all situations. While the highest functional connectivity is not always considered the best, due to the potential spread of disease and invasive species, Langdon Bend did have the highest number of species found for both tadpoles and adults. This included species that are less common such as the Great Plains Toad and the Plains Spadefoot.

It is possible that this analysis did not consider all potential habitats. There were small wetlands within the complexes that were not surveyed and may have provided additional habitat and increased functional connectivity, though there was a positive relationship between wetland size and persistence. The 2011 flood altered and created many wetlands. However, due to the extreme nature of the drought the only wetlands holding water were deep, permanent wetlands. It is also possible that the river itself

provides connectivity as refuge in drought conditions. They may use the shallower, slower water at the edges to travel and survive, but it is unlikely that they use it to lay eggs. Conversely, since channelization, there is little shallow water habitat. In addition to deepening the river, the channelization also made the river wider and straighter. All of this made the current more rapid and decreased the likelihood that the frogs could or would use the river in any way, especially not to cross to habitat on the other side.

Therefore, using the information gained from the functional connectivity analysis a successful flood plains wetland is part of a compact (non-linear) wetland complex that has other successful wetlands and deeper, larger, less ideal wetlands that are all close enough to one another that habitat is provided in extreme conditions. When this is combined with the definitions from previous chapters (Chapter 2, Chapter 3, and Chapter 4) a successful wetland has aquatic vegetation, a shallow slope (preferably less than 0.30), is ephemeral, has at least one wetland within 500 m, and is part of a compact (non-linear) wetland complex that has other successful wetlands and deeper, larger, less ideal wetlands that are all close enough to one another that habitat is provided in extreme conditions. This analysis also provided data for specific recommendations for both management and general placement of new restored wetlands to improve functional connectivity. The importance of each node (wetland) was also analyzed (Appendix D). This information shows the different roles each wetland plays in the different situations and could be useful in guiding conservation and management. River flood plains are variable and complex systems. Traditionally, flooding occurred annually altering and renewing the landscape. More recently, global climate change has increased extreme conditions including both flooding and drought. Thus, creating functionally connected, and therefore more resilient, wetland complexes is essential to wetland restoration success on a flood plain.

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CHAPTER 6: CONCLUSION

Wetlands are important and necessary for ecosystem function. They act as filters and sinks for the hydrological cycle, in river ecosystems they provide nutrient rich soil and flood protection, and all provide essential habitat for many plants, fish, and wildlife species. Most of this critical habitat has been altered or destroyed worldwide. In river ecosystems damming, channelization, and flood control structures have cut flood plains off from the annual flooding that maintained the variable, but successful wetland complexes.

It is unlikely that these structures will be removed, so it is up to programs like the Missouri River Fish and Wildlife Mitigation Project, implemented by the USACE, to restore wetlands and monitor them to determine if they are successful. Amphibian monitoring and occupancy analysis were used to determine success. Amphibians rely on these complex ecosystems, and the decline in wetland habitat is directly linked to the global decline in amphibian populations. Amphibians' permeable skin and biphasic life cycle make them sensitive to environmental perturbation in both the aquatic and terrestrial environment. This makes them good indicators of success for restored wetlands, which can be altered by disturbances in either. The results of the amphibian occupancy analysis were used to define success and create wetland restoration guidelines for the future.

In chapter 2 I conducted an occupancy analysis of the frog call survey monitoring data to determine the factors affecting adult amphibian detection and occupancy. The covariates in the top occupancy models were used to define what a successful wetland is. Overall, a successful wetland was defined as an ephemeral wetland with a shallow slope, aquatic vegetation, and is part of a functionally connected complex with a variety of different wetland types.

In chapter 3 I determined the factors affecting larval amphibian detection and occupancy, and then compared this with the adult amphibian results reported in the previous chapter. The results of the frog call surveys and tadpole dip net surveys differed in many ways. The data from the tadpole dip net

surveys produced fewer results, but these results were more accurate because of their link to reproduction and the lack of spurious data. If only one method is used it should be the tadpole dip net surveys, but if the surveys are conducted in a drought year it is possible there will be insufficient data to conduct an occupancy analysis. Therefore, conducting both methods is the best way to produce the most accurate wetland restoration guidelines. Using the occupancy results of both methods, it was determined that a successful wetland contains aquatic vegetation, has a shallow slope (less than 0.20), is ephemeral, and has at least one other wetland nearby (within 500 m).

In chapter 4 I used a novel co-occurrence analysis to inform wetland restoration. As in Chapter 2, this Chapter focuses on the factors affecting adult amphibian detection and occupancy, but it used the occupancy of different species combinations as opposed to the occupancy of single species. This method was both more efficient and also produced support for the covariate, distance to the nearest wetland, which was not decidedly supported in the single species frog call survey analysis, and the main reason for suggesting both the tadpole surveys and frog call surveys should be used. It was concluded that, a successful wetland contains aquatic vegetation, has a shallow slope (less than 0.30), is ephemeral, and has at least one other wetland nearby (within 500 m). This is similar to the previous guideline conclusions, but the results were more unanimous, supported, and quicker to produce.

In chapter 5 I used functional connectivity based on anuran dispersal distances to further inform wetland restoration. This chapter not only added a complex systems component to the wetland restoration guidelines, but also provided a method that can help focus management by identifying wetlands within and across complexes that are most important to functional connectivity and that can determine functional connectivity of future wetlands. By comparing the differences in the functional connectivity of three wetland complexes during drought conditions and normal conditions, it was determined that a successful flood plains wetland is part of a compact (non-linear) wetland complex that has other successful wetlands and deeper, larger, less ideal wetlands that are all close enough to one another that habitat is provided in extreme conditions.

From the information gained in this thesis I have a few recommendations. I would suggest that complexes are modeled from Langdon Bend, as compact, non-linear complexes with several large, deep, permanent wetlands, but with even more successful wetlands like Kansas Bend. Langdon Bend is probably the smallest a flood plains wetland complex should be. In drought conditions it only had five wetlands holding water, all of them large, deep, and thin. Subsequently, wetland complexes should contain at least five deep, permanent wetlands, but I would recommend more. Hamburg Bend should have more wetlands in the southern half of the complex. Unfortunately, this portion of the complex is stuck between the bluffs and the river. The functional connectivity of Hamburg Bend could still be increased by creating more successful, ephemeral wetlands closer to the river. Kansas Bend needs more deep, permanent wetlands on the northern end of the complex. Some of the surrounding land for all three bends belongs to private landowners and this will potentially impede the ability to add wetlands to each complex. However, if a few new wetlands could be added to each complex in the suggested manner and monitoring was continued, it could add detail to the wetland restoration guidelines and confirm the validity of the above recommendations.

While the wetland restorations guidelines determined are general, they do give a starting point for creating successful wetlands on a flood plain. River flood plains are variable and complex systems. Traditionally, flooding occurred annually altering and renewing the landscape. In the future global climate change is likely to increased extreme conditions, including both flooding and drought. Thus, more monitoring is needed to increase our understanding of the restored wetlands in all situations. However, monitoring data is useless without analysis and the application of the knowledge gained from analysis. Thus, continued longitudinal monitoring in an adaptive management framework is needed to build upon the wetland restoration guidelines. It is our hope that this research will continue and lead to more specific wetland restoration guidelines and the creation of more successfully restored wetlands.

APPENDIX A: DETECTION AND OCCUPANCY CONFIDENCE SETS FOR ADULT AMPHIBIANS

Table 1. Blanchard's Cricket Frog call detection confidence set for April 2012. The models for water temperature, time, wind, and size were taken out because they would not converge. Also, agriculture and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Air Temperature	11	62.39	0	0.38	0.38	0.25	0.16
Null	10	62.4	0.01	0.38	0.77	-1.58E-05	0.45
Moonshine	11	63.73	1.35	0.2	0.96	-5.69	3.26

Table 2. Blanchard's Cricket Frog call detection confidence set for May 2012. The models for air temperature, size, and environmental covariates 1 were taken out because they would not converge. Also, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	89.82	0	0.36	0.36	0.00	0.29
Wind	9	90.25	0.43	0.29	0.66	-1.78	0.67
Moonshine	9	90.76	0.94	0.23	0.88	1.82	1.05
Time	9	93.96	4.14	0.05	0.93	-0.21	0.50
Water Temperature	9	94.5	4.68	0.04	0.96	-0.35	0.17

Table 3. Blanchard's Cricket Frog call detection confidence set for June 2012. The models for water temperature and size were taken out because they would not converge. Also, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	69.05	0	0.79	0.79	0.65	0.46
Water Temperature	8	73.74	4.69	0.08	0.86	0.23	0.29
Moonshine	8	73.95	4.89	0.07	0.93	1.20	1.71
Time	8	75.13	6.07	0.04	0.97	-0.43	0.79

Table 4. Woodhouse's Toad call detection confidence set for April 2012. The models for time, size, and air temperature were taken out because they would not converge. Also, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Water Temperature	9	29.27	0	0.79	0.79	9.00	22.50
Null	8	32.84	3.57	0.13	0.92	-0.003	0.82
Wind	9	34.5	5.22	0.06	0.98	-2.87	2.50

Table 5. Gray Treefrog Complex call detection confidence set for April 2012. The models for size, environmental covariates 1, and environmental covariates 2 were taken out because they would not converge. Also, size, adjacent, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Water Temperature	9	33.52	0	0.94	0.94	3.24	3.53
Null	8	39.75	6.23	0.04	0.98	-0.69	0.61

Table 6. Gray Treefrog Complex call detection confidence set for June 2012. The models for time, environmental covariates 1, and environmental covariates 2 were taken out because they would not converge. Also, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	10	51	0	0.89	0.89	1.10	0.82
Moonshine	11	55.88	4.88	0.08	0.96	8.08	6.53

Table 7. Plains Leopard Frog call detection confidence set for April 2012. The models for moonshine, air temperature, and wind were taken out because they would not converge. Also, aquatic vegetation, adjacent vegetation, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Water Temperature	9	70.63	0	1	1	0.57	0.16

Table 8. Plains Leopard Frog call detection confidence set for May 2012. Aquatic vegetation, adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Time	8	71.85	0	0.96	0.96	-2.63	0.85

Table 9. Plains Leopard Frog call detection confidence set for June 2012. The model for environmental covariates 2 was taken out because it would not converge. Also, ephemeral, adjacent vegetation, distance to nearest wetland, bend, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Moonshine	5	37.11	0	0.91	0.91	25.50	21.10
Null	4	43.45	6.34	0.04	0.95	-0.06	0.54

Table 10. American Bullfrog call detection confidence set for April 2012. The model for water temperature was taken out because it would not converge. Also, adjacent vegetation, distance to nearest wetland, bend, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	5	35.79	0	0.31	0.31	-1.61	0.63
Air Temperature	6	36.04	0.25	0.27	0.58	0.32	0.18
Moonshine	6	37.57	1.78	0.13	0.71	3.69	3.32
Wind	6	37.76	1.97	0.12	0.83	1.97	1.52
Water Temperature + Air Temperature + Wind	8	37.88	2.09	0.11	0.94	-11.46	166
Water Temperature + Air Temperature + Wind						7.23	149
Water Temperature + Air Temperature + Wind						44.38	721
Time	6	40.47	4.68	0.03	0.96	0.71	0.57

Table 11. American Bullfrog Call Detection Confidence Set for May 2012. The model for size was taken out because it would not converge. Also, adjacent vegetation, ephemeral, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	9	62.08	0	0.55	0.55	-0.68	0.42
Time	10	63.71	1.63	0.24	0.80	-1.45	0.83
Water Temperature	10	66.16	4.08	0.07	0.87	0.29	0.20
Wind	10	66.26	4.19	0.07	0.94	-0.84	1.42
Air Temperature	10	67.45	5.37	0.04	0.97	-0.62	1.32

Table 12. American Bullfrog call detection confidence set for June 2012. The models for size and time were taken out because they would not converge. Also, size, adjacent vegetation, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Air Temperature	7	44.13	0	0.42	0.42	-0.91	0.53
Null	6	45.11	0.98	0.26	0.68	-1.13	0.86
Moonshine	7	45.15	1.02	0.25	0.93	3.74	4.21
Wind	7	49.68	5.55	0.03	0.96	0.61	0.40

Table 13. Boreal Chorus Frog call detection confidence set for April 2012. The models for environmental covariates 1 and environmental covariates 2 were taken out because they would not converge. Also, size, adjacent vegetation, slope, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Water Temperature	7	50.09	0	0.96	0.96	0.57	0.16

Table 14. Blanchard's Cricket Frog call detection confidence set for May 2013. The models for time and wind were taken out because they would not converge. Also, size, adjacent vegetation, time, distance to nearest wetland, aquatic vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	4	58.66	0	0.54	0.54	0.61	0.40
Moonshine	5	60.78	2.12	0.19	0.73	-1.68	1.39
Air Temperature	5	61.44	2.78	0.13	0.86	0.12	0.13
Water Temperature	5	61.76	3.09	0.12	0.98	0.13	0.17

Table 15. Blanchard's Cricket Frog call detection confidence set for June 2013. Adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	121.31	0	0.47	0.47	0.44	0.25
Moonshine	9	123.08	1.77	0.2	0.67	2.30	2.02
Air Temperature	9	124.41	3.1	0.1	0.77	0.03	0.08
Wind	9	124.48	3.17	0.1	0.87	-0.10	0.44
Time	9	124.63	3.32	0.09	0.96	0.003	0.03

Table 16. Woodhouse's Toad call detection confidence set for May 2013. The model for environmental covariates 2 was taken out because it would not converge. Also, adjacent vegetation, size, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	6	55.67	0	0.55	0.55	0.11	0.65
Moonshine	7	57.79	2.12	0.19	0.73	7.85	4.74
Wind	7	59.87	4.2	0.07	0.80	-0.44	0.65
Water Temperature	7	59.93	4.27	0.06	0.87	0.26	0.33
Time	7	59.94	4.27	0.06	0.93	0.23	0.59
Air Temperature	7	60.31	4.65	0.05	0.98	0.09	0.24

Table 17. Woodhouse's Toad call detection confidence set for June 2013. Adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	86.22	0	0.40	0.40	-1.20	0.31
Moonshine	9	86.58	0.36	0.33	0.73	-11.48	6.04
Air Temperature	9	87.63	1.41	0.20	0.92	-0.12	0.11
Size	10	90.08	3.86	0.06	0.98		
Small						-1.82	0.64
Medium						1.13	0.75
Large						0.42	1.02

Table 18. Great Plains Toad call detection confidence set for June 2013. The model for environmental covariates 2 was taken out because it would not converge. Also, adjacent vegetation, size, bend, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Air Temperature	6	32.35	0	0.62	0.62	-0.80	0.35
Water Temperature	6	33.93	1.58	0.28	0.90	-0.71	0.30
Water Temperature + Air Temperature + Wind	8	37.91	5.57	0.04	0.94	0.25	0.64
Water Temperature + Air Temperature + Wind						-1.21	0.91
Water Temperature + Air Temperature + Wind						-1.35	2.00
Water Temperature + Air Temperature + Wind + Moonshine	9	38.68	6.33	0.03	0.97	0.54	0.75
Water Temperature + Air Temperature + Wind + Moonshine						-1.48	1.13
Water Temperature + Air Temperature + Wind + Moonshine						-2.20	2.33
Water Temperature + Air Temperature + Wind + Moonshine						-36.99	46.79

Table 19. Gray Treefrog Complex call detection confidence set for May 2013. The model for moonshine was taken out because it would not converge. Also, adjacent vegetation, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	57.52	0	0.86	0.86	0.66	0.44
Air Temperature	8	62.95	5.43	0.06	0.92	0.17	0.17
Wind	8	63.63	6.11	0.04	0.96	0.44	0.72

Table 20. Gray Treefrog Complex call detection confidence set for June 2013. The model for water temperature was taken out because it would not converge. Also, adjacent vegetation, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Wind	8	108.11	0	0.40	0.40	-0.37	0.72
Null	7	108.45	0.34	0.34	0.74	1.22	0.30
Size	9	111.32	3.21	0.08	0.82		
Small						1.75	0.65
Medium						-0.38	0.79
Large						-0.14	1.05
Time	8	111.37	3.26	0.08	0.89	-0.01	0.03
Moonshine	8	111.48	3.37	0.07	0.97	0.05	2.10

Table 21. Plains Leopard Frog call detection confidence set for April 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included. Also, water temperature and time were removed from the global detection model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Moonshine	11	65.87	0	0.87	0.87	5.98	3.47
Null	10	70.59	4.72	0.08	0.95	1.52	0.943

Table 22. Plains Leopard Frog call detection confidence set for May 2013. Adjacent vegetation, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	61.53	0	0.63	0.63	1.64	0.82
Moonshine	8	64.64	3.11	0.13	0.77	4.65	3.45
Air Temperature	8	65.22	3.69	0.10	0.87	-0.31	0.23
Wind	8	66.52	4.99	0.05	0.92	-0.47	0.53
Time	8	66.99	5.47	0.04	0.96	-0.10	0.25

Table 23. Plains Leopard Frog call detection confidence set for June 2013. Adjacent vegetation, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	120.02	0	0.34	0.34	-0.01	0.24
Air Temperature	8	121.09	1.07	0.20	0.53	0.09	0.07
Water Temperature	8	121.88	1.86	0.13	0.67	0.13	0.09
Time	8	121.92	1.9	0.13	0.79	-0.03	0.03
Size	9	122.74	2.72	0.09	0.88		
Small						-0.06	0.42
Medium						0.50	0.57
Large						-0.56	0.63
Wind	8	123.36	3.34	0.06	0.94	0.16	0.50
Moonshine	8	124.02	4	0.05	0.99	-1.57	1.72

Table 24. American Bullfrog call detection confidence set for May 2013. The models for environmental covariates 1 and environmental covariates 2 were taken out because they would not converge. Also, adjacent vegetation, ephemeral, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	39.9	0	0.58	0.58	-1.61	0.78
Moonshine	9	41.46	1.55	0.27	0.85	-6.44	8.67
Wind	9	43.78	3.87	0.08	0.94	1.16	0.88
Water Temperature	9	46.51	6.61	0.02	0.96	0.24	0.45

Table 25. American Bullfrog call detection confidence set for June 2013. The model for size was taken out because it would not converge. Also, adjacent vegetation, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	SE
Water Temperature	10	67.76	0	0.58	0.58	0.69	0.27
Water Temperature + Air Temperature + Wind	12	68.77	1.01	0.35	0.94	1.33	0.50
Water Temperature + Air Temperature + Wind						-0.28	0.25
Water Temperature + Air Temperature + Wind						-1.53	0.73
Water Temperature + Air Temperature + Wind + Moonshine	13	72.38	4.62	0.06	1	1.23	0.49
Water Temperature + Air Temperature + Wind + Moonshine						-0.26	0.25
Water Temperature + Air Temperature + Wind + Moonshine						-1.44	0.74
Water Temperature + Air Temperature + Wind + Moonshine						1.89	3.92

Table 26. Boreal Chorus Frog call detection confidence set for April 2013. Adjacent vegetation, size, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	66.01	0	0.62	0.62	0.66	0.39
Moonshine	8	68.54	2.54	0.17	0.79	7.01	3.90
Water Temperature	8	70.45	4.44	0.07	0.86	0.07	0.09
Air Temperature	8	70.46	4.46	0.07	0.93	0.06	0.07
Wind	8	70.81	4.81	0.06	0.98	0.68	1.34

Table 27. Boreal Chorus Frog call detection confidence set for May 2013. Adjacent vegetation, size, ephemeral, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	7	65.29	0	0.69	0.69	0.52	0.52
Moonshine	8	68.56	3.27	0.14	0.83	4.32	3.29
Time	8	70.75	5.46	0.05	0.88	0.39	0.96
Water Temperature	8	71	5.71	0.04	0.92	0.05	0.23
Wind	8	71.04	5.75	0.04	0.95	0.08	0.54

Table 28. Boreal Chorus Frog call detection confidence set June 2013. Adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Moonshine	9	90.94	0	0.42	0.42	-8.97	4.84
Time	9	91.79	0.85	0.28	0.70	0.09	0.05
Null	8	92.95	2.01	0.15	0.85	-0.07	0.69
Wind	9	95.11	4.18	0.05	0.90	-0.94	0.92
Water Temperature	9	95.83	4.9	0.04	0.94	-0.08	0.14
Air Temperature	9	96.12	5.18	0.03	0.97	-0.03	0.12

Table 29. Blanchard's Cricket Frog call occupancy confidence set for April 2012. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland + Land Cover						-1.21	0.79
Distance to Nearest Wetland + Land Cover	6	48.1	0	0.67	0.67	-0.03	NaN
Distance to Nearest Wetland + Land Cover							
Agriculture						25.61	NaN
Forest						-9.22	NaN
Aquatic Vegetation	4	50.8	2.7	0.17	0.84	8.39	22.60
Distance to Nearest Wetland	4	52.92	4.82	0.06	0.90	-0.003	0.002
Null	3	53.57	5.47	0.04	0.94	0.16	1.13
Land Cover	5	54.72	6.63	0.02	0.96		
Agriculture						0.29	15006
Forest						-0.99	15006

Table 30. Blanchard's Cricket Frog call occupancy confidence set for May 2012. The model for land cover was taken out because it would not converge. Also, adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	4	81.97	0	0.28	0.28		
Hamburg						-0.41	0.70
Kansas						1.57	1.19
Langdon						8.15	50.24
Null	2	82.4	0.43	0.23	0.51	0.78	0.64
Ephemeral	3	84.17	2.2	0.09	0.60	-0.95	1.13
Slope	3	84.69	2.72	0.07	0.68	0.32	0.52
Aquatic Vegetation	3	84.81	2.84	0.07	0.74	0.34	0.49
Distance to Nearest Wetland	3	84.93	2.96	0.06	0.81	0.0004	0.0012
Adjacent Vegetation	5	84.96	2.98	0.06	0.87		
Bare						-2.57	1.76
Herb/Forb						0.0009	1.54
Tree/Shrub						-9.00	48.45
Combination						1.43	1.37
Distance to Nearest Wetland + Land Cover	5	84.96	2.99	0.06	0.93	0.0013	0.0020
Distance to Nearest Wetland + Land Cover							
Agriculture						3.51	2.37E+04
Forest						-2.16	2.37E+04
Aquatic Vegetation + Adjacent Vegetation	6	86.05	4.08	0.04	0.97	-8.91	67.09
Aquatic Vegetation + Adjacent Vegetation							
Bare						-10.17	67.09
Herb/Forb						1.00	1.49
Tree/Shrub						-11.18	238.49
Combination						9.33	67.08

Table 31. Blanchard's Cricket Frog call occupancy confidence set for June 2012. The models for land cover, distance to the nearest wetland, and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to nearest wetland, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	3	58.82	0	0.33	0.33	3.76	3.33
Bend	4	59.18	0.35	0.28	0.61		
Hamburg						8.92	32.70
Kansas						-9.65	32.70
Langdon						-8.29	32.70
Ephemeral	3	60.8	1.98	0.12	0.73	8.47	44.07
Null	2	60.99	2.17	0.11	0.84	1.22	1.01
Size	4	61.8	2.97	0.07	0.92		
Small						2.64	4.83
Medium						2.40	26.87
Large						-3.23	4.83
Aquatic Vegetation + Adjacent	5	63.09	4.27	0.04	0.96	15.54	69.55
Aquatic Vegetation + Adjacent							
Bare						-14.36	69.63
Herb/Forb						-7.67	39.47
Combination						0.57	1.98

Table 32. Woodhouse's Toad call occupancy confidence set for April 2012. The model for aquatic vegetation plus adjacent vegetation was taken out because it would not converge. Also, size, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	5	23.62	0	0.33	0.33		
Hamburg						-0.51	0.73
Kansas						-14.45	591.78
Langdon						-12.91	581.54
Null	3	24.07	0.46	0.26	0.58	-1.70	0.63
Aquatic Vegetation	4	25.25	1.63	0.14	0.73	14.66	70.90
Ephemeral	4	26.56	2.94	0.07	0.8	5.84	34.69
Slope	4	26.84	3.22	0.07	0.87	-4.21	8.57
Distance to Nearest Wetland	4	28.08	4.46	0.03	0.9	0.04	0.10
Size	5	28.18	4.56	0.03	0.94		
Small						-1.11	0.82
Medium						-0.81	1.34
Langdon						-9.84	118.76
Land Cover	5	28.35	4.73	0.03	0.97		
Agriculture						0.19	27397
Forest						-1.32	27397

Table 33. Gray Treefrog call occupancy confidence set for April 2012. The models for adjacent vegetation and adjacent vegetation plus aquatic vegetation were taken out because they would not converge. Also, adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	10.39	0	0.85	0.85	-173.6	160.5
Aquatic Vegetation	4	14.35	3.95	0.12	0.96	39.8	54795

Table 34. Gray Treefrog Complex call occupancy confidence set for June 2012. The model for distance to the nearest wetland plus land cover was taken out because it would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Ephemeral	3	27.83	0	0.76	0.76	4.33	2.30
Adjacent Vegetation	4	33.38	5.55	0.05	0.81		
Bare						8.17	39.10
Herb/Forb						9.34	39.00
Combination						-9.41	39.00
Null	2	33.43	5.6	0.05	0.85	-1.30	0.61
Aquatic Vegetation	3	33.73	5.89	0.04	0.89	8.07	43.00
Bend	4	34.12	6.29	0.03	0.92		
Hamburg						-0.07	0.90
Kansas						-9.67	52.32
Langdon						-1.58	1.39
Aquatic Vegetation + Adjacent Vegetation	5	34.46	6.62	0.03	0.95	8.74	53.20
Aquatic Vegetation + Adjacent Vegetation							
Bare						9.12	49.40
Herb/Forb						9.88	49.40
Combination						-18.37	72.60

Table 35. Plains Leopard Frog call occupancy confidence set for April 2012. The models for distance to the nearest wetland, and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to nearest wetland, size, adjacent vegetation were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	3	52.66	0	0.68	0.68	1.62	0.72
Bend	5	57.19	4.52	0.07	0.75		
Hamburg						1.16	0.93
Kansas						0.29	1.22
Langdon						5.89	19.36
Size	5	57.32	4.66	0.07	0.81		
Small						1.73	1.14
Medium						1.42	4.00
Large						-0.97	1.45
Aquatic Vegetation	4	57.64	4.98	0.06	0.87	4.38	NaNs
Ephemeral	4	57.64	4.98	0.06	0.92	1.91	NaNs
Slope	4	57.64	4.98	0.06	0.98	1.47	NaNs

Table 36. Plains Leopard Frog Call Occupancy Confidence Set for May 2012. The models for adjacent vegetation and adjacent vegetation plus aquatic vegetation were taken out because they would not converge. Also, adjacent vegetation and bend were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	4	60.47	0	0.31	0.31	-0.002	0.001
Null	3	60.92	0.45	0.24	0.55	0.64	0.58
Slope	4	61.97	1.49	0.14	0.69	2.43	7.30
Size	5	62.8	2.33	0.10	0.79		
Small						-0.23	0.82
Medium						4.07	9.88
Large						0.74	1.27
Ephemeral	4	63.37	2.89	0.07	0.86	-0.55	1.14
Aquatic Vegetation	4	63.53	3.05	0.07	0.93	-0.40	1.67
Land Cover	5	65.06	4.59	0.03	0.96		
Agriculture						0.782	NaNs
Forest						-0.422	NaNs

Table 37. Plains Leopard Frog call occupancy confidence set for June 2012. The model for distance to the nearest wetland plus land cover was taken out because it would not converge.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Ephemeral	4	31.94	0	0.47	0.47	20.46	8522.83
Size	5	32.86	0.92	0.30	0.77		
Small						-0.19	1.42
Medium						13.09	726.03
Large						-18.30	4566.19
Aquatic Vegetation	4	34.94	2.99	0.11	0.87	10.20	85.40
Null	3	35.92	3.97	0.06	0.94	-0.45	0.59
Slope	4	38.64	6.7	0.02	0.95	3.94	6.60

Table 38. American Bullfrog call occupancy confidence set for April 2012. The models for land cover and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, ephemeral was removed from the global occupancy model because the models would not converge with this covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	2	32.73	0	0.26	0.26	5.06	29.00
Slope	3	33.43	0.7	0.18	0.45	-34.10	38.78
Ephemeral	3	33.51	0.78	0.18	0.62	-14.58	85.90
Size	4	34	1.28	0.14	0.76		
Small						-8.32	40.90
Medium						10.52	41.70
Large						15.39	69.90
Aquatic Vegetation	3	34.25	1.52	0.12	0.88	-6.80	41.80
Distance to Nearest Wetland	3	35.3	2.58	0.07	0.96	0.001	0.004

Table 39. American Bullfrog call occupancy confidence set for May 2012. Also, adjacent vegetation was removed from the global occupancy model because the models would not converge with this covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	4	50.09	0	0.74	0.74		
Hamburg						-8.32	27.80
Kansas						19.03	434.70
Langdon						11.67	31.50
Slope	3	54.5	4.41	0.08	0.83	23.23	16.87
Null	2	54.97	4.88	0.06	0.89	0.73	2.42
Aquatic Vegetation	3	57.1	7.01	0.02	0.91	1.12	2.46
Ephemeral	3	57.43	7.35	0.02	0.93	0.30	2.44
Distance to the Nearest Wetland	3	57.56	7.47	0.02	0.95	0.0004	0.002

Table 40. American Bullfrog call occupancy confidence set for June 2012. The model for distance to the nearest wetland plus land cover was taken out because it would not converge. Also, bend, size, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included. There were only five wetlands out of 21 that the American Bullfrog occurred at in June 2012. The low occupancy caused low inference and all the models except the top model would not produce estimates and standard errors.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	3	32.94	0	0.39	0.39	53.66	48.03
Size	4	33.49	0.55	0.30	0.69		
Small						-15.3	698.6
Medium						-15.3	698.6
Large						12.75	NaNs
Null	2	35.33	2.39	0.12	0.81	4.04	NaNs
Ephemeral	3	35.81	2.86	0.09	0.90	-10.75	170.4
Aquatic Vegetation	3	37.67	4.72	0.04	0.94	-7.05	NaNs
Distance to Nearest Wetland	3	38.06	5.11	0.03	0.97	0.0027	0.003

Table 41. Boreal Chorus Frog call occupancy confidence set for April 2012. . Also, adjacent vegetation was removed from the global occupancy model because the models would not converge with it included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	44.54	0	0.43	0.43	7.28	29.45
Null	3	46.22	1.69	0.18	0.61	0.17	0.69
Ephemeral	4	47.15	2.61	0.12	0.72	6.12	32.36
Distance to Nearest Wetland	4	47.6	3.06	0.09	0.82	-0.003	0.01
Bend	5	47.72	3.18	0.09	0.90		
Hamburg						0.94	1.29
Kansas						-0.15	1.59
Langdon						-8.03	21.88
Slope	4	48.92	4.38	0.05	0.95	-2.24	6.42

Table 42. Blanchard's Cricket Frog call occupancy confidence set for May 2013. The models for distance to the nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, adjacent vegetation, distance to the nearest wetland , agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	3	57.36	0	0.49	0.49	19.30	11.03
Null	2	59.35	2	0.18	0.67	1.61	1.15
Aquatic Vegetation	3	60.08	2.72	0.13	0.80	2.74	2.34
Ephemeral	3	60.72	3.36	0.09	0.89	7.64	44.85
Land Cover	4	62.42	5.07	0.04	0.93		
Agriculture						6.97	19373
Forest						-3.21	19373
Bend	4	62.78	5.43	0.03	0.96		
Hamburg						0.39	0.92
Kansas						4.33	23.62
Langdon						4.33	23.62

Table 43. Blanchard's Cricket Frog call occupancy confidence set for June 2013. The models for distance to the nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, adjacent vegetation, distance to the nearest wetland were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	3	118	0	0.70	0.70	3.10	2.71
Null	2	121.62	3.62	0.11	0.82	1.56	0.75
Slope	3	121.93	3.93	0.10	0.92	-6.21	4.64
Ephemeral	3	123.88	5.88	0.04	0.95	0.35	1.49

Table 44. Woodhouse's Toad call occupancy confidence set for May 2013. The model for distance to the nearest wetland plus land cover was taken out because it would not converge. Also, adjacent vegetation and bend were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	4	46.72	0	0.33	0.33		
Hamburg						1.16	1.77
Kansas						-10.80	62.29
Langdon						5.56	63.90
Ephemeral	3	47.36	0.64	0.24	0.57	9.29	46.71
Aquatic Vegetation	3	48.33	1.61	0.15	0.72	8.70	33.50
Slope	3	48.82	2.1	0.12	0.84	-35.70	49.70
Null	2	48.94	2.22	0.11	0.95	0.51	1.22

Table 45. Woodhouse's Toad call occupancy confidence set for June 2013. The model for land cover was taken out because it would not converge. Also, adjacent vegetation agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Size	4	79.45	0	0.23	0.23		
Small						-0.98	1.10
Medium						4.17	12.67
Large						0.46	1.36
Slope	3	79.55	0.1	0.22	0.45	-55.20	102.40
Null	2	79.89	0.45	0.18	0.63	0.28	1.21
Ephemeral	3	80.07	0.62	0.17	0.80	2.76	7.17
Distance to Nearest Wetland	3	81.9	2.45	0.07	0.87	0.001	0.002
Aquatic Vegetation	3	82.14	2.69	0.06	0.93	0.33	1.25
Bend	4	83.08	3.63	0.04	0.97		
Hamburg						1.74	3.84
Kansas						-2.02	3.32
Langdon						-1.70	3.30

Table 46. Great Plains Toad call occupancy confidence set for June 2013. The model for distance to the nearest wetland plus land cover was taken out because it would not converge. Also, size and adjacent vegetation were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	29.57	0	0.64	0.64	-45.98	50.82
Distance to Nearest Wetland	4	31.55	1.98	0.24	0.88	-0.01	0.01
Null	3	34.64	5.07	0.05	0.93	4.32	13.30
Size	5	36.1	6.53	0.02	0.96		
Small						6.57	34.20
Medium						-7.44	34.10
Large						-15.75	88.90

Table 47. Gray Treefrog call occupancy confidence set for May 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also, distance to nearest wetland, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	3	53.05	0	0.62	0.62	9.27	36.80
Null	2	55.71	2.66	0.16	0.79	0.58	0.66
Land Cover	4	58.1	5.05	0.05	0.84		
Agriculture						1.61	16777
Forest						-0.96	16777
Ephemeral	3	58.3	5.25	0.05	0.88	0.77	1.93
Distance to Nearest Wetland	3	58.48	5.43	0.04	0.92	-7.91E-05	0.00112
Slope	3	58.5	5.44	0.04	0.96	-0.25	3.45

Table 48. Gray Treefrog call occupancy confidence set for June 2013. The models for distance to nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to the nearest wetland was removed from the global occupancy model because the models would not converge with it included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	106.29	0	0.51	0.51	2.29	1.05
Slope	4	107.71	1.42	0.25	0.77	-8.48	4.38
Null	3	110.08	3.79	0.08	0.84	1.44	0.47
Ephemeral	4	110.58	4.29	0.06	0.91	2.02	2.49
Aquatic Vegetation + Adjacent Vegetation	8	111.38	5.08	0.04	0.95	2.33	1.08
Aquatic Vegetation + Adjacent Vegetation							
Bare						0.98	1.33
Grass						0.60	1.42
Herb/Forb						10.96	96.18
Tree/Shrub						6.56	40.28
Combination						-0.80	0.91

Table 49. Plains Leopard Frog call occupancy confidence set for April 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	3	44.57	0	0.29	0.29	-0.73	0.47
Adjacent Vegetation	5	45.57	0.99	0.18	0.47		
Grass						1.25	1.36
Herb/Forb						-9.20	59.08
Combination						-0.54	0.57
Distance to Nearest Wetland	4	46.25	1.68	0.13	0.60	0.001	0.001
Bend	5	46.49	1.91	0.11	0.71		
Hamburg						0.12	0.88
Kansas						-0.09	1.21
Langdon						-2.30	1.38
Aquatic Vegetation	4	47.29	2.72	0.07	0.78	0.54	1.00
Ephemeral	4	47.49	2.91	0.07	0.85	-0.41	1.27
Slope	4	47.51	2.93	0.07	0.92	-1.04	3.64
Aquatic Vegetation + Adjacent Vegetation	6	47.83	3.25	0.06	0.97	1.48	1.27
Aquatic Vegetation + Adjacent Vegetation							
Grass						1.87	1.58
Herb/Forb						-9.85	70.48
Combination						-1.56	1.12

Table 50. Plains Leopard Frog call occupancy confidence set for May 2013. The models for distance to nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to the nearest wetland, size, and adjacent vegetation were removed from the global occupancy model because the models would not converge with them included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	3	50.31	0	0.79	0.79	9.73	37.30
Null	2	54.86	4.55	0.08	0.88	0.80	0.55
Ephemeral	3	55.12	4.81	0.07	0.95	8.29	46.59

Table 51. Plains Leopard Frog call occupancy confidence set for June 2013. The models for distance to nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to the nearest wetland, and adjacent vegetation were removed from the global occupancy model because the models would not converge with them included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	119.32	0	0.30	0.30	-9.60	5.70
Adjacent Vegetation	7	119.56	0.25	0.26	0.56		
Bare						4.41	56.46
Grass						-2.32	1.43
Herb/Forb						9.60	100.36
Shrub/Tree						-10.08	66.15
Combination						1.22	0.85
Land Cover	5	121.28	1.97	0.11	0.67		
Agriculture							
Forest							
Null	3	121.54	2.23	0.10	0.77	1.64	1.10
Aquatic Vegetation + Adjacent Vegetation	8	122.35	3.03	0.07	0.84	0.57	1.44
Aquatic Vegetation + Adjacent Vegetation							
Bare						2.72	8.94
Grass						-2.39	1.48
Herb/Forb						7.04	41.85
Shrub/Tree						-10.95	94.52
Combination						0.82	1.23
Aquatic Vegetation	4	122.89	3.57	0.05	0.89	0.56	1.50
Bend	5	123.77	4.46	0.03	0.92		
Hamburg						4.96	21.80
Kansas						-4.11	21.70
Langdon						-2.90	21.40
Ephemeral	4	123.98	4.66	0.03	0.95	0.11	1.72

Table 52. American Bullfrog call occupancy confidence set for May 2013. The covariates bend, size, and adjacent vegetation were removed from the global occupancy model because the models would not converge with them included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland + Land Cover						1.69	3.29
Distance to Nearest Wetland + Land Cover	5	20.17	0	0.2	0.2	0.03	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						3.50	NaNs
Forest						-32.25	NaNs
Slope	3	20.23	0.06	0.20	0.40	47.30	66.80
Null	2	20.59	0.42	0.16	0.56	5.42	63.80
Distance to Nearest Wetland	3	20.64	0.47	0.16	0.72	0.03	0.04
Bend	4	21.48	1.31	0.11	0.83		
Hamburg						-9.78	84.90
Kansas						-3.78	655.80
Langdon						16.95	112.30
Aquatic Vegetation	3	22.71	2.54	0.06	0.89	13.26	100.00
Ephemeral	3	22.71	2.54	0.06	0.94	-12.82	95.20
Land Cover	4	23.22	3.06	0.04	0.99	-1.28	38745
Agriculture						8.24	38745
Forest						-9.52	38745

Table 53. American Bullfrog call occupancy confidence set for June 2013.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	62.29	0	0.93	0.93	46.06	37.74
Aquatic Vegetation	4	68.79	6.49	0.04	0.97	5.68	27.28

Table 54. Boreal Chorus Frog call occupancy confidence set for April 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also, distance to the nearest wetland, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	2	60.66	0	0.22	0.22	0.62	0.55
Aquatic Vegetation	3	60.93	0.27	0.20	0.42	1.65	1.13
Ephemeral	3	61.01	0.35	0.19	0.61	-1.98	1.36
Size	4	61.11	0.44	0.18	0.79		
Small						-0.97	1.20
Medium						3.16	1.96
Large						0.86	1.45
Slope	3	62.73	2.07	0.08	0.87	-3.02	3.88
Adjacent Vegetation	4	63.92	3.25	0.04	0.91		
Grass						-1.84	1.53
Herb/Forb						-1.54	1.27
Combination						1.29	0.86
Bend	4	64.2	3.54	0.04	0.95		
Hamburg						2.38	2.35
Kansas						-2.18	2.46
Langdon						-2.18	2.39

Table 55. Boreal Chorus Frog call occupancy confidence set for May 2013. The models for distance to nearest wetland and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, distance to the nearest wetland, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with them included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	2	53.23	0	0.43	0.43	0.25	0.57
Ephemeral	3	55.62	2.4	0.13	0.56	0.98	1.73
Aquatic Vegetation	3	55.62	2.4	0.13	0.69	0.90	1.43
Distance to Nearest Wetland	3	55.62	2.4	0.13	0.81	-0.001	0.001
Slope	3	55.93	2.7	0.11	0.93	-0.85	2.77
Bend	4	57.83	4.6	0.04	0.97		
Hamburg						0.25	0.84
Kansas						0.76	1.23
Langdon						0.85	1.42

Table 56. Boreal Chorus Frog Call Occupancy Confidence Set for June 2013. Adjacent vegetation was removed from the global occupancy model because the models would not converge with it included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	80.5	0	0.45	0.45	2.15	1.19
Null	3	82.59	2.08	0.16	0.60	-0.41	0.50
Ephemeral	4	83.3	2.8	0.11	0.71	1.20	1.01
Distance to Nearest Wetland	4	83.5	2.99	0.10	0.81	-0.001	0.001
Aquatic Vegetation + Adjacent Vegetation	8	84.5	3.99	0.06	0.87	10.38	69.54
Aquatic Vegetation + Adjacent Vegetation							
Bare						9.98	69.54
Grass						-0.22	1.42
Herb/Forb						0.33	1.03
Shrub/Tree						10.77	120.84
Combination						-10.87	69.54
Slope	4	84.68	4.18	0.06	0.93	-2.11	3.66
Land Cover	5	86.6	6.1	0.02	0.95		
Agriculture						-0.56	10611
Forest						0.23	10611

APPENDIX B. DETECTION AND OCCUPANCY CONFIDENCE SETS FOR AMPHIBIAN LARVAE

Table 1. American Bullfrog tadpole detection confidence set for April 2012. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	10	81.34	0	0.43	0.43	1.64	0.69
Slope	11	82.83	1.5	0.21	0.64	-13.40	7.15
Water Temperature	11	83.44	2.1	0.15	0.79	0.16	0.13
Ephemeral	11	84.73	3.4	0.08	0.87	5.71	26.04
Aquatic Vegetation	11	85.4	4.07	0.06	0.93	0.24	1.45
Time	11	85.46	4.13	0.06	0.98	0.05	8.87

Table 2. American Bullfrog tadpole detection confidence set for May 2012. The models for water temperature, time, julian day, and water temperature plus slope were taken out because they would not converge. Also, ephemeral, aquatic vegetation, adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	47.48	0	0.75	0.75	14.30	376
Aquatic Vegetation	9	51.89	4.41	0.08	0.83	2.48	578
Ephemeral	9	51.9	4.42	0.08	0.92	NaNs	NaNs
Slope	9	51.9	4.42	0.08	1.00	0.32	14327

Table 3. American Bullfrog tadpole detection confidence set for June 2012. The models for water temperature, and water temperature plus slope were taken out because they would not converge. Also, size, bend, adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	6	43.85	0	0.55	0.55	2.00	1.11
Slope	7	45.36	1.5	0.26	0.81	42.73	31.71
Aquatic Vegetation	7	47.97	4.12	0.07	0.88	-5.52	27.20
Time	7	48.33	4.48	0.06	0.94	0.34	3.30
Ephemeral	7	48.45	4.6	0.06	1.00	NaNs	NaNs

Table 4. Boreal Chorus Frog tadpole detection confidence set for April 2012. Adjacent vegetation, distance to nearest wetland, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Time	10	48.72	0	0.42	0.42	5.08	5.60
Ephemeral	10	49.39	0.67	0.3	0.72	NaNs	NaNs
Null	9	50.76	2.04	0.15	0.87	1.46	0.83
Slope	10	51.49	2.77	0.1	0.97	45.85	32.88

Table 5. Boreal Chorus Frog tadpole detection confidence set for June 2012. Adjacent vegetation, distance to nearest wetland, aquatic vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	41.5	0	0.75	0.75	1.45	0.84
Ephemeral	9	45.36	3.87	0.11	0.86	-29.70	33554
Time	9	47.31	5.82	0.04	0.90	5.05	5.49
Water Temperature	9	47.69	6.2	0.03	0.94	-0.19	0.09
Slope	9	47.85	6.35	0.03	0.97	37.01	28.49

Table 6. Blanchard's Cricket Frog tadpole detection confidence set for June 2013. Adjacent vegetation, size, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	9	49.02	0	0.32	0.32	-59.10	15.50
Null	8	49.11	0.1	0.3	0.62	-0.76	0.48
Ephemeral	9	49.68	0.67	0.23	0.85	2.42	1.29
Slope	9	52	2.99	0.07	0.93	-14.52	8.99
Time	9	53.15	4.13	0.04	0.97	-0.71	0.40

Table 7. Smallmouth Salamander larvae detection confidence set for May 2013. Adjacent vegetation, size, ephemeral, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	8	34.58	0	0.35	0.35	18.70	23.68
Aquatic Vegetation	8	34.72	0.14	0.33	0.68	-16.30	1737
Null	7	35.46	0.88	0.23	0.90	-0.41	0.65
Water Temperature	8	39.38	4.8	0.03	0.93	-0.66	0.65
Time	8	39.55	4.97	0.03	0.96	0.63	0.69

Table 8. Toad species tadpole detection confidence set for May 2013. The models for water temperature, and water temperature plus slope were taken out because they would not converge. Also, size, bend, adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	7	26.55	0	0.70	0.70	-66.5	NaNs
Null	6	29.21	2.66	0.19	0.89	1.49	1.16
Water Temperature + Slope	8	31.39	4.84	0.06	0.95	-68.78	804.11
Water Temperature + Slope						-0.61	5.29

Table 9. Gray Treefrog Complex tadpole detection confidence set for June 2013. Adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	58.39	0	0.31	0.31	2.76	1.06
Time	9	59.04	0.65	0.23	0.54	-2.25	2.12
Ephemeral	9	59.38	0.99	0.19	0.73	-6.06	12.30
Water Temperature	9	60.24	1.84	0.12	0.86	-0.66	0.56
Aquatic Vegetation	9	61.56	3.17	0.06	0.92	-2.67	10.20
Slope	9	61.77	3.38	0.06	0.98	0.73	20.95

Table 10. Plains Leopard Frog tadpole detection confidence set for May 2013. Adjacent vegetation, size, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	61.54	0	0.73	0.73	1.66	0.85
Water Temperature	9	66.56	5.02	0.06	0.79	-0.32	0.44
Slope	9	66.59	5.05	0.06	0.85	5.97	6.91
Aquatic Vegetation	9	66.75	5.21	0.05	0.90	-3.85	9.64
Ephemeral	9	66.83	5.29	0.05	0.95	3.51	8.14

Table 11. Plains Leopard Frog tadpole detection confidence set for June 2013. Size, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	12	88.05	0	0.42	0.42	1.43	0.60
Aquatic Vegetation	13	88.77	0.72	0.29	0.71	2.49	1.38
Water Temperature	13	90.46	2.41	0.13	0.83	0.23	0.16
Ephemeral	13	92.16	4.11	0.05	0.89	-0.81	1.20
Time	13	92.54	4.48	0.04	0.93	0.10	0.34
Slope	13	92.61	4.56	0.04	0.97	0.05	9.92

Table 12. Boreal Chorus Frog tadpole detection confidence set for May 2013. The models for aquatic vegetation and ephemeral were taken out because they would not converge. Also, ephemeral, aquatic vegetation, adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	37.1	0	0.45	0.45	1.39	0.79
Slope	9	38.76	1.65	0.20	0.64	75.50	94.70
Water Temperature	9	38.81	1.71	0.19	0.83	-2.08	2.93
Time	9	39.14	2.04	0.16	0.99	-2.97	2.68

Table 13. Boreal Chorus Frog tadpole detection confidence set for June 2013. The model for ephemeral was taken out because it would not converge. Also, size, adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	9	61.16	0	0.74	0.74	-47.60	30.99
Water Temperature + Slope	10	63.63	2.48	0.21	0.96	-47.32	21.18
Water Temperature + Slope						0.44	0.38

Table 14. American Bullfrog tadpole occupancy confidence set for April 2012. Agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetaion	3	65.37	0	0.23	0.23	-1.39	0.77
Null	2	66.44	1.07	0.13	0.36	-0.82	0.37
Ephemeral	3	66.55	1.19	0.13	0.49	-1.50	1.14
Slope	3	66.65	1.28	0.12	0.61	5.50	3.83
Bend	4	67.04	1.67	0.1	0.71		
Hamburg						-1.92	0.76
Kansas						1.67	0.94
Langdon						1.67	1.09
Aquatic Vegetaion + Adjacent Vegetation	7	67.55	2.18	0.08	0.79	-2.08	1.09
Aquatic Vegetaion + Adjacent Vegetation							
Bare						1.30	1.53
Grass						-8.58	130.64
Herb/Forb						1.84	1.37
Tree/Shrub						13.26	134.32
Combination						0.85	0.39
Distance to Nearest Wetland	3	68.04	2.68	0.06	0.85	-0.0007	0.0008
Size	4	68.57	3.2	0.05	0.89		
Small						-1.76	0.77
Medium						1.33	0.96
Langdon						1.40	1.01
Adjacent Vegetation	6	68.65	3.28	0.04	0.94		
Bare						1.97	1.48
Grass						-7.75	89.88
Herb/Forb						1.04	1.18
Tree/Shrub						10.92	63.79
Combination						-1.92	1.07
Land Cover	4	68.68	3.31	0.04	0.98		
Agriculture						0.34	23727
Forest						-0.88	23727

Table 15. American Bullfrog tadpole occupancy confidence set for May 2012. The model for distance to nearest forest plus land cover was taken out because it would not converge. Also, distance to the nearest wetland, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Ephemeral	3	33.4	0	0.29	0.29	-9.86	104.31
Null	2	33.58	0.17	0.26	0.55	-1.30	0.46
Slope	3	34.13	0.73	0.20	0.75	-8.04	6.49
Aquatic Vegetation	3	35.98	2.58	0.08	0.83	0.33	0.97
Distance to Nearest Wetland	3	36.18	2.77	0.07	0.91	0.0001	0.0008
Land Cover	4	37.57	4.16	0.04	0.94		
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs
Size	4	38.28	4.87	0.03	0.97		
Small						-0.92	0.84
Medium						-0.79	1.14
Large						-0.18	1.17

Table 16. American Bullfrog tadpole occupancy confidence set for June 2012. The model for distance to nearest forest plus land cover was taken out because it would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	2	34.11	0	0.26	0.26	-1.15	0.52
Adjacent Vegetation	4	35	0.89	0.17	0.43		
Bare						-9.25	50.79
Herb/Forb						-1.58	1.28
Combination						-0.20	0.68
Ephemeral	3	35.07	0.96	0.16	0.59	-7.95	49.52
Distance to Nearest Wetland	3	35.48	1.37	0.13	0.73	-0.001	0.001
Aquatic Vegetation	3	36.8	2.69	0.07	0.79	0.29	1.26
Slope	3	36.82	2.71	0.07	0.86	-1.23	6.85
Size	4	37.48	3.37	0.05	0.91		
Small						-8.71	39.10
Medium						7.88	39.10
Large						7.81	39.10
Land Cover	4	37.81	3.70	0.04	0.95		
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs

Table 17. Boreal Chorus Frog tadpole occupancy confidence set for April 2012. The models for adjacent vegetation and adjacent vegetation plus aquatic vegetation were taken out because they would not converge. Also, adjacent vegetation and distance to the nearest wetland were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	39.73	0	0.50	0.50	14.90	711
Size	5	41.92	2.19	0.17	0.67		
Small						-0.58	0.56
Medium						-1.91	1.18
Large						-16.37	1512
Distance to Nearest Wetland	4	42.81	3.08	0.11	0.78	-0.003	0.001
Bend	5	43.71	3.98	0.07	0.84		
Hamburg						-0.78	0.54
Kansas						-1.78	1.17
Langdon						-16.03	1688
Null	3	44.27	4.54	0.05	0.90	-1.63	0.45
Ephemeral	4	44.5	4.77	0.05	0.94	1.42	0.94
Distance to Nearest Wetland + Land Cover	6	45.56	5.82	0.03	0.97	NaNs	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs

Table 18. Boreal Chorus Frog tadpole occupancy confidence set for June 2012. The models for land cover and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, adjacent vegetation, distance to the nearest wetland, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	4	25.05	0	0.38	0.38		
Hamburg						6.95	44.80
Kansas						-17.10	107.30
Langdon						-17.10	107.30
Null	2	26.28	1.23	0.2	0.58	5.53	45.80
Distance to Nearest Wetland	3	27.3	2.25	0.12	0.70	NaNs	NaNs
Aquatic Vegetation	3	27.33	2.28	0.12	0.82	14.84	95.40
Ephemeral	3	28.5	3.45	0.07	0.89	6.26	36.42
Slope	3	29.03	3.98	0.05	0.94	NaNs	NaNs
Size	4	29.57	4.52	0.04	0.98		
Small						6.87	63.60
Medium						NaNs	NaNs
Large						-16.09	105.60

Table 19. Blanchard's Cricket Frog tadpole occupancy confidence set for June 2013. Adjacent vegetation, agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	4	44.54	0	0.28	0.28	-0.004	0.003
Size	5	45.09	0.55	0.21	0.5		
Small						-0.76	0.87
Medium						-10.00	62.80
Large						-0.46	1.12
Null	3	45.62	1.09	0.16	0.66	-1.66	0.56
Slope	4	46.04	1.5	0.13	0.8	-7.92	6.41
Ephemeral	4	47.3	2.77	0.07	0.87	0.97	1.06
Aquatic Vegetation	4	47.61	3.07	0.06	0.93	0.91	1.38
Distance to Nearest Wetland + Land Cover	6	49.61	5.08	0.02	0.95	NaNs	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs

Table 20. Smallmouth Salamander larvae occupancy confidence set for May 2013. Agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	5	24.91	0	0.45	0.45		
Hamburg						-8.48	51.10
Kansas						-3.65	217.60
Langdon						13.68	101.90
Null	3	26.12	1.2	0.25	0.70	-0.84	0.75
Aquatic Vegetation	4	28.75	3.84	0.07	0.77	-1.06	1.65
Ephemeral	4	29.05	4.14	0.06	0.82	-5.61	62.62
Slope	4	29.14	4.22	0.05	0.88	0.31	4.66
Distance to Nearest Wetland	4	29.3	4.39	0.05	0.93	-0.0009	0.0008
Land Cover	5	29.57	4.66	0.04	0.97		
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs

Table 21. Toad species tadpole occupancy confidence set for May 2013. The models for land cover and distance to the nearest wetland plus land cover were taken out because they would not converge. Also, adjacent vegetation, distance to the nearest wetland, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	5	23.25	0	0.41	0.41		
Hamburg						-24.95	9.13E-01
Kansas						-2.47	1.34E+05
Langdon						25.36	0.00E+00
Slope	4	23.65	0.4	0.33	0.74	42.40	29.93
Null	3	26.62	3.36	0.08	0.81	-1.66	0.63
Distance to Nearest Wetland + Land Cover	6	27.46	4.21	0.05	0.86	NaNs	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs
Land Cover	5	27.86	4.61	0.04	0.90		
Agriculture						3.89	23727
Forest						-8.43	23727
Aquatic Vegetation	4	28.09	4.83	0.04	0.94	10.10	158.00
Ephemeral	4	28.9	5.64	0.02	0.96	-11.00	369.84

Table 22. Gray Treefrog Complex tadpole occupancy confidence set for June 2013. Agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	3	52.06	0	0.35	0.35	-0.004	0.003
Slope	3	52.4	0.34	0.29	0.64	-8.50	4.95
Null	2	54.13	2.07	0.12	0.76	-1.20	0.38
Aquatic Vegetation	3	55.01	2.95	0.08	0.84	1.23	1.13
Ephemeral	3	55.83	3.77	0.05	0.89	0.69	0.84
Distance to Nearest Wetland + Land Cover	5	57.16	5.10	0.03	0.92	NaNs	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs
Size	4	57.56	5.50	0.02	0.94		
Small						-0.81	0.60
Medium						-1.06	0.97
Large						-0.17	0.91
Adjacent Vegetation	6	57.73	5.67	0.02	0.96		
Bare						-8.44	90.50
Grass						0.29	1.28
Herb/Forb						1.68	0.89
Tree/Shrub						-7.99	88.69
Combination						-1.67	0.63

Table 23. Plains Leopard Frog tadpole occupancy confidence set for May 2013. The model for distance to nearest forest plus land cover was taken out because it would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	2	43.3	0	0.34	0.34	-0.72	0.47
Distance to Nearest Wetland	3	43.96	0.66	0.25	0.59	0.001	0.001
Aquatic Vegetation	3	45.56	2.26	0.11	0.70	0.79	1.24
Ephemeral	3	45.69	2.39	0.10	0.81	0.86	1.54
Slope	3	45.99	2.69	0.09	0.89	-0.31	2.54
Size	4	47.25	3.94	0.05	0.94		
Small						-0.65	0.88
Medium						0.48	1.12
Large						-1.11	1.40
Bend	4	48.08	4.78	0.03	0.97		
Hamburg						-4.65E-01	0.75
Kansas						3.70E-06	1.05
Langdon						-1.11E+00	1.33

Table 24. Plains Leopard Frog tadpole occupancy confidence set for June 2013. Agriculture and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	3	68.73	0	0.58	0.58	-10.59	4.90
Bend	4	70.58	1.85	0.23	0.82		
Hamburg						0.44	0.63
Kansas						-1.39	0.86
Langdon						-2.79	1.22
Null	2	73.3	4.57	0.06	0.88	-0.75	0.36
Size	4	74.21	5.49	0.04	0.91		
Small						-0.40	0.59
Medium						0.07	0.80
Large						-1.85	1.21
Ephemeral	3	74.67	5.94	0.03	0.94	0.81	0.81
Aquatic Vegetation	3	75.13	6.40	0.02	0.97	-0.57	0.79

Table 25. Boreal Chorus Frog tadpole occupancy confidence set for May 2013. Ephemeral, adjacent vegetation and aquatic vegetation were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	3	34.22	0	0.37	0.37	-0.004	0.002
Land Cover	4	35.01	0.8	0.25	0.61		
Agriculture						3.69	33554
Forest						-7.45	33554
Bend	4	37.17	2.96	0.08	0.70		
Hamburg						-9.29	38.00
Kansas						8.28	38.00
Langdon						9.42	38.00
Distance to Nearest Wetland + Land Cover	5	37.33	3.11	0.08	0.77	-0.002	1.67E-03
Distance to Nearest Wetland + Land Cover							
Agriculture						2.43	1.94E+04
Forest						-4.29	1.94E+04
Aquatic Vegetation	3	37.43	3.22	0.07	0.85	8.31	43.50
Null	2	37.72	3.50	0.06	0.91	-1.14	0.53
Size	4	38.95	4.74	0.03	0.95		
Small						-9.22	42.30
Medium						8.05	42.30
Large						9.05	42.30

Table 25. Boreal Chorus Frog tadpole occupancy confidence set for May 2013. The models for adjacent vegetation and adjacent vegetation plus aquatic vegetation were taken out because they would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the global model would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	4	48.62	0	0.55	0.55	-0.004	0.002
Null	3	51.06	2.44	0.16	0.71	-0.74	0.41
Slope	4	53.02	4.40	0.06	0.78	5.52	7.99
Ephemeral	4	53.25	4.63	0.05	0.83	-0.51	0.95
Aquatic Vegetation	4	53.48	4.86	0.05	0.88	0.26	0.96
Distance to Nearest Wetland + Land Cover	6	53.5	4.87	0.05	0.93	NaNs	NaNs
Distance to Nearest Wetland + Land Cover							
Agriculture						NaNs	NaNs
Forest						NaNs	NaNs
Size	5	54.61	5.98	0.03	0.95		
Small						-0.27	0.60
Medium						-1.20	0.99
Large						-0.37	1.07

APPENDIX C. DETECTION AND OCCUPANCY CONFIDENCE SETS FOR CO-OCCURANCE

Table 1. Blanchard's Cricket Frog and Gray Treefrog call detection confidence set for June 2012. Distance to nearest wetland, ephemeral, adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	6	41.19	0	0.56	0.56	-0.39	1.13
Moonshine	7	44.29	3.11	0.12	0.68	2.86	2.47
Air Temperature	7	44.74	3.55	0.10	0.78	0.44	0.49
Wind	7	44.96	3.78	0.09	0.86	-1.12	1.24
Water Temperature	7	45.6	4.42	0.06	0.93	-0.23	0.49
Time	7	45.78	4.59	0.06	0.98	-0.19	1.11

Table 2. Blanchard's Cricket Frog and Gray Treefrog call detection confidence set for May 2013. The models for time and size were taken out because they would not converge. Also, adjacent vegetation, size, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Moonshine	7	53.87	0	0.53	0.53	-1.48	1.56
Null	6	54.8	0.93	0.33	0.86	0.76	0.71
Air Temperature	7	58.81	4.94	0.04	0.90	0.19	0.20
Water Temperature	7	59.26	5.39	0.04	0.94	0.21	0.29
Wind	7	59.52	5.64	0.03	0.97	0.26	0.71

Table 3. Blanchard's Cricket Frog and Gray Treefrog call detection confidence set for June 2013. Distance to nearest wetland, ephemeral, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Air Temperature	7	114.26	0	0.38	0.38	0.10	0.08
Water Temperature	7	115.74	1.48	0.18	0.56	0.05	0.09
Moonshine	7	115.96	1.71	0.16	0.72	0.45	1.76
Time	7	115.99	1.74	0.16	0.88	0.01	0.03
Size	8	118.79	4.53	0.04	0.92		
Small						0.31	0.40
Medium						-0.20	0.56
Large						-0.31	0.62
Null	6	119	4.74	0.04	0.96	0.50	0.60

Table 4. Blanchard's Cricket Frog and Boreal Chorus Frog call detection confidence set for May 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	10	89	0	0.97	0.97	5.37E-18	1.22

Table 5. Blanchard's Cricket Frog and Boreal Chorus Frog call detection confidence set for June 2013. Adjacent vegetation, aquatic vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Time	8	55.31	0	0.85	0.85	0.25	0.27
Null	7	60.12	4.8	0.08	0.92	-0.63	0.40
Wind	8	62.19	6.88	0.03	0.95	-1.63	1.57

Table 6. Woodhouse's Toad and Plains Leopard Frog call detection confidence set for May 2013. Adjacent vegetation, aquatic vegetation, size, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	5	53.57	0	0.45	0.45	0.14	0.63
Moonshine	6	54.91	1.34	0.23	0.68	7.60	4.67
Water Temperature	6	57.04	3.48	0.08	0.76	0.27	0.34
Wind	6	57.1	3.54	0.08	0.84	-0.45	0.65
Time	6	57.22	3.66	0.07	0.91	0.19	0.44
Air Temperature	6	57.6	4.03	0.06	0.97	0.08	0.24

Table 7. Woodhouse's Toad and Plains Leopard Frog call detection confidence set for June 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Water Temp + Air Temp + Wind + Moonshine	14	57.36	0	0.96	0.96	3.70	1.73
Water Temp + Air Temp + Wind + Moonshine						-0.97	0.53
Water Temp + Air Temp + Wind + Moonshine						1.73	2.96
Water Temp + Air Temp + Wind + Moonshine						-33.88	14.44

Table 8. Gray Treefrog and Boreal Chorus Frog call detection confidence set for May 2013. Adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	55.49	0	0.48	0.48	0.60	0.58
Water Temperature	9	55.81	0.32	0.41	0.90	0.85	0.47
Air Temperature	9	61.11	5.61	0.03	0.93	0.43	0.31
Time	9	61.13	5.63	0.03	0.96	0.43	1.02

Table 9. Gray Treefrog and Boreal Chorus Frog call detection confidence set for June 2013. Adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Moonshine	9	84.45	0	0.56	0.56	-8.72	4.87
Time	9	87.05	2.59	0.15	0.72	0.09	0.05
Null	8	87.41	2.96	0.13	0.84	NaNs	NaNs
Wind	9	89.38	4.93	0.05	0.89	-0.94	0.94
Water Temperature	9	89.48	5.03	0.05	0.94	-0.15	0.14
Air Temperature	9	90.36	5.91	0.03	0.97	-0.07	0.14

Table 10. Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call detection confidence set for May 2013. Adjacent vegetation, aquatic, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	9	73.62	0	0.91	0.91	0.83	0.68
Time	10	80.86	7.24	0.02	0.93	1.02	1.23
Air Temperature	10	81.19	7.58	0.02	0.95	0.22	0.22

Table 11. Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call detection confidence set for June 2013. Adjacent vegetation, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Air Temperature	9	87.29	0	0.30	0.30	0.18	0.10
Null	8	87.3	0	0.30	0.59	-0.02	0.30
Wind	9	87.84	0.55	0.23	0.82	1.10	0.82
Size	10	89.33	2.04	0.11	0.93		
Small						0.51	0.52
Medium						-0.71	0.69
Large						-0.92	0.83
Water Temperature	9	91.24	3.95	0.04	0.97	0.09	0.11

Table 12. Boreal Chorus Frog, Gray Treefrog, and Plains Leopard Frog call detection confidence set for May 2013. Adjacent vegetation, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	63.37	0	0.57	0.57	0.37	0.97
Moonshine	9	66.17	2.81	0.14	0.71	-0.79	1.54
Wind	9	66.19	2.82	0.14	0.85	0.31	0.54
Water Temperature	9	67.58	4.21	0.07	0.92	0.51	0.41
Air Temperature	9	68.18	4.81	0.05	0.97	0.29	0.24

Table 13. Boreal Chorus Frog, Gray Treefrog, and Plains Leopard Frog call detection confidence set for May 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Time	11	67.95	0	0.58	0.58	0.81	0.54
Moonshine	11	69.64	1.68	0.25	0.82	-7.42	4.77
Null	10	71.3	3.35	0.11	0.93	-1.32	0.38
Air Temperature	11	74.03	6.08	0.03	0.96	0.07	0.13

Table 14. Woodhouse's Toad, Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call detection confidence set for June 2013. Adjacent vegetation, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	8	60.79	0	0.72	0.72	NaNs	NaNs
Water Temperature	9	64.87	4.08	0.09	0.81	0.15	0.23
Time	9	65.81	5.02	0.06	0.87	0.11	0.30
Moonshine	9	65.87	5.08	0.06	0.93	-1.41	1.88
Air Temperature	9	66.5	5.71	0.04	0.97	0.12	0.17

Table 15. Blanchard's Cricket Frog and Gray Treefrog call occupancy confidence set for June 2012. Bend, size, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Ephemeral	3	28.26	0	0.72	0.72	8.75	32.27
Adjacent Vegetation	4	33.72	5.47	0.05	0.77		
Bare						8.40	41.40
Herb/Forb						9.91	41.40
Combination						-9.19	41.40
Null	2	33.77	5.52	0.05	0.82	-0.86	0.98
Distance to Nearest Wetland	3	33.93	5.67	0.04	0.86	0.08	0.12
Aquatic Vegetation	3	34.06	5.81	0.04	0.90	8.93	60.80
Bend	4	34.46	6.20	0.03	0.93		
Hamburg						0.71	2.05
Kansas						-10.49	63.08
Langdon						-1.95	1.99
Aquatic Vegetation + Adjacent Vegetation	5	34.8	6.54	0.03	0.96	9.13	50.30
Aquatic Vegetation + Adjacent Vegetation							
Bare						9.26	46.50
Herb/Forb						10.45	46.60
Combination						-18.31	68.50

Table 16. Blanchard's Cricket Frog and Gray Treefrog call occupancy confidence set for May 2013. Size, adjacent vegetation, and bend were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	50.39	0	0.28	0.28	8.73	43.30
Distance to Nearest Wetland	4	50.67	0.28	0.24	0.52	-0.002	0.001
Null	3	50.8	0.41	0.23	0.75	-0.17	0.56
Land Cover	5	52.79	2.40	0.08	0.83		
Agriculture						1.16	NaN
Forest						-1.26	NaN
Ephemeral	4	53.01	2.62	0.08	0.91	1.45	1.70
Slope	4	53.75	3.36	0.05	0.96	1.39	3.17

Table 17. Blanchard's Cricket Frog and Gray Treefrog call occupancy confidence set for June 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also, size, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	113.24	0	0.96	0.96	5.754	19.013

Table 18. Blanchard's Cricket Frog and Boreal Chorus Frog call detection confidence set for May 2013.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Land Cover	4	43.54	0	0.81	0.81		
Agriculture						2.02	NaN
Forest						-2.10	NaN
Null	2	48.26	4.72	0.08	0.89	-0.20	0.54
Size	4	50.76	7.22	0.02	0.91		
Small						-8.26	38.10
Medium						8.26	38.10
Large						8.51	38.10
Slope	3	50.91	7.37	0.02	0.93	1.06	2.83
Ephemeral	3	50.99	7.45	0.02	0.95	-0.36	1.41

Table 19. Blanchard's Cricket Frog and Boreal Chorus Frog call detection confidence set for June 2013. Agriculture and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	4	55.65	0	0.83	0.83	-0.009	0.005
Aquatic Vegetation	4	59.31	3.66	0.13	0.96	7.91	18.30

Table 20. Woodhouse's Toad and Plains Leopard Frog call occupancy confidence set for May 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also, adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Bend	4	46.72	0	0.33	0.33		
Hamburg						1.16	1.77
Kansas						-10.80	62.29
Langdon						5.56	63.90
Ephemeral	3	47.36	0.64	0.24	0.57	9.29	46.71
Aquatic Vegetation	3	48.33	1.61	0.15	0.72	8.70	33.50
Slope	3	48.82	2.10	0.12	0.84	-35.70	49.70
Null	2	48.94	2.22	0.11	0.95	0.51	1.22

Table 21. Woodhouse's Toad and Plains Leopard Frog call occupancy confidence set for June 2013. The model for distance to nearest wetland plus land cover was taken out because it would not converge. Also agriculture and forest were removed from the global occupancy model because the models would not converge with these covariates included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Null	6	48.03	0	0.50	0.50	-0.85	0.44
Distance to Nearest Wetland	7	49.37	1.34	0.26	0.76	0.002	0.001
Slope	7	50.38	2.34	0.16	0.92	-7.89	5.77
Bend	8	52.19	4.15	0.06	0.98		
Hamburg						17.80	NaN
Kansas						-19.50	NaN
Langdon						-10.30	NaN

Table 22. Gray Treefrog and Boreal Chorus Frog call occupancy confidence set for May 2013. Size, bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to the Nearest Wetland	3	43.84	0	0.30	0.30	-0.003	0.001
Aquatic Vegetation	3	44.26	0.42	0.24	0.55	8.14	39.30
Null	2	44.32	0.49	0.24	0.78	-0.50	0.51
Slope	3	47.06	3.22	0.06	0.84	0.62	2.60
Ephemeral	3	47.11	3.28	0.06	0.90	-0.09	1.38
Land Cover	4	47.18	3.35	0.06	0.96		
Agriculture						0.74	NaN
Forest						-1.08	NaN

Table 23. Gray Treefrog and Boreal Chorus Frog call occupancy confidence set for June 2013.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	4	76.11	0	0.32	0.32	1.95	1.17
Null	3	77.45	1.35	0.16	0.48	-0.59	0.47
Ephemeral	4	77.61	1.51	0.15	0.63	1.33	0.96
Slope	4	78.33	2.22	0.11	0.74	-4.76	4.04
Distance to Nearest Wetland	4	78.91	2.80	0.08	0.82	-0.001	0.001
Aquatic Vegetation + Adjacent Vegetation	8	79.04	2.93	0.07	0.89	9.17	39.92
Aquatic Vegetation + Adjacent Vegetation							
Bare						9.11	39.93
Grass						0.11	1.42
Herb/Forb						0.66	1.03
Shrub/Tree						10.06	70.28
Combination						-10.03	39.93
Land Cover	5	80.47	4.37	0.04	0.93		
Agriculture						-0.82	13699
Forest						0.32	13699
Bend	5	81.5	5.40	0.02	0.95		
Hamburg						-0.64	0.82
Kansas						0.31	0.93
Langdon						-0.63	1.13

Table 24. Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call occupancy confidence set for May 2013. Bend, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Aquatic Vegetation	3	47.64	0	0.26	0.26	8.44	39.30
Distance to Nearest Wetland	3	47.74	0.10	0.24	0.50	-0.002	0.001
Null	2	48.26	0.62	0.19	0.69	-0.20	0.54
Land Cover	4	49.19	1.55	0.12	0.81		
Agriculture						1.19	19373
Forest						-1.27	19373
Ephemeral	3	50.03	2.38	0.08	0.89	1.52	1.74
Slope	3	50.86	3.21	0.05	0.94	1.27	2.98
Distance to Nearest Wetland + Land Cover	5	51.99	4.34	0.03	0.97	-0.001	0.002
Distance to Nearest Wetland + Land Cover							
Agriculture						1.26	NaN
Forest						-0.60	NaN

Table 25. Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call occupancy confidence set for June 2013. Adjacent vegetation, aquatic vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	100.06	0	0.39	0.39	-11.24	5.93
Aquatic Vegetation	4	100.27	0.21	0.35	0.74	2.81	2.04
Null	3	103.19	3.14	0.08	0.82	0.63	0.84
Distance to Nearest Wetland	4	103.44	3.38	0.07	0.89	-0.002	0.001
Bend	5	104.82	4.76	0.04	0.93		
Hamburg						2.79	5.06
Kansas						-2.93	4.84
Langdon						-2.67	4.89
Ephemeral	4	105.56	5.51	0.02	0.95	0.31	1.26

Table 26. Boreal Chorus Frog, Gray Treefrog, and Plains Leopard Frog call occupancy confidence set for May 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Distance to Nearest Wetland	3	45.22	0	0.30	0.30	-0.003	0.002
Aquatic Vegetation	3	45.64	0.42	0.24	0.54	8.75	51.00
Null	2	45.71	0.49	0.24	0.78	-0.34	0.60
Slope	3	48.44	3.22	0.06	0.84	0.73	2.95
Ephemeral	3	48.50	3.28	0.06	0.90	-0.10	1.47
Land Cover	4	48.57	3.34	0.06	0.96		
Agriculture						0.88	NaN
Forest						-1.08	NaN

Table 27. Boreal Chorus Frog, Gray Treefrog, and Plains Leopard Frog call occupancy confidence set for June 2013. Adjacent vegetation, agriculture, and forest were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	4	55.34	0	0.34	0.34	-54.00	66.00
Null	3	56.36	1.01	0.20	0.54	1.18	3.65
Aquatic Vegetation	4	56.84	1.50	0.16	0.70	5.65	34.28
Distance to Nearest Wetland + Land Cover	6	57.9	2.55	0.09	0.79	-0.02	NaN
Distance to Nearest Wetland + Land Cover							
Agriculture						-11.55	23727
Forest						27.75	23727
Ephemeral	4	58.15	2.81	0.08	0.87	5.08	39.17
Distance to Nearest Wetland	4	58.91	3.57	0.06	0.93	5.44E-05	0.002
Bend	5	59.8	4.46	0.04	0.96		
Hamburg						5.61	33.20
Kansas						-5.76	33.10
Langdon						-5.66	33.10

Table 28. Woodhouse's Toad, Blanchard's Cricket Frog, Gray Treefrog, and Plains Leopard Frog call occupancy confidence set for May 2013. Adjacent vegetation and bend were removed from the global occupancy model because the models would not converge with these covariate included.

Model	K	AICc	Delta_AICc	AICcWt	Cum.Wt	Estimate	±SE
Slope	3	39.13	0	0.45	0.45	-63.30	64.00
Null	2	41.48	2.34	0.14	0.59	0.46	1.90
Bend	4	41.91	2.78	0.11	0.71		
Hamburg						1.11	2.46
Kansas						-10.36	59.11
Langdon						4.18	60.60
Aquatic Vegetation	3	41.91	2.78	0.11	0.82	8.48	36.20
Ephemeral	3	42.39	3.25	0.09	0.91	7.74	51.16
Distance to Nearest Wetland	3	43.26	4.13	0.06	0.97	-0.001	0.002

APPENDIX D: NODE IMPORTANCE

The wetland in Hamburg Bend most important to connectivity for all dispersal distances and conditions was H18/59. It always had the highest dIIC, but at 500 m it contributed more to dIICintra, at 1500 m it contributed more to dIICflux, and at 3000 m it was the most important wetland for connectivity or dIICconnector (Table 1-6). While this was overall the most important wetland, there were other important wetlands at different conditions and distances. During drought conditions, H65N and H34 were the only wetlands connected when the dispersal distance was 500 and therefore were the only two with dIICflux over 0 (Table 1). For the dispersal distance of 1500 m the wetland H14 had the highest dIICconnector and H59 had both a high dIICconnector value and dIICflux value, which gave it the second highest dIIC (Table 2). During normal conditions, H08 and H62 had both the highest dIICflux and the best dIICconnector values, making the second and third highest dIIC values for the dispersal distance of 500 m (Table 4). At 1500 m, H18/61 had the best dIICintra and dIICflux, but H15S and H59 had the highest dIICconnector values (Table 5).

Table 1. Node importance for Hamburg Bend during drought conditions for the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H09	0.109163	0.1091632	0	0
H12	5.284494	5.284494	0	0
H14	0.224128	0.224127	0	0
H34	0.792566	0.2394287	0.553137	0
H59	4.192641	4.192641	0	0
H18/H61	88.11913	88.11913	0	0
H65N	1.831014	1.277877	0.553137	0

Table 2. Node importance for Hamburg Bend during drought conditions for the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H09	2.034327	0.0687501	1.965577	0
H12	14.15417	3.32813	10.82604	0
H14	14.67937	0.1411538	3.487199	11.05102
H34	3.071626	0.1507902	2.920836	0
H59	29.78744	2.640491	16.26349	10.88346
H18/H61	88.38443	55.4967	30.35471	2.533025
H65N	12.2978	0.8047964	8.920532	2.572475

Table 3. Node importance for Hamburg Bend during drought conditions for the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H09	2.395309	0.064773	2.330536	0
H12	16.66576	3.13559	13.53017	0
H14	4.387	0.132988	4.254012	0
H34	4.065373	0.142067	3.923307	0
H59	18.97421	2.487732	16.48648	0
H18/H61	87.99163	52.28608	32.82683	2.878724
H65N	9.391973	0.758237	8.633736	0

Table 4. Node importance for Hamburg Bend during normal conditions for the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H08	12.70935	5.691553	6.539407	0.4783942
H09	1.24942	0.083139	1.166281	0
H12	3.965455	3.965455	0	0
H15S	1.869291	0.186264	1.683027	0
H34	1.193249	0.179666	1.013583	0
H59	3.146135	3.146135	0	0
H18/H61	66.12411	66.12411	0.0000001	0
H62	11.95165	4.687347	6.515665	0.7486385
H63	2.635151	2.635151	0	0
H65N	3.975011	1.993794	1.981217	0
H65S	2.756688	0.958912	1.797776	0

Table 5. Node importance for Hamburg Bend during normal conditions at the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H08	16.66346	2.706134	13.95732	0
H09	2.013964	0.03953	1.974434	0
H12	18.97058	1.885434	11.17982	5.905319
H15S	25.91579	0.088562	3.188501	22.63872
H34	2.394956	0.085425	2.309532	0
H59	34.27215	1.495877	12.85297	19.92331
H18/H61	70.08194	31.43969	32.83465	5.807597
H62	15.12214	2.228669	12.89347	0
H63	8.695219	1.252922	7.442297	0
H65N	10.34459	0.947979	9.396607	0
H65S	7.174016	0.455929	6.718088	0

Table 6. Node importance for Hamburg Bend during normal conditions at the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
H08	19.36422	2.23526	17.12896	0
H09	1.980307	0.032651	1.947655	0
H12	13.67657	1.557365	12.1192	0
H15S	3.411355	0.073152	3.338202	0
H34	2.896928	0.070561	2.826367	0
H59	14.02011	1.235591	12.78452	0
H18/H61	71.98225	25.96912	38.87594	7.137191
H62	17.57309	1.840875	15.73221	0
H63	10.68028	1.034911	9.645371	0
H65N	9.65039	0.783029	8.867362	0
H65S	6.692588	0.376596	6.315992	0

The wetland most important to connectivity during the drought condition was K24. For all dispersal distances it had the highest dIIC, dIICintra, and dIICflux (Table 7-9). However, at the 500 m dispersal distance the wetlands K23, K68, and K71 were important to connectivity and had the highest dIICconnector value. The wetlands most important to connectivity during normal conditions were K07 and K25. For all dispersal distances they had the highest dIIC, dIICintra, and dIICflux (Table 10-12). At the 500 m dispersal distance the wetlands K30 and K68 had the highest dIICconnector values and were essential stepping stones to the two components containing linked wetlands (Table 10). At the 1500 m dispersal distance the wetlands K56I and K74 had the highest dIICconnector values. In fact, K74 was the only wetland connecting K32 to the rest of the wetland complex (Table 11).

Table 7. Node importance for Kansas Bend during drought conditions at the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K23	46.1816	5.237496	22.96732	17.97679
K24	55.25975	25.68456	29.57519	0
K67S	16.41647	3.451546	12.96492	0
K68	45.40845	7.297599	24.0746	14.03625
K71	30.55282	4.02444	17.29047	9.237919
K74	0.868112	0.868112	0	0

Table 8. Node importance for Kansas Bend during drought conditions at the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K23	27.92428	4.590348	23.33394	0
K24	56.11719	22.51096	33.60623	0
K67S	19.91802	3.025071	16.89295	0
K68	32.96175	6.395904	26.56584	0
K71	21.5076	3.527178	17.98043	0
K74	0.760848	0.760848	0	0

Table 9. Node importance for Kansas Bend during drought conditions at the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K23	25.74808	3.967112	21.78097	0
K24	55.82673	19.45463	36.3721	0
K67S	20.9021	2.614355	18.28775	0
K68	30.39297	5.527527	24.86544	0
K71	22.57022	3.04829	19.52193	0
K74	9.290436	0.657547	8.632889	0

Table 10. Node importance for Kansas Bend during normal conditions at the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K02	10.65102	1.347524	9.303493	0
K06	4.671569	0.149418	4.522151	0
K07	54.99628	20.06928	34.92699	0
K10	7.519318	0.387109	7.13221	0
K23	7.098535	0.287535	6.811	0
K24	13.39256	1.410064	11.9825	0
K25	31.21581	8.324502	22.89131	0
K27	6.682661	0.29808	6.384581	0
K30	0.920832	0.549629	0.336848	0.034355
K32	0.180832	0.180832	0	0
K56I	0.116711	0.010802	0.105909	0
K56O	0.148364	0.017455	0.130909	0
K66	1.245083	0.014906	1.230177	0
K67F	7.680476	0.874577	6.8059	0
K67S	3.575027	0.189487	3.38554	0
K68	21.58117	0.400633	7.904965	13.27558
K69	1.769398	0.046417	1.722982	0
K71	4.793566	0.220939	4.572627	0
K74	0.243861	0.047659	0.196202	0

Table 11. Node importance for Kansas Bend during normal conditions at the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K02	13.19335	0.985144	12.20821	0
K06	4.275965	0.109236	4.166729	0
K07	49.55636	14.67219	34.88417	0
K10	7.123018	0.283006	6.840011	0
K23	6.156196	0.21021	5.945986	0
K24	12.86576	1.030865	11.8349	0
K25	32.53046	6.08585	26.4446	0
K27	6.040813	0.217919	5.822895	0
K30	7.111843	0.401821	6.710022	0
K32	2.601493	0.132202	2.469291	0
K56I	1.480051	0.007897	1.155523	0.31663
K56O	1.272128	0.012761	1.259367	0
K66	1.372145	0.010897	1.361248	0
K67F	8.713804	0.639383	8.074421	0
K67S	4.070025	0.13853	3.931496	0
K68	7.381111	0.292894	7.088217	0
K69	2.014389	0.033934	1.980455	0
K71	5.345293	0.161523	5.183769	0
K74	4.15274	0.034842	1.716477	2.401422

Table 12. Node importance for Kansas Bend during normal conditions at the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
K02	12.66968	0.913808	11.75587	0
K06	4.218889	0.101326	4.117563	0
K07	48.89488	13.60976	35.28512	0
K10	6.790688	0.262513	6.528175	0
K23	5.852516	0.194989	5.657527	0
K24	12.90175	0.956219	11.94554	0
K25	31.49028	5.645168	25.84511	0
K27	5.958863	0.202139	5.756724	0
K30	8.162826	0.372725	7.790101	0
K32	3.547445	0.122629	3.424816	0
K56I	1.144336	0.007325	1.13701	0
K56O	1.45469	0.011837	1.442853	0
K66	1.344253	0.010108	1.334145	0
K67F	10.29685	0.593085	9.70377	0
K67S	4.792871	0.128499	4.664372	0
K68	6.969136	0.271685	6.697451	0
K69	2.372149	0.031477	2.340673	0
K71	5.175373	0.149827	5.025546	0
K74	2.345082	0.032319	2.312763	0

The wetland most important to connectivity for all dispersal distances and conditions was L5370. It had the highest dIIC, dIICintra, and dIICflux for all dispersal distances and conditions, except for the dIICflux for the drought condition at the 500 m dispersal distance. This was lower because it was not connected to any of the other wetlands (Table 13-18). During the drought condition L55E had the highest dIICconnector value for the 500 m dispersal distance, acting as a stepping stone for the only component with links (Table 13). For the dispersal distance of 1500 m L43 had a very high dIICconnector value because it connected L7053 to the rest of the complex (Table 14). During the normal condition L43, L54W, and L55E functioned as stepping-stones for the 500 m dispersal distance (Table 15).

Table 13. Node importance for Langdon Bend during drought conditions at the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L43	23.82611	8.202574	15.62354	0
L44	0.027733	2.25E-05	0.02771	0
L55E	7.520807	0.811889	6.68548	0.023438
L4655W	37.85962	20.71077	17.14884	0
L7053	50.53196	50.53195	1.00E-07	0

Table 14. Node importance for Langdon Bend during drought conditions at the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L43	47.90339	5.60935	24.60989	17.68415
L44	0.042345	1.54E-05	0.04233	0
L55E	8.047225	0.555212	7.492013	0
L4655W	40.64396	14.16311	26.48084	0
L7053	66.16315	34.55639	31.60675	0

Table 15. Node importance for Langdon Bend during drought conditions at the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L43	27.7643	5.153659	22.61064	0
L44	0.045964	1.41E-05	0.04595	0
L55E	8.734941	0.510108	8.224834	0
L4655W	44.11739	13.01253	31.10485	0
L7053	68.91198	31.74911	37.16286	0

Table 16. Node importance for Langdon Bend during normal conditions at the 500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L37	0.735408	0.392842	0.342567	0
L39	22.86697	7.42356	15.44341	0
L43	24.56044	5.215049	14.02886	5.316537
L44	0.021142	1.43E-05	0.021128	0
L51ns	0.641292	0.298725	0.342567	0
L54E	2.697388	0.227912	2.469477	0
L54W	12.29278	2.176274	8.351299	1.765205
L55E	5.663192	0.516185	5.128596	0.018412
L4655W	28.5054	13.16754	15.33787	0
L7053	47.57072	32.12731	15.44341	0

Table 17. Node importance for Langdon Bend during normal conditions at the 1500m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L37	6.135044	0.216594	5.91845	0
L39	24.81783	4.092982	20.72485	0
L43	22.8566	2.875319	19.98128	0
L44	0.031192	7.90E-06	0.031185	0
L51ns	4.506854	0.164702	4.342152	0
L54E	4.675271	0.125659	4.549611	0
L54W	14.76519	1.199889	13.5653	0
L55E	5.927816	0.284599	5.643217	0
L4655W	30.72199	7.259925	23.46206	0
L7053	51.62914	17.7134	33.91574	0

Table 18. Node importance for Langdon Bend during normal conditions at the 3000m dispersal distance.

Wetland	dIIC	dIICintra	dIICflux	dIICconnector
L37	5.818632	0.200898	5.617734	0
L39	25.29403	3.796378	21.49766	0
L43	21.20026	2.666955	18.5333	0
L44	0.035097	7.30E-06	0.03509	0
L51ns	5.07397	0.152767	4.921203	0
L54E	4.431956	0.116553	4.315403	0
L54W	13.69521	1.112938	12.58228	0
L55E	6.669825	0.263975	6.40585	0
L4655W	33.68715	6.733824	26.95332	0
L7053	52.6198	16.42977	36.19002	0

APPENDIX E: TURTLE TRAPPING DATA

Table 1. The number of each sex and and life stage captured in turtle traps every year for each species. The traps were set in three different wetland complexes along the Missouri River in SE Nebraska.

Species	Year	Females	Males	Adults	Juveniles
Western Painted Turtle	2011	20	6	20	6
Common Snapping Turtle	2011	NA	NA	14	4
False Map Turtle	2011	5 (2 NA)	3	5	5
Spiny Softshell Turtle	2011	1	0	1	0
Red-eared Slider	2011	1	0	1	0
Western Painted Turtle	2012	25	45	32	38
Common Snapping Turtle	2012	9 (4 NA)	8	11	10
False Map Turtle	2012	0	0	0	0
Spiny Softshell Turtle	2012	0	1	1	0
Red-eared Slider	2012	0	2	2	0
Western Painted Turtle	2013	25	31	51	5
Common Snapping Turtle	2013	NA	NA	44	12
False Map Turtle	2013	0	4	3	1
Spiny Softshell Turtle	2013	0	0	0	0
Red-eared Slider	2013	1	4	5	0

Table 2. Turtle trapping results from three different wetland complexes along the Missouri River in SE Nebraska. A total of 18 turtles were captured in 2010, 56 in 2011, 93 in 2012, and 146 in 2013.

Species	Year	Bend	Captured	Recaptured
Western Painted Turtle	2010	Hamburg	0	0
Western Painted Turtle	2010	Kansas	10	0
Western Painted Turtle	2010	Langdon	3	0
Common Snapping Turtle	2010	Hamburg	2	0
Common Snapping Turtle	2010	Kansas	2	0
Common Snapping Turtle	2010	Langdon	1	0
Western Painted Turtle	2011	Hamburg	1	0
Western Painted Turtle	2011	Kansas	14	1
Western Painted Turtle	2011	Langdon	11	0
Common Snapping Turtle	2011	Hamburg	7	0
Common Snapping Turtle	2011	Kansas	5	0
Common Snapping Turtle	2011	Langdon	6	1
False Map Turtle	2011	Hamburg	0	0
False Map Turtle	2011	Kansas	10	0
False Map Turtle	2011	Langdon	0	0
Spiny Softshell Turtle	2011	Hamburg	0	0
Spiny Softshell Turtle	2011	Kansas	1	0
Spiny Softshell Turtle	2011	Langdon	0	0
Red-eared Slider	2011	Hamburg	0	0
Red-eared Slider	2011	Kansas	1	0
Red-eared Slider	2011	Langdon	0	0
Western Painted Turtle	2012	Hamburg	39	14
Western Painted Turtle	2012	Kansas	8	4
Western Painted Turtle	2012	Langdon	23	0
Common Snapping Turtle	2012	Hamburg	9	0
Common Snapping Turtle	2012	Kansas	4	1
Common Snapping Turtle	2012	Langdon	8	0
Spiny Softshell Turtle	2012	Hamburg	1	0
Spiny Softshell Turtle	2012	Kansas	0	0
Spiny Softshell Turtle	2012	Langdon	0	0
Red-eared Slider	2012	Hamburg	0	0
Red-eared Slider	2012	Kansas	0	0
Red-eared Slider	2012	Langdon	2	0
Western Painted Turtle	2013	Hamburg	10	3
Western Painted Turtle	2013	Kansas	25	18
Western Painted Turtle	2013	Langdon	21	0
Common Snapping Turtle	2013	Hamburg	11	0
Common Snapping Turtle	2013	Kansas	31	3
Common Snapping Turtle	2013	Langdon	14	0
False Map Turtle	2013	Hamburg	2	0
False Map Turtle	2013	Kansas	1	0
False Map Turtle	2013	Langdon	1	0
Red-eared Slider	2013	Hamburg	0	0
Red-eared Slider	2013	Kansas	0	0
Red-eared Slider	2013	Langdon	5	1