SITE-SPECIFIC HABITAT AND LANDSCAPE ASSOCIATIONS OF RUSTY BLACKBIRDS WINTERING IN LOUISIANA

A Thesis

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Sinéad Mary Borchert
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ABSTRACT

The Rusty Blackbird (*Euphagus carolinus*) has gained notoriety in recent years as one of the fastest declining North American bird species, with a global population loss of as much as 95%. Causes of the decline are not completely understood, but the high rate of forested wetland change in the southeastern United States suggests that wintering habitat degradation may be a primary driver. To better inform management on critical wintering grounds, I surveyed 68 sites in Louisiana where Rusty Blackbirds had been known to occur to address how occupancy changes with habitat type and colonization and extinction rates vary with ground cover, rainfall, and invertebrate biomass.

Rusty Blackbirds use a large area while foraging on the wintering ground, therefore management may need to be targeted to even larger spatial scales. I assessed the relationship between statewide Rusty Blackbird abundance data from the Louisiana Winter Bird Atlas and landscape scale habitat within 512 unique USGS 7.5-minute quadrangles using datasets on land cover, cropland cover, and soil type.

Results indicate that forested wetlands are important habitats associated with Rusty Blackbird presence, but only under certain conditions. Rusty Blackbirds prefer shallow water for foraging. At my sites, deep water cover increased with the cover of forested wetlands and may have deterred Rusty Blackbirds from using primarily forested wetland sites. The most important variables associated with transience were wet leaf litter and invertebrate biomass, which were both positively associated with colonization and negatively associated with extinction probability.

For the Louisiana Winter Bird Atlas data, the top model included all explanatory variables for Rusty Blackbird abundance. Abundance increased with cover of soil hydrologic

groups C, C/D, and D, which are capable of retaining surface water, suggesting that at larger scales water cover is more important than any particular habitat type. Pecans are an important food source for wintering Rusty Blackbirds and pecan orchards had the strongest positive relationship with abundance. In addition to maintaining pecan groves on the landscape, Rusty Blackbirds may benefit from management for shallowly flooded forested wetlands that can support high amounts of wet leaf litter on the ground's surface and abundant invertebrates.

CHAPTER 1. GENERAL INTRODUCTION

1.1 The Decline

Rusty Blackbirds (*Euphagus carolinus*) were once ubiquitous throughout the southeastern United States. Historical accounts often described flocks of hundreds of birds, but over the last two centuries the species has become progressively less common, prompting concerns about population health. The decline has been staggering; estimates of population decline range from 85 – 95% since the 1960s (Greenberg and Droege 1999, Greenberg and Matsuoka 2010, Greenberg et al. 2011). The last estimates of population size vary widely and range from 2 million to as few as 158,000 Rusty Blackbirds remaining (Rich et al. 2004). Data suggest that the decline continues (Figure 1.1) at a rate of at least 5% per year (Niven et al. 2004).

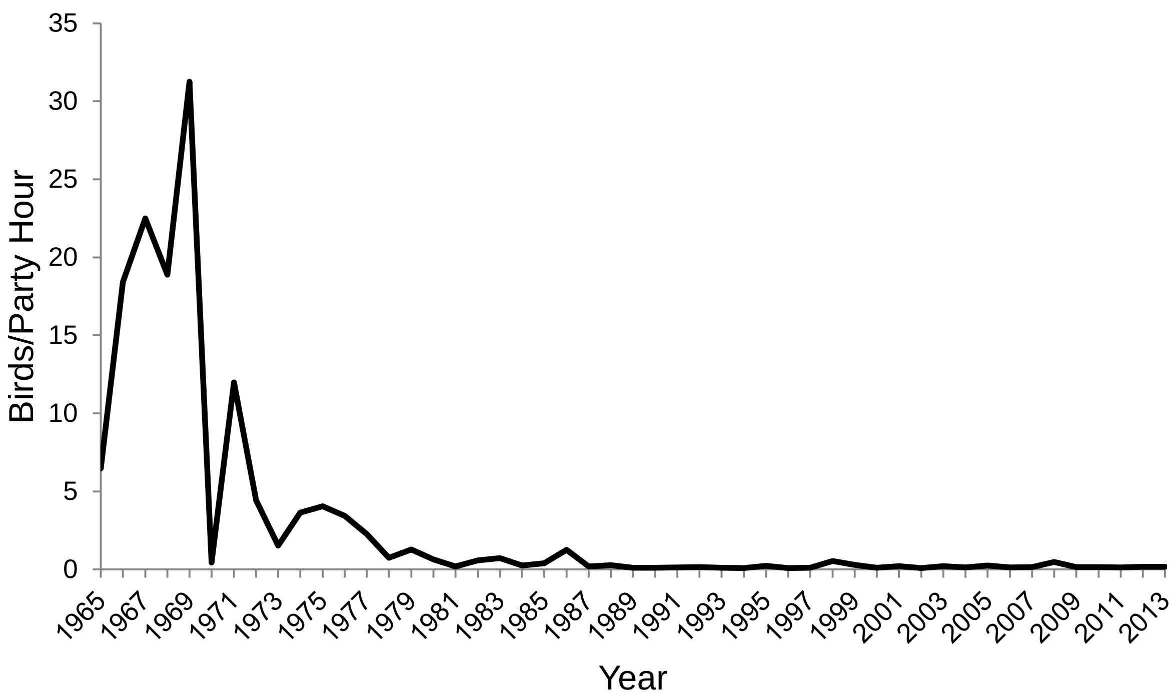

FIGURE 1.1 Rusty Blackbird decline trend based on Christmas Bird Count data (National Audubon Society 2010). I omitted years prior to 1965 because the relative number of sampling circles was significantly lower and effort was not consistently reported (Niven et al. 2004).

Multiple hypotheses could explain the Rusty Blackbird decline, including: habitat loss and degradation across its range (Greenberg and Droege 1999, Hamel et al. 2009, Greenberg and Matsuoka 2010), loss of historic stopover sites, bioaccumulation of methyl-mercury from wetland acidification (Greenberg and Droege 1999, Edmonds et al. 2010), blackbird control programs (Greenberg and Droege 1999), parasite susceptibility due to stress (Barnard et al. 2010), competition with other blackbirds (Avery 1995), and range retractions in the breeding range due to climate change (Powell 2008, McClure et al. 2012). With such a multitude of factors, it can be difficult to determine whether any one factor is more worthy of blame.

Although there is likely an interactive effect of these factors on the global Rusty Blackbird population, evidence suggests that wintering ground issues may be of greatest importance. Rusty Blackbirds are associated with forested wetland and bottomland hardwood forest systems throughout their range (Avery 1995). They breed in the boreal forests of Canada, Alaska, and the northeastern United States and winter in the southeastern United States, with the highest concentrations occurring in the Mississippi Alluvial Valley and the Coastal Plain of the Carolinas and Georgia (Avery 1995, Hamel and Ozdenerol 2009). If wintering ground conditions are driving the Rusty Blackbird decline, then we may also see declines in other blackbirds that share the same range and have partial niche overlaps. Red-winged Blackbirds (*Agelaius phoeniceus*) and Common Grackles (*Quiscalus quiscula*) are also experiencing rapid declines that are much steeper in southeastern states relative to their overall decline (-3.65%/year vs. -0.9%/year for RWBL and -3.93%/year vs. -1.6%/year for COGR in Mississippi, Alabama, Georgia, and South Carolina), providing support for the idea that we must address wintering ground changes (Newell 2013).

In contrast, there is less evidence suggesting that population limitations are related to migratory stopover and breeding ground habitats. Degradation of historic migratory stopover habitat could be a major cause, but few data are available to assess survival during the migratory phases of the annual cycle (Johnson et al. 2012). Studies on the breeding ground have shown that nest success is comparable to other passerines, thus poor reproductive success is likely not playing a major role (Matsuoka et al. 2010, Buckley 2013, Newell 2013). Additionally, habitat degradation in the breeding range has occurred at a relatively slower rate compared to the wintering range (Greenberg and Droege 1999).

1.2 Rusty Blackbird Natural History

Although closely related to generalist blackbirds, the Rusty Blackbird is thought to be more specialized in its habitat use (Greenberg and Droege 1999, Lanyon and Omland 1999). Recent research suggests that the species is able to use anthropogenically altered areas and may not be as specialized as previously thought. Shallow water, wet leaf litter, and grass are the best indicators for occupancy at a site and wet leaf litter is especially important for larger flocks (DeLeon 2012). Pecan groves and agricultural fields are also utilized, especially if they are adjacent to wetlands (Luscier et al. 2010, Newell 2013). Rusty Blackbirds will use greentree reservoirs in bottomland hardwood forest, provided that the water levels are lowered to expose more foraging habitat (Luscier 2009).

During the winter Rusty Blackbirds forage for invertebrates, seeds, and acorns by flipping over wet leaves and probing shallow water at the edges of ponds and marshes (Avery 1995, Luscier et al. 2010). They also occasionally probe mud and rotting woody debris (Avery 1995). Rusty Blackbirds will utilize domestic pecan (*Carya illinoinensis*) nuts and the mast of small-seeded oaks, such as water (*Quercus nigra*) and willow (*Quercus phellos*) oak, as readily available food resources (Newell 2013). They lack the bill musculature to crack large seeds open however, and must rely on other animals to break them into smaller pieces, such as the Common Grackles they sometimes associate with (Luscier 2009, Newell 2013, S. Borchert personal observation). Rusty Blackbirds can also forage on acorns crushed on roads by cars (S. Borchert, personal observation).

As a wetland associated species, aquatic macroinvertebrates make up the greatest proportion of winter Rusty Blackbird diets (Newell 2013). These invertebrates are often associated with wet leaf litter that provides habitat, an attachment substrate, and food as it decomposes (Fredrickson and Batema 1992, Cummins and Merritt 2008). On the wintering ground Rusty Blackbirds have been noted to consume invertebrates such as dragonfly (Odonata) larvae, fly (Diptera) larvae, small aquatic worms, snails, spiders, and large terrestrial worms (DeLeon 2012, Newell 2013). Following precipitation events, Rusty Blackbirds frequently forage on Oligochaete worms as they emerge from the ground (Newell 2013, S. Borchert, personal observation), presumably to avoid unfavorable soil or to use the moisture for migration across the surface (Ellis et al. 2010).

1.3 Habitat Loss, Degradation, and Hydrologic Alteration in the Southeast

<u>Habitat Loss</u>

The best explanation for the decline may be the loss and degradation of the forested wetland and bottomland hardwood forest systems that the species relies on while wintering (Greenberg and Droege 1999). The Rusty Blackbird decline began as early as the 1800s, at which point there was little anthropogenic development of boreal forests within the breeding range, but significant landscape change in the wintering range (Greenberg and Droege 1999). During this time period, Rusty Blackbirds went from being described as common to uncommon in regional checklists (Greenberg and Droege 1999). Wetland conversion follows a similar trajectory: from the 1780s to the 1880s, all wetland types decreased by 49% across the southeastern United States (Dahl 1990). As much as 75 - 80% of bottomland hardwood forest, which once represented the most extensive wetland type in the United States, has been converted to agriculture (Hefner and Brown 1984). The loss of bottomland hardwood forest is especially high in the Mississippi Alluvial Valley (MAV), where only 24% of the floodplain is still forested and the existing forest cover is highly fragmented (Twedt and Loesch 1999). Although the MAV supports high concentrations of Rusty Blackbirds (Hamel and Ozdenerol 2009), it also has the highest rate of Rusty Blackbird loss by region, at an estimated 6.5% loss per year (Niven et al. 2004). As agricultural conversion has slowed, urbanization and silvicultural practices still drive loss of forested wetlands in the southeast; predictions suggest continued losses with increases in human populations (Faulkner 2004, Hamel et al. 2009).

If wintering habitat is connected to the decline, the Rusty Blackbird population may mirror the pattern of forested wetland losses and gains. Hamel et al. (2009) found that a trend of high rates of freshwater wetland loss from 1950s to 1980s corresponded to the higher rates of Rusty Blackbird population loss observed over the same period. Agricultural conservation programs have been largely responsible for restoring wetland acreage (Dahl 2000), which may help explain the slowing of the decline after the 1980s. Particularly effective programs included the Wetland Reserve Program and Conservation Reserve Program, as well as legislation such as the "Swampbuster" provision of the 1985 Food Security Act (Dahl 2006, King et al. 2006). Additionally, the primary cause of wetland loss in the 1980s switched from agriculture to silvicultural practices (tending, harvest, and replanting of trees), which does not necessarily destroy wetland function (Hamel et al. 2009). The slowing, and perhaps increase, in forested wetlands is visible in the leveling off of the decline after the 1980s (DeLeon 2012; Figure 1.1). However, while there was a 1.1% increase in forested wetlands from 1998 to 2004 (Dahl 2006), they again decreased by 1.2% from 2004 to 2009 (Dahl 2011), suggesting that habitat loss is still a cause for concern and will continue to affect the population in the near future.

Hydrologic Alterations

Much of extant bottomland hardwood systems have degraded hydrologic function, which likely affects Rusty Blackbirds because of their shallow water foraging habits. Widespread hydrologic alterations in the Mississippi Alluvial Valley, such as levee construction, channelization, and other flood control measures, increased after 1927 following a major flood (King et al. 2006). Historically, rivers were allowed to meander and form new wetlands, while also filling in existing wetlands through sedimentation (King et al. 2006). With flood control measures this process has been all but eliminated (King et al. 2006) and the timing, duration, and amount of flooding has changed. Forested wetlands have been lost to agriculture and urban development, but due to hydrologic changes, they are also not being created at their historic rates through natural hydrogeomorphic processes. Channelization, in particular, tends to dry upstream floodplain forests, while downstream in the watershed it causes more frequent flooding of shorter durations (Shankman 1997). River levees impede overbank flooding, leading to drier bottomland hardwood forests.

With flood control measures causing such drastic changes, one would expect plant and animal communities to change over time, which could negatively affect Rusty Blackbirds. Gee et al. (2014) studied the changing bottomland hardwood forest community within a ring levee at Richard K. Yancey Wildlife Management Area, another location where I have study sites. The floodplain at Yancey WMA is completely disconnected from the river and the only water inputs are from precipitation and seepage. With lack of freshwater inputs, the forest community is transitioning from flood-tolerant overcup oak (*Quercus lyrata*) to drier association species (Gee et al. 2014). Invertebrate communities, as well as tree communities, can change in response to flood control. These hydrologic disconnections can decrease macroinvertebrate diversity and densities by removing the link between aquatic and riparian habitats (Kennedy and Turner 2011). For Rusty Blackbirds, which forage on mast from flood tolerant oaks and on aquatic macroinvertebrates, these results are particularly troublesome.

In coastal baldcypress-tupelo swamp forests, another type of forested wetland system, lack of riverine input is contributing to conversion of habitat to open water and marsh (Shaffer et al. 2003, Shaffer et al. 2009). The lack of sediment-depositing overbank flooding magnifies natural rates of land subsidence by preventing land accretion. Subsiding wetlands lead to deepening water, which is mostly unusable for Rusty Blackbird foraging, and also stresses forested wetland trees (Shaffer et al. 2003). At Maurepas Swamp in Louisiana, one of my study areas, lack of nutrient inputs, stagnant water, land subsidence, sea-level rise, and saltwater intrusion are leading to the swamp's deterioration and tree die-offs in some areas (Chambers et al. 2005, Shaffer et al. 2009).

1.4 Objectives

Rusty Blackbirds in Louisiana have lost much of their historic forested wetland wintering habitat to agriculture and development. Coupled with these great land losses are additional hydrologic alterations that have led to three undesirable scenarios for Rusty Blackbirds: not enough water, too much deep water, and increasingly saline water. Levee construction has vastly reduced overbank flooding, which brings fresh water, sediments, and nutrients into forested wetlands, potentially affecting the trees, invertebrates, and shallow water that Rusty Blackbirds rely on. This can contribute to drier bottomland hardwood forests, as well as deeper water in coastal baldcypress-tupelo forests. Climate change contributes to sea-level rise, leading to salt water intrusion and deepening water in coastal forested wetlands. Oil-field canals have also contributed to subsidence and facilitated the movement of high salinity water into freshwater forested wetlands, which is not being balanced by freshwater inputs.

Besides species such as the American Woodcock (*Scolopax minor*), which uses successional forested wetlands during the winter, or the resident Wood Duck (*Aix sponsa*), no other bird species is quite as emblematic of winter season forested wetlands as the Rusty Blackbird. The species' decline could be reflecting the vegetative and hydrological changes to forested wetland ecosystems. Given that there are multiple detrimental factors on the wintering ground that could be affecting Rusty Blackbirds, it is important to study their winter habitat. At local scales, previous studies found that wet bottomland hardwood forest and shallow water, wet leaf litter, and grass cover were positively associated with Rusty Blackbird presence (Luscier 2009, Luscier et al. 2010, DeLeon 2012). However, these studies addressed habitat at smaller scales of 11.3 – 100 m around sites. Rusty Blackbirds can use much larger areas for foraging (average home range 5.08 km^2; Newell (2013), unpublished data analyzed by Borchert). With this in mind, I took a multi-scale approach to determining Rusty Blackbird habitat associations; specifically, I was interested in how local conditions increased or decreased the suitability of general habitats. In this thesis, my five objectives were: 1) determine how transience (colonization and extinction) at the site-scale (100 m) changes with ground cover variables, invertebrate biomass, and rainfall; 2) determine how Rusty Blackbird site occupancy changes with field-estimated (100 m) habitat cover and landscape-scale habitat cover (600 m); 3) determine whether site-level invertebrate biomass is correlated with ground cover and habitat type; 4) determine landscape-scale (~160 km^2) Rusty Blackbird habitat associations over the state of Louisiana using spatial land cover, crop land cover, and soils datasets; and 5) determine whether Rusty Blackbird counts are correlated with variation in annual winter rainfall.

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CHAPTER 2. USING DYNAMIC OCCUPANCY ESTIMATION TO MODEL RUSTY BLACKBIRD SITE-SPECIFIC AND LANDSCAPE HABITAT ASSOCIATIONS

2.1 Introduction

Winter habitat can limit population size in many long-distance migratory birds (reviewed in: Rappole and McDonald 1994, Sherry and Holmes 1996, Keller and Yahner 2006). Degraded winter habitat can reduce food resources, affect fat stores used during migration, decrease cover from predators, and reduce overall survival (Sherry and Holmes 1996). Winter habitat quality can also carry over to affect reproductive success during the breeding season; Norris et al. (2004) found that female American Redstarts using higher quality wintering habitat produced more than two additional young compared to females in poor quality winter territories. For effective conservation of migratory birds we must closely evaluate areas, such as the wintering ground, that have a significant effect on their survival and reproductive success.

Habitat loss and degradation on the wintering ground may have played a major role in the decline of the Rusty Blackbird (*Euphagus carolinus*), a temperate migrant, whose global population has decreased by as much as 93% since the 1960s (Greenberg and Droege 1999, Niven et al. 2004, Greenberg and Matsuoka 2010). Rusty Blackbirds historically relied on forested wetland and bottomland hardwood forest systems throughout their range (Avery 1995). The Mississippi Alluvial Valley, some of which is in Louisiana, supports the highest concentration of wintering Rusty Blackbirds (Niven et al. 2004, Hamel and Ozdenerol 2009). Only 24% of the Mississippi Alluvial Valley's original forest cover remains and what is left is highly fragmented (Twedt and Loesch 1999). Urban development, rather than agriculture, now drives forested wetland loss in the southeast and is expected to increase along with human populations (Faulkner 2004). In contrast to the wintering range, habitat degradation in the boreal breeding range has occurred at a slower rate, and nest success is relatively high (Matsuoka et al. 2010, Powell et al. 2010, Buckley 2013), indicating a need to focus research efforts on the Rusty Blackbird's wintering ecology (Greenberg and Droege 1999, Greenberg and Matsuoka 2010). Although many factors likely contribute to Rusty Blackbird population loss, dramatic land cover change in the wintering range parallels the population decline (Greenberg and Droege 1999). Additionally, flood control measures such as levee construction, channelization, and damming have significantly altered historic hydrological regimes and disconnected floodplains from rivers (Shankman 1997, Day et al. 2000, Shaffer et al. 2009, Day et al. 2012), which could have major impacts on a species that forages in shallow water.

Previous studies on wintering habitat in Arkansas and Louisiana addressed site-scale habitat associations of Rusty Blackbirds (Luscier 2009, DeLeon 2012). Rusty Blackbirds are ground foragers and require shallow water, often picking through wet leaf litter for aquatic invertebrates and mast (Avery 1995). Although previously thought to specialize in flooded forest areas, recent research has revealed that these birds also use areas such as suburban lawns and pecan groves (DeLeon 2012, Newell 2013). Using ground cover estimates at the 100 m scale, DeLeon (2012) found that Rusty Blackbird occupancy was associated with shallow water, wet leaf litter, and grass. Even though birds may be selecting sites for both habitat and food resources, few models also include a measure of food availability (Wolfe et al. 2014). More abundant food resources have positive effects on migratory bird survival and productivity (Jones et al. 2003, Seward et al. 2013). Because Rusty Blackbirds are likely choosing food-rich sites, I linked site-scale occupancy to habitat and invertebrate biomass.

Landscape-scale habitat could also be important to the wide-ranging Rusty Blackbird, especially since flooded habitat can have a variable distribution. Measuring the cover of forested wetlands and water-retaining soils, in addition to the amount of rainfall at sites, gives us a more complete picture of Rusty Blackbird habitat use. If habitat variables are measured at the wrong spatial scale, their importance may be under or overestimated (Girvetz and Greco 2009). For Northern Bobwhites, landscape-scale variables had a greater effect on occupancy than site variables (Duren et al. 2011). Since Rusty Blackbirds have an average foraging range of 5.08 km^2 on the wintering ground (Newell 2013; unpublished data analyzed by Borchert), I addressed habitat associations at a larger landscape scale of 600 m. A multi-scale approach to habitat may be more biologically meaningful than investigating smaller microhabitat scales alone.

If wintering habitat loss and degradation is responsible for the decline of Rusty Blackbirds, identification of important habitat components may greatly aid and target conservation actions. My primary objective was to quantify habitat associations with Rusty Blackbirds. To address this objective, I integrated variables at the site and landscape scales to: 1) determine how Rusty Blackbird site occupancy changes with landscape-scale (600 m) habitat cover, using remotely sensed datasets on land cover and mapped soils, and site-scale (100 m) habitat cover; 2) determine how transience (colonization and extinction) at the site-scale changes with ground cover variables, invertebrate biomass, and rainfall; and 3) link invertebrate biomass to site-scale ground cover and habitat type.

2.2 Methods

2.2.1 Study Area and Site Selection

My survey sites were concentrated in southeastern Louisiana, U.S.A., in the Lower Mississippi Alluvial Valley (Figure 2.1). Most sites included flooded areas, although water levels varied between sites and with rainfall. The majority of sites (97%) had some cover of either bottomland hardwood forest or baldcypress-tupelo-blackgum swamp (*Taxodium distichum*, *Nyssa aquatica*, and *Nyssa sylvatica*), but I also surveyed in suburban neighborhoods, parks, and pecan groves.

To fulfill detection requirements and model habitat associations, I selected sites where Rusty Blackbirds were already known to occur. Sites previously chosen through randomly stratified sampling yielded insufficient numbers of Rusty Blackbirds for modeling (DeLeon 2012), so I added more sites by using coordinates from birders subscribed to the LABIRD list-serve and from birder checklists posted to eBird (Sullivan et al. 2009), in addition to continuing surveys at existing sites. In order for a site to be surveyed, a coordinate had to be taken exactly where Rusty Blackbirds were previously observed.

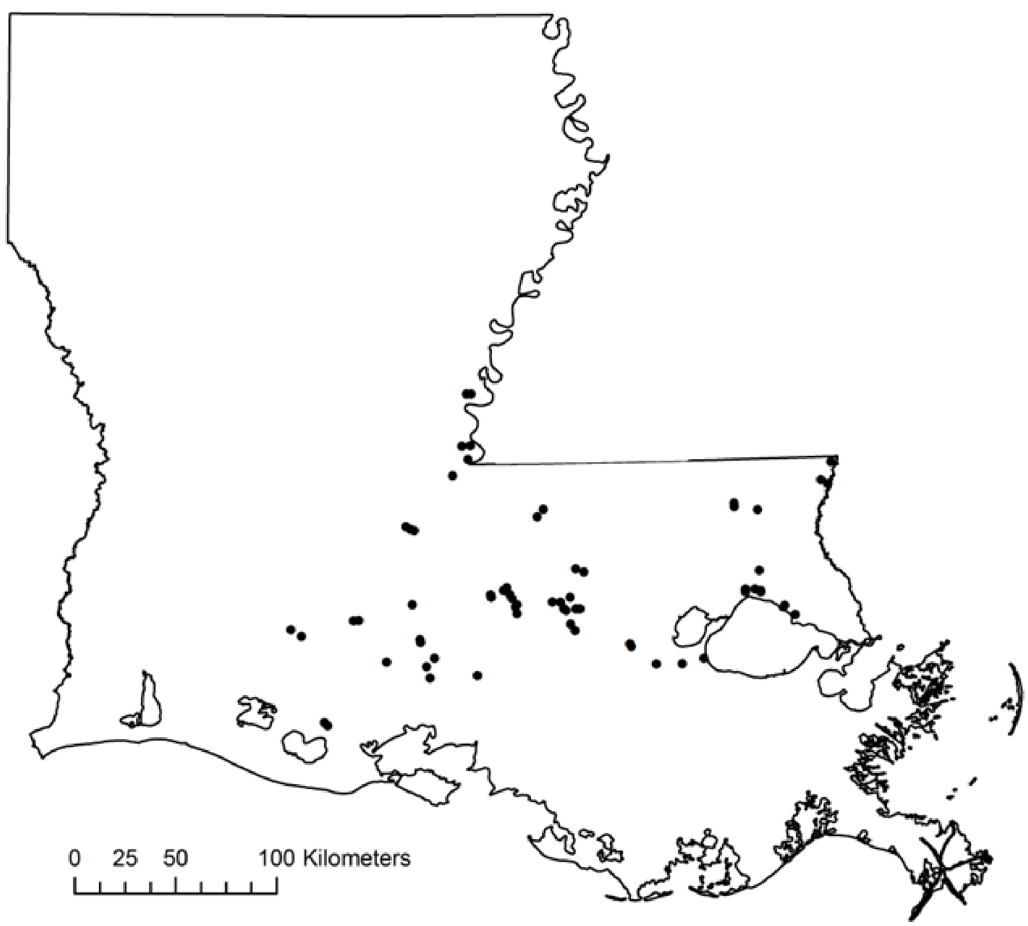

FIGURE 2.1. Locations of all 68 survey sites throughout Louisiana.

2.2.2 Avian Surveys, Site-level Habitat Surveys, and Invertebrate Sampling

I surveyed for Rusty Blackbirds in 2013 and 2014 from January 2 to March 9 to avoid migration periods (Luscier 2009). I visited 68 sites total, but I was only able to analyze a subset of sites depending on the analysis. During the first year, I surveyed consecutively twice during two rounds for a total of four surveys. I increased surveys during the second year to three each round for a total of six surveys. To satisfy the closure assumption for occupancy modeling, I completed surveys within a round within a four day window, but the majority of surveys were completed within 2 – 3 days. I surveyed clustered sites during the same time period to avoid the possibility of double counting flocks of birds.

Surveys began 30 minutes after sunrise and ended an hour before sunset to avoid roost-related travel (Avery 1995, Luscier et al. 2009). One observer recorded the number of Rusty Blackbirds, time, date, Beaufort wind speed, cloud cover (air moisture), and temperature. To examine flock dynamics, the observer also noted the number of blackbirds, European Starlings (*Sturnus vulgaris*), and American Robins (*Turdus migratorius*), in addition to their distance from each Rusty Blackbird group. I began surveys with a 10 minute stationary point count within 200 m, followed by a 30 minute walking extended search recording birds within 600 m (Figure 2.2). These distances were based on previous surveys, where 600 m was the approximate maximum distance that could be covered during extended searches (DeLeon 2012). The extended search maximized my detections because Rusty Blackbirds are often on the ground and patchily dispersed at sites, possibly due to water availability. To ensure spatial independence, each site was spaced at least 1200 m apart with no overlap between sites.

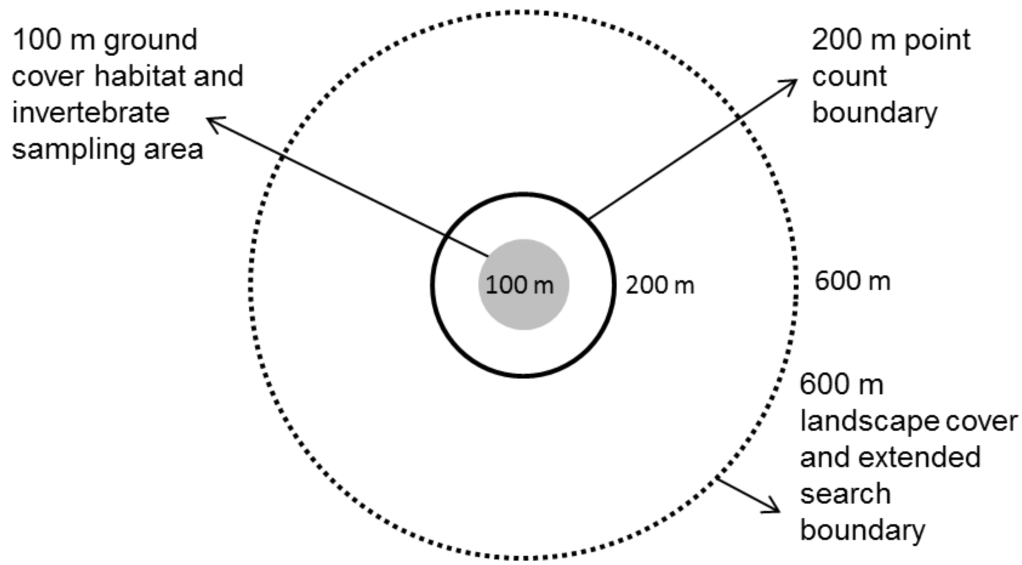

FIGURE 2.2. Rusty Blackbird surveys consisted of a 200 m radius stationary point count and a 600 m radius extended search (solid lines). I qualitatively surveyed habitat and collected invertebrate samples in the field within a 100 m radius circle. I only considered presence/absence data obtained during the 10 min point count within a 200 m radius.

I sampled habitat and invertebrates at the 100 m site-scale because this scale previously yielded significant habitat associations (DeLeon 2012). During each round, I visually estimated the percent of ground covered by water, leaf litter, grass, leafy vegetation, woody debris, impervious ground, and an “other” category which included all habitats not fitting into the previous categories (Appendix IV, Table IV.1). I also visually estimated the percent cover of general habitat types, including bottomland hardwood forest, baldcypress-tupelo-blackgum swamp, lawn, agriculture, developed land, open water, and an “other” category for undefined habitats (Appendix IV, Table IV.2). To sample invertebrates, five spatially independent (≥ 20 m apart) core samples (16 cm diameter) were collected within 100 m of the site center during each round. I paced out the distance between sampling locations and double checked the distance with a GPS. If I detected foraging Rusty Blackbirds during the extended search, I performed another 100 m habitat survey and collected another set of invertebrate samples.

Due to the patchiness of water at my sites, I used a location selection process to first sample the available wet substrates that Rusty Blackbirds would be foraging in before I sampled other substrates (Appendix I, Table I.1). My corer sampling protocol differed between wet and dry conditions. All samples included all available organic surface matter, including leaves, grass, and woody debris, and the first cm of soil. When the ground was wet, I targeted sampling to shallow water (< 5 cm) and leaf litter at the edge of standing water, followed by wet grass, if litter was unavailable. Shallow water and leaf litter were associated with larger flocks of Rusty Blackbirds (DeLeon 2012), and aquatic macroinvertebrates tend to be most dense at the water's margins (Ward 1992, Thorp and Covich 2010, and Lancaster and Downes 2013). If water depth increased rapidly (e.g. an eroded stream bank or canal), I sampled within 15 cm of the maximum water level. In dry conditions, the sampling priority was dry leaf litter followed by dry grass. At the end of each day, I rinsed samples in a 250 μm sieve and stored the organic matter in 95% ethanol. I removed invertebrates from each sample and classified them to order, separating only Diptera, Coleoptera, and Lepidoptera by life stage (adult, pupa, or larva). Some easily recognizable groups (e.g. ants) were identified to a lower taxonomic level. Each sample was dried at 60°C in an oven for a minimum of 48 hours, placed in a desiccator for at least 24 hours, and then massed to obtain the total dry mass and dry mass by order at each site/round.

2.2.3 Landscape-level (600 m) Habitat

Rusty Blackbirds are wide ranging and likely use habitat at larger scales than my 100 m site. Southeastern forested wetland ecosystems have transitioned from continuous tracts of land to patches embedded within an agricultural matrix (King et al. 2006). It is reasonable to assume that Rusty Blackbird presence at a site could be dependent on the cover of the surrounding matrix. I determined land cover composition within 600 m of my sites using 30 m resolution land cover data from the U.S. Geological Survey GAP Analysis Program (USGS GAP 2011) in ArcGIS (ESRI 2013). I determined the percentage of cover types for two different reclassification schemes (Appendix IV, Table IV.3). Class values from the GAP land cover were reclassified by the International Rusty Blackbird Working Group (IRBWG) for the entire wintering range and included floodplain forest, woody wetland, and developed land cover categories. To make Louisiana-specific management recommendations, I also reclassified GAP class values according to the habitat types identified in the Louisiana Comprehensive Wildlife Conservation Strategy (Lester et al. 2005; LCWCS, also known as the Wildlife Action Plan). I used the 600 m scale because it was the bound of the extended search, a reasonable approximation of foraging movements, and also the scale at which all sites were spatially independent. To verify that this scale was similar to a scale at which they would use the landscape, I used data collected from a study of radio-tracked Rusty Blackbirds wintering in South Carolina and Georgia to determine their average home range size (Newell 2013; unpublished data). I obtained kernel estimates of their utilization distribution to calculate their home range area (Worton 1989, 1995), which I defined as the minimum area in which a bird had a 95% probability of being located, for 17 birds ($\geq$ 25 locations per individual, 1474 locations total) using package ADEHABITAT (Calenge 2006) for Program R (R Core Team 2013). I averaged the 17 estimates to obtain an average home range size of 5.08 km^2. The 600 m scale accounts for 22% of their home range area; although the scale is not the exact same size as their range, it is reasonable to assume that Rusty Blackbirds would be using a subset of that area over the 2 – 3 day period of my surveys.

Rusty Blackbirds rely on shallow water; soil composition at sites may influence the persistence of ephemeral water after rainfall. I calculated percent cover of soil types at all scales

(100 m and 600 m) using the U.S. Department of Agriculture National Resources Conservation Service Soil Survey Geographic (SSURGO) Database vector layers, which are digitized from soil survey maps (Soil Survey Staff 2014). In Louisiana there are 315 soil series, but I aggregated them by their associated hydrologic group (Appendix IV, Table IV.4) before calculating cover.

To determine whether rainfall is related to Rusty Blackbird presence, I obtained the total rainfall at each site in the three days prior to each round using 4 km resolution daily precipitation raster data from Oregon State University's PRISM Climate Group (PRISM Climate Group 2014). The Parameter-elevation Relationships on Independent Slopes Model (PRISM) takes into account spatial climate patterns and adjusts precipitation in each pixel (4 km^2) using its location, elevation, coastal proximity, topographic facet orientation, vertical atmospheric layer, topographic position, and the orographic effectiveness of the terrain (Daly et al. 2008). Some sites overlapped with multiple rainfall grid cells; in those cases I weighted the precipitation value of the grid cell by the area located within each scale (100 m and 600 m) boundary.

2.2.4 Dynamic Occupancy Modeling Design

Determining why a target species may be using a site is a major goal of many wildlife ecology studies. Occupancy modeling can be used to estimate the number of sites occupied (presence or absence), while also accounting for the fact that a site may be occupied even if an animal is not detected. Failing to account for imperfect detectability can lead to an underestimation of occupancy probability (MacKenzie et al. 2002) and biased estimates of colonization and extinction probabilities (MacKenzie et al. 2003). I modeled detectability and occupancy as a function of survey and habitat characteristics and compared them using a maximum-likelihood procedure in package UNMARKED (Fiske and Chandler 2011) for program R (R Core Team 2013). Since Rusty Blackbirds may be using sites in response to changing habitat conditions, such as the presence of shallow water or the moistness of substrates, I used dynamic (also known as multi-season) occupancy models to compare changes in occupancy status to changes in habitat conditions at sites. A survey design of rounds within winters within years allowed me to examine how site occupancy (presence of birds), colonization (gaining of birds), and extinction (loss of birds) changed with ephemeral water or moisture within a winter. Unlike single-season occupancy, dynamic occupancy models relax closure assumptions (no movement into or out of sites) between rounds to allow for examination of changes in occupancy, colonization, and extinction over time (MacKenzie et al. 2006). Each round was assumed to be closed to immigration and emigration because I surveyed the same sites on mostly consecutive days. During the 2013 season, I surveyed twice per round and then increased the effort to three surveys per round during the 2014 season. Because the two years did not have a balanced number of surveys for the analysis of all four rounds, I used the first two surveys in the second year and disregarded the third (Figure 2.3). I used a separate model set for 2014 because invertebrate biomass was only measured in this year. I included all three surveys in the analysis (Figure 2.4).

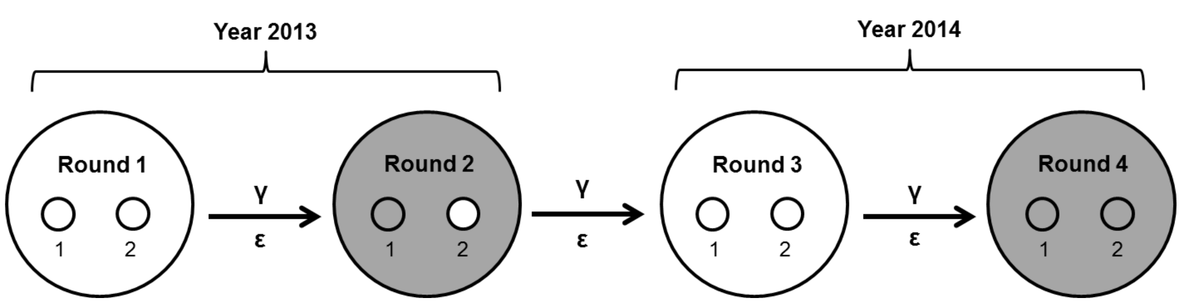

FIGURE 2.3. Survey design example for 2013-2014 incorporated four approximately monthly rounds (two per winter) with two surveys each. Occupancy (ψ) probability was estimated for all four rounds and colonization (γ) and extinction (ε) were estimated for the three time periods between rounds. At this site Rusty Blackbirds were not detected in rounds 1 and 3 (unoccupied), but they were detected during one survey in round 2 (gray = occupied) and during both surveys in round 4 (gray = occupied).

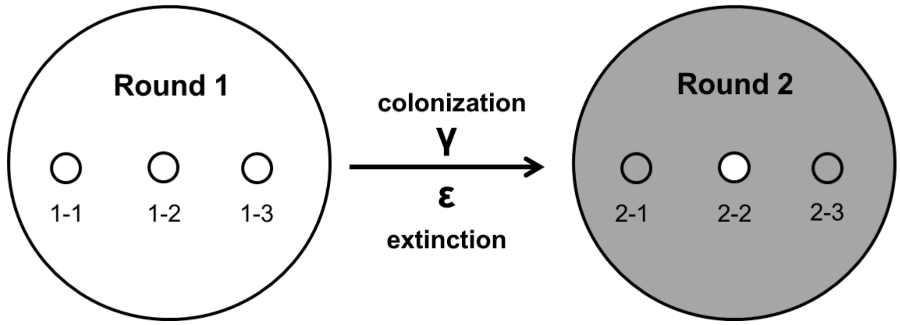

FIGURE 2.4. My survey design for 2014 incorporated two approximately monthly rounds per winter with three surveys each. In round one (Jan-Feb), this site was unoccupied during all three surveys (small circles 1-1, 1-2, and 1-3). During round 2 (Feb-Mar), birds were detected in surveys one and three (small circles 2-1 and 2-3) and the site was occupied (gray color).

After organizing the initial three model sets (Figures 2.3 and 2.4), the data were screened for completeness in detection history and covariates, which is required for dynamic occupancy models. During the first year (2013), some sites were dropped because they were located on inaccessible private property, were not spaced at least 1200 m apart, or were very unlikely to be occupied (e.g. a Rusty Blackbird had been seen perched at the site once, but it was an isolated incident and it would not actually forage there) and therefore not surveyed each round. The screening process indicated the need for two separate model sets: a dynamic (four round, two survey) model set with 100 m scale habitat variables and a dynamic (two round, three survey) model set that included invertebrate biomass. All four round, two survey models for 2013 – 2014 (Figure 2.3) included only 36 of 68 possible sites. The two round, three survey models incorporating 2014 invertebrate biomass included all 57 sites surveyed in that year. Although I intended to use larger spatial scales for modeling (600 m IRBWG and LCWCS), during initial modeling, unexpected relationships between landscape-scale variables and occupancy probabilities were detected. Therefore, I included 100 m scale variables in an additional set of models to determine the influence of scale. Additionally, the complete detection histories in the two round, three survey 2014 model permitted abundance-adjusted modeling of sites where at least four birds (the median of counts at occupied sites) were detected, to explore whether habitat associations might change for larger flocks (Luscier et al. 2010, DeLeon 2012). Finally, it should be noted that the invertebrate biomass data included estimates from the 100 m scale (47 sites) and 200 – 400 m scale (10 sites), which were judged to be equivalent for modeling.

2.2.5 Detectability and Occupancy Model Building and Selection

When building models with explanatory variables for Rusty Blackbird presence-absence, I only used habitat variables that I thought would be important for the birds, based on my observations and the aforementioned literature. I included detectability covariates that I believed would influence my ability to detect Rusty Blackbirds at sites (Table 2.1). Prior to constructing all models, I eliminated variables that lacked biological relevance or were highly correlated (Spearman Rank Correlation Test $|\rho| \geq 0.5$) with other variables. My first step was to construct a set of candidate detectability models. Once determining a best-fit detectability model (selection described below), I included it as the base model for my habitat-related occupancy models.

TABLE 2.1 Covariates used to model detectability and their correlations. Correlations between pairs of variables used are listed. No variables were significantly correlated, although the number of RWBL and the flock size were correlated in less than 50% of surveys.

covariate	description	correlations
year	year surveyed (2013 or 2014)	none
julian	julian date of survey	none
time	standardized time of day	none
weather	measure of air moisture (sunny = 1, partly cloudy = 2, overcast = 3, rain = 4)	none
wind	Beaufort scale (1-5) wind speed	none
prior	Rusty Blackbirds previously detected within 200 m? (yes/no)	none
flock	# other blackbirds, American Robins, and European Starlings detected within 200 m	(<50% of surveys correlated with RWBL)
COGR	# Common Grackles detected within 200 m	none
RWBL	# Red-winged Blackbirds detected within 200 m	(<50% of surveys correlated with flock)

Five habitat covariates that changed between rounds, including ground cover, rainfall, and biomass, were used to estimate colonization and extinction (Table 2.2). I was interested in how these dynamic habitat covariates could make habitat at sites more or less attractive to Rusty Blackbirds (Table 2.3).

TABLE 2.2. Dynamic habitat covariates used to model colonization (γ) and extinction (ε) at sites. For a full list of collected covariates see Appendix IV. Correlations between pairs of variables used are listed. I only included biomass in the model set for the year 2014.

covariate	description	correlations
shallow	% ground (100 m) covered by shallow water	2 rounds: wetlitter, soild
wetlitter	% ground covered (100 m) by wet litter (damp and saturated categories)	2 rounds: shallow, lawn
wetgrass	% ground covered (100 m) by wet grass	4 rounds: lawn (100 m) 2 rounds: lawn (100 m)
rain	total rainfall (mm) within 600 m in the 3 days prior to a round	none
biomass	total invertebrate dry mass (5 samples/round)	none

For the 600 m landscape variables, I constructed candidate model sets corresponding to two different habitat classifications (IRBWG and LCWCS), with each set including the previously identified detectability variables. I ranked models using Akaike's Information Criterion to determine the models best explaining observed occupancy and detectability rates. I did not use AICc because the effective sample size for hierarchical models remains unclear (R. Chandler and J. A. Royle, pers. communication). To test goodness of fit, I used the MacKenzie-Bailey test included in package AICCMODAVG (Mazerolle 2015) for Program R (R Core Group 2013). MacKenzie-Bailey tests goodness of fit of dynamic occupancy models by computing a Pearson chi-square statistic for the occupancy estimates from each round (season), summing them, and using a parametric bootstrap procedure to determine whether the observed statistic is unexpectedly large (MacKenzie and Bailey 2004). If necessary (i.e., multiple models within 2 ΔAIC), the best supported model parameter estimates underwent model averaging following Burnham and Anderson (2002).

TABLE 2.3. Fixed habitat covariates at site (100 m) and landscape (600 m) scales used to model occupancy (ψ) at sites.

covariate	description	correlations
International Rusty Blackbird Working Group Landscape Cover (600 m)		
floodplain forest	% cover of floodplain forest	none
woody wetland	% cover of woody wetland	none
developed	% cover of developed land (grassy areas, pavement, buildings, etc.)	none
soil C	% cover of soil hydrologic group C (slow rate of water transmission)	soil D
soil D	% cover of soil hydrologic group D (very slow rate of water transmission)	soil C
Louisiana Comprehensive Wildlife Conservation Strategy Landscape Cover (600 m)		
bottomland hardwood forest	% cover of bottomland hardwood forest	none
cypress-tupelo-blackgum swamp	% cover of cypress-tupelo-blackgum swamp	none
lawn	% cover of low intensity development (lawn)	none
soil C	% cover of soil hydrologic group C (slow rate of water transmission)	soil D
soil D	% cover of soil hydrologic group D (very slow rate of water transmission)	soil C
Field Estimated Habitat Cover (100 m)		
bottomland hardwood forest	% cover of bottomland hardwood forest	4 rounds: lawn 2 rounds: lawn
cypress-tupelo-blackgum swamp	% cover of cypress-tupelo-blackgum swamp	4 rounds: lawn, soilc 2 rounds: water
lawn	% cover of lawn	4 rounds: blh, wetgrass 2 rounds: blh, wetgrass, wetlitter
soil C	% cover of soil hydrologic group C (slow rate of water transmission)	4 rounds: swamp, soild 2 rounds: soild
soil D	% cover of soil hydrologic group D (very slow rate of water transmission)	4 rounds: soilc 2 rounds: soilc, shallow

***** Cover of soil C and soil D data were obtained from the Soil Survey Geographic Database (USDA NRCS 2009) and included in the field estimated habitat cover model set.

2.2.6 Deep Water and Detection Probability in Forested Wetlands

To investigate the relationship between important habitat types and deep water on detection probability, I fit several types of models using packages GNM (Turner and Firth 2012) and GAM (Hastie 2013) in program R (R Core Group 2013). The first model set included intrinsically linear, linearized power, exponential decay, polynomial quadratic, spline, and loess, which were ranked by lowest AIC value to determine which type best modeled the relationship of the data. Intrinsically linear models of log transformed data were the most appropriate for describing the relationship of deep water cover with forested wetland habitat types. Although the deep water data were transformed for analysis, I applied power trend lines to the raw data to aid in interpretation.

Additionally, I ran linear models of detection probability (averaged over the 4 rounds) with the percent cover of forested wetlands to determine how detection probability changes in these habitats. I assumed there was a functional relationship between detection probability and the percent cover of floodplain forest/bottomland hardwood forest or woody wetland/cypress-tupelo-blackgum swamp if a model was significant at $\alpha = 0.05$ (Gotelli and Ellison 2004).

I used Nagelkerke's (1991) pseudo-R^2 for all regressions because percent cover has a binomial distribution. For the logistic regressions (percent cover of deep water with forested wetlands) in particular, the fit must be calculated using a pseudo-R^2 because the model estimates from a regression model are based on maximum likelihood and are not calculated to minimize variance.

2.2.7 Mixed-effects Models of Invertebrate Biomass

I used linear mixed-effects modeling to examine the relationship between invertebrate biomass and 100 m site-scale ground cover and habitat covariates as fixed effects (with no interactions). As a random effect, I included round (two rounds total). Macroinvertebrates are frequently more influenced by local rather than large-scale influences (Lammert and Allan 1999, Sponseller et al. 2001, Batzer 2013), therefore, invertebrate models included only 100 m site scale data. I did not include correlated variables (Spearman Rank Correlation Test, $|\rho| \geq 0.50$) in the same models. To better meet model assumptions, I applied a natural log transformation to the response variable (Zuur et al. 2009). I fitted models using restricted maximum likelihood estimation in package LME4 (Bates et al. 2014) for program R (R Core Team 2013). Highest ranked models had the lowest Akaike Information Criteria (AIC); I considered models within Δ2 AIC to have the most support. I obtained p-values for the best supported models through restricted maximum likelihood t-tests using the Satterthwaite approximation to degrees of freedom in package LMERTEST (Kuznetsova et al. 2014).

2.3 Results

2.3.1 Detectability Analysis

For the dynamic (four round, two survey) model set which included data from 36 sites, the top model p(cogr+time) received the best support (within Δ2 AIC) and accounted for 67% of the available weight (Table 2.4). The top model showed adequate fit, indicating that my covariates acceptably predicted detection probability ($\chi^2 = 11.5$, $p = 0.20$, $\hat{c} = 1.35$). The number of Common Grackles was positively associated with Rusty Blackbird detectability, indicating that the probability of detecting a Rusty Blackbird increased if there were more grackles present during surveys (Figure 2.6). Time was also important; detectability decreased the later in the day a survey was conducted (Figure 2.6). Of the two covariates, time had a larger influence on detectability (Figure 2.5). For the dynamic (two round, three survey) models of 57 sites, p(cogr+time) was again the top model; Common Grackles had a positive relationship with detection probability (0.44 ± 0.28) and time of day had a negative relationship (-5.63 ± 2.11).

TABLE 2.4. Top six detectability models for 36 sites/4 rounds accounting for 94% of the model weight. One model within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	-2 log likelihood
1	p(cogr+time)	270.54	0.00	0.67	6	129.27
2	p(flock+time)	274.47	3.93	0.09	6	131.24
3	p(cogr+weather)	274.56	4.02	0.09	6	131.28
4	p(flock+weather)	276.10	5.55	0.04	6	132.05
5	p(cogr)	276.34	5.80	0.04	5	133.17
6	p(cogr+wind)	278.22	7.67	0.01	6	133.11
18	**p(null)**	288.70	18.16	0.00	4	140.35
29	**p(global)**	309.43	38.89	0.00	13	141.71

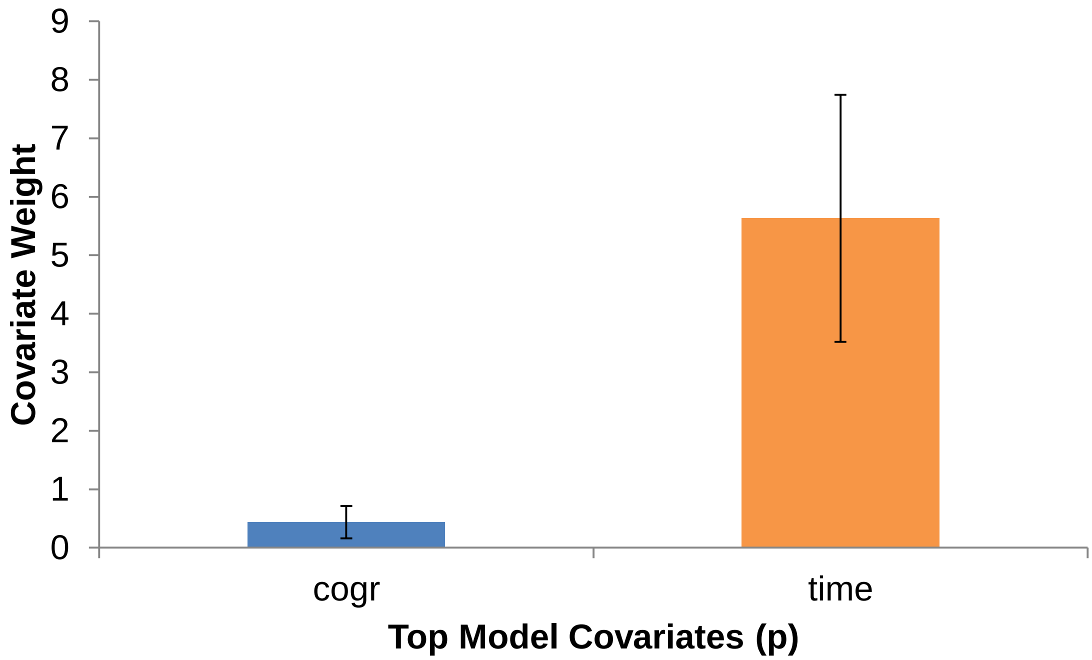

FIGURE 2.5. The two covariates (estimates ± SE) influencing detectability at 36 sites/4 rounds.

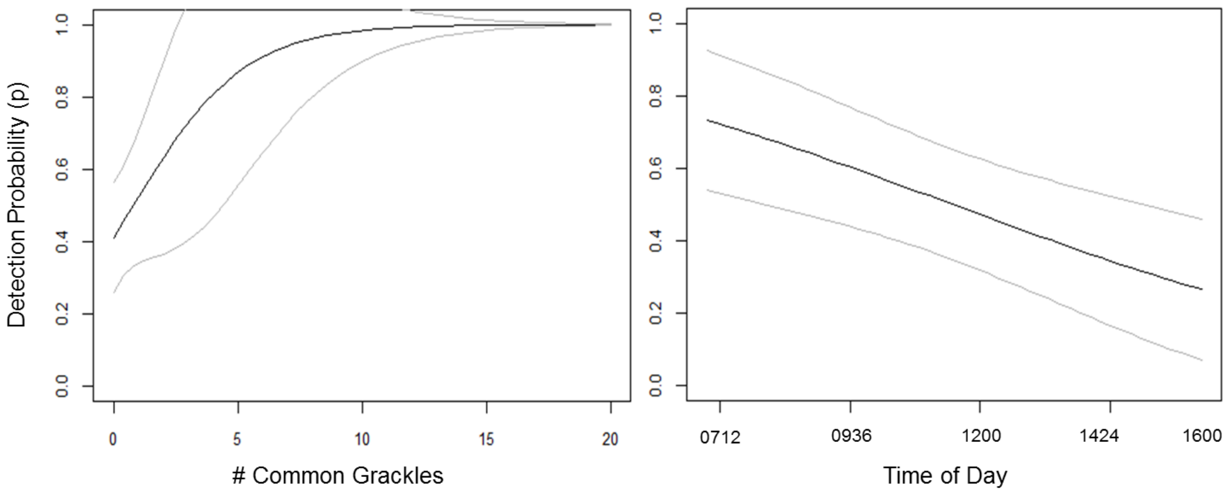

FIGURE 2.6. Predicted detection probability with Common Grackles (using model p(cogr)) or Time of Day (using model p(time)). For time, 0700 to 1600 hours was the approximate time interval when surveys took place (survey start and end times varied with sunrise and sunset). Gray lines represent the 95% confidence intervals of the estimates.

2.3.2 Dynamic (Four Round, Two Survey) Habitat Analysis

Over two years and four rounds, naïve occupancy at my sites was relatively high (69%). For the IRBWG habitat classes, three top models explained associations between habitat variables and occupancy, colonization, and extinction rates and accounted for 54% of the available model weight (Table 2.5). The best model exhibited acceptable fit to the data, indicating that model set results should reflect real relationships between occupancy and habitat covariates ($\chi^2 = 13.2$, $p = 0.25$, $\hat{c} = 1.34$).

Because multiple models were within 2 ΔAIC, the following results represent model averaged estimates (Burnham and Anderson 2002). After model averaging, floodplain forest received about six times more support than woody wetland (0.06 vs 0.01) (Figure 2.7). Of the 36 sites, 34 (94%) had at least some floodplain forest present and 29 (81%) had some woody wetland present. Unexpectedly, the relationship between occupancy and floodplain forest or woody wetland was negative (Figure 2.8). Original covariate weight estimates ± SE were -0.06 ± 0.03 for floodplain forest and -0.01 ± 0.02 for woody wetland.

TABLE 2.5. Top three habitat association models for 36 sites/4 rounds (600 m International Rusty Blackbird Working Group fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	**model**	**AIC**	**ΔAIC**	**AICwt**	**k**	**n**	**-2 log likelihood**
1	ψ(FF)	267.36	0.00	0.25	7	36	126.68
2	ψ(FF),γε(wetlitter)	267.84	0.48	0.20	9	36	124.92
3	ψ(FF+WW)	269.13	1.77	0.10	8	36	126.57
27	**ψγε(global)**	277.41	10.04	0.00	18	36	120.70
30	**ψγεp(null)**	288.70	21.33	0.00	4	36	140.35

*FF = floodplain forest, WW = woody wetland

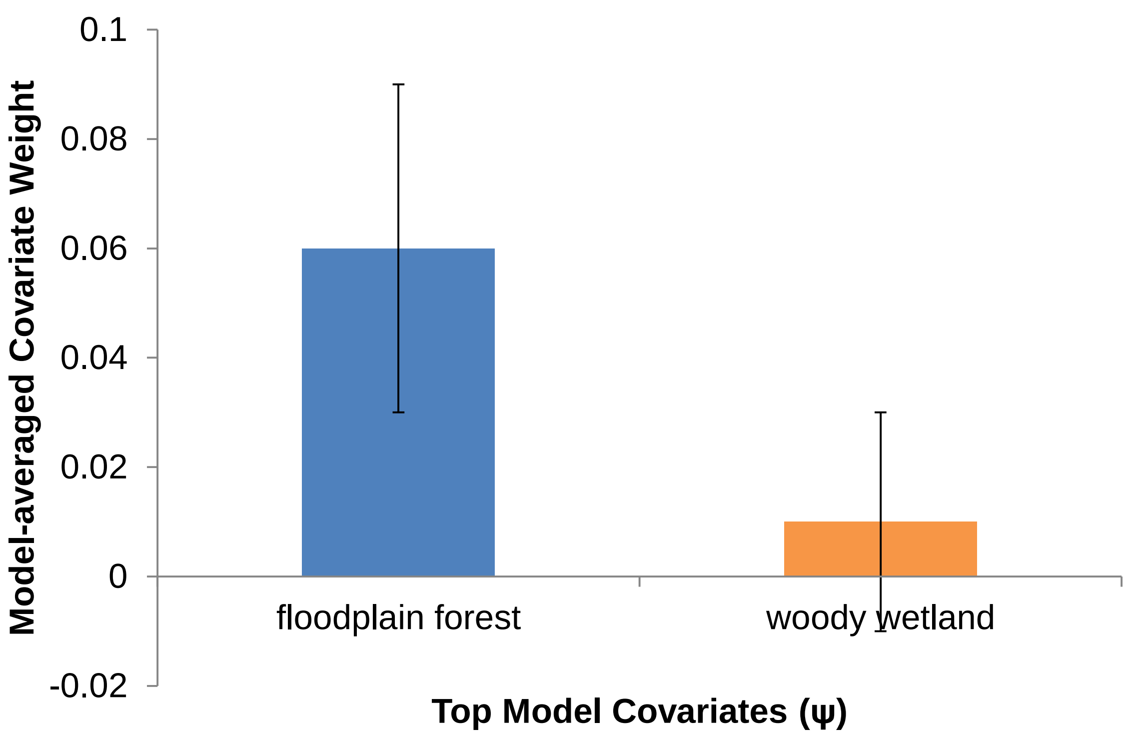

FIGURE 2.7. Absolute values of model averaged covariate weights ± SE for the most important International Rusty Blackbird Working Group habitat classes (600 m) associated with occupancy at 36 sites/4 rounds.

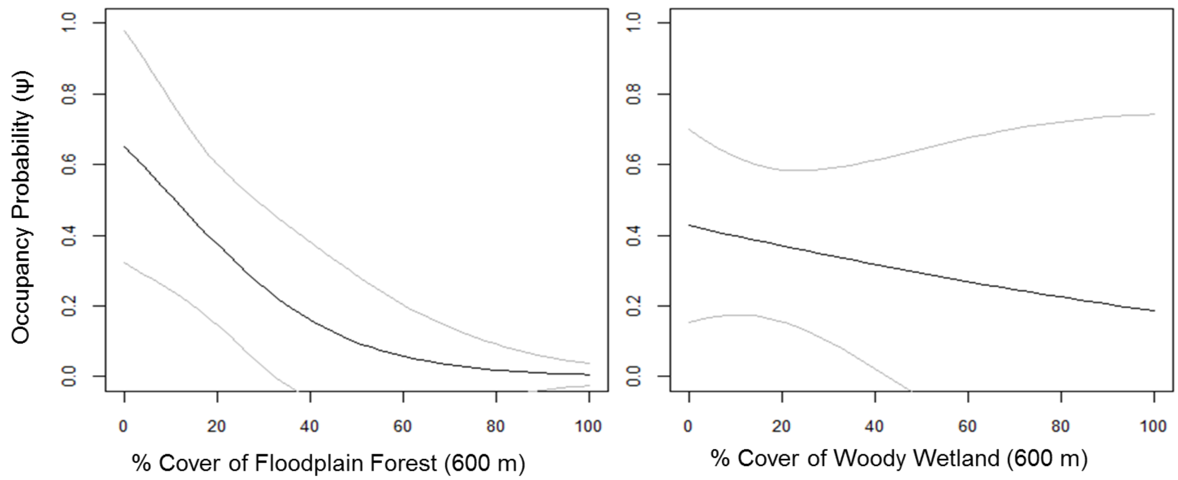

FIGURE 2.8. Predicted occupancy probability with the most important International Rusty Blackbird Working Group landscape (600 m) habitat covariates (based on models ψ(FF) and ψ(WW), respectively). Gray lines represent the 95% confidence intervals of the estimates.

Wet leaf litter was the most important variable for colonization (0.06 ± 0.05) or extinction (-0.01 ± 0.04) at a site. As expected, there was a positive relationship between wet leaf litter and colonization and a negative relationship with extinction (Figure 2.9). The confidence intervals are wide, especially for extinction probability, reflecting the low sample of colonization or extinction events at a particular level of wet leaf litter.

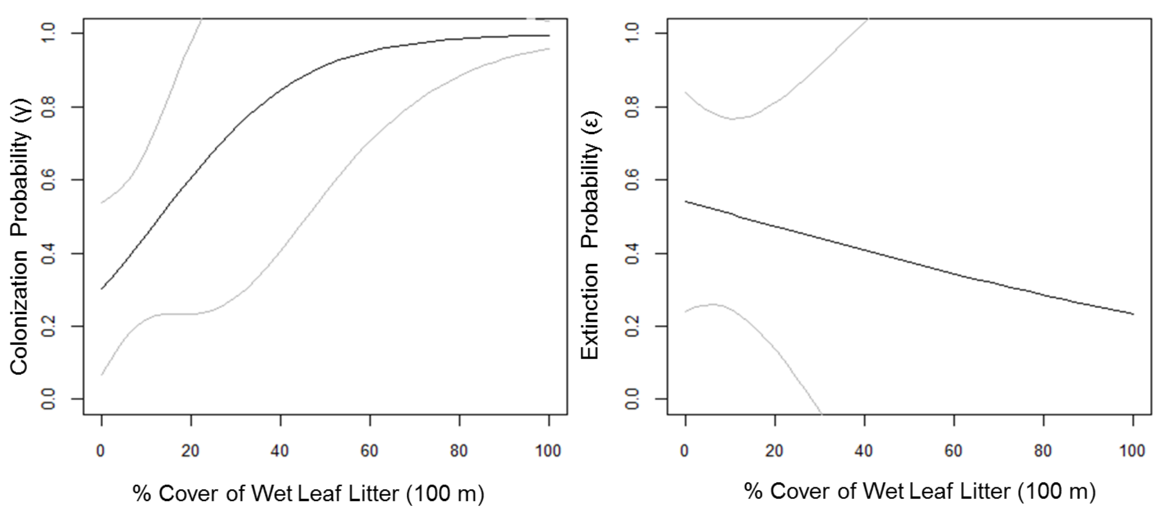

FIGURE 2.9. Predicted colonization and extinction probability with wet leaf litter (100 m) at 36 sites/4 rounds (based on model ψ(FF),γε(wetlitter)). Gray lines represent 95% confidence intervals of the estimates.

For the LCWCS habitat classes, nine top models accounted for 74% of the available weight (Table 2.6). The top model fit adequately ($\chi^2 = 11.2$, $p = 0.23$, $\hat{c} = 1.29$).

TABLE 2.6. Top nine habitat association models for 36 sites/4 seasons (600 m Louisiana Comprehensive Wildlife Conservation Strategy fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	n	-2 log likelihood
1	ψ(BLH)	270.59	0.00	0.14	7	36	128.29
2	ψ(BLH),γε(wetlitter)	271.06	0.47	0.11	9	36	126.53
3	ψ(BLH+swamp)	271.47	0.88	0.09	8	36	127.73
4	ψ(swamp)	271.52	0.94	0.09	7	36	128.76
5	ψ(BLH+swamp),γε(wetlitter)	271.96	1.37	0.07	10	36	125.98
6	ψ(swamp),γε(wetlitter)	272.02	1.43	0.07	9	36	127.01
7	ψ(soilc)	272.11	1.52	0.07	7	36	129.05
8	ψ(lawn)	272.35	1.76	0.06	7	36	129.17
9	ψ(soild)	272.54	1.95	0.05	7	36	129.27
13	**ψγε(global)**	274.43	3.84	0.02	19	36	118.21
30	**ψγεp(null)**	288.70	18.11	0.00	4	36	140.35

*BLH = bottomland hardwood forest, soild = soil hydrologic group D, soilc = soil hydrologic group C

After model averaging, the bottomland hardwood forest estimate had the greatest magnitude, but it was within the range of error for all variables (Figure 2.10). The confidence intervals of every variable overlapped with zero, suggesting that they lacked model support. Again, the top habitat variables (bottomland hardwood forest and cypress-tupelo-blackgum swamp) had a negative relationship with occupancy probability (Figure 2.11). The original covariate weight estimates ± SE were bottomland hardwood forest -0.03 ± 0.03, cypress-tupelo-blackgum swamp -0.02 ± 0.02, lawn -0.01 ± 0.03, soil hydrologic type C 0.01 ± 0.01, and soil hydrologic type D 0 ± 0.01. Wet leaf litter was the only well supported dynamic variable; it positively affected colonization probability (0.06 ± 0.05) and negatively affected extinction probability at my sites (-0.01 ± 0.04).

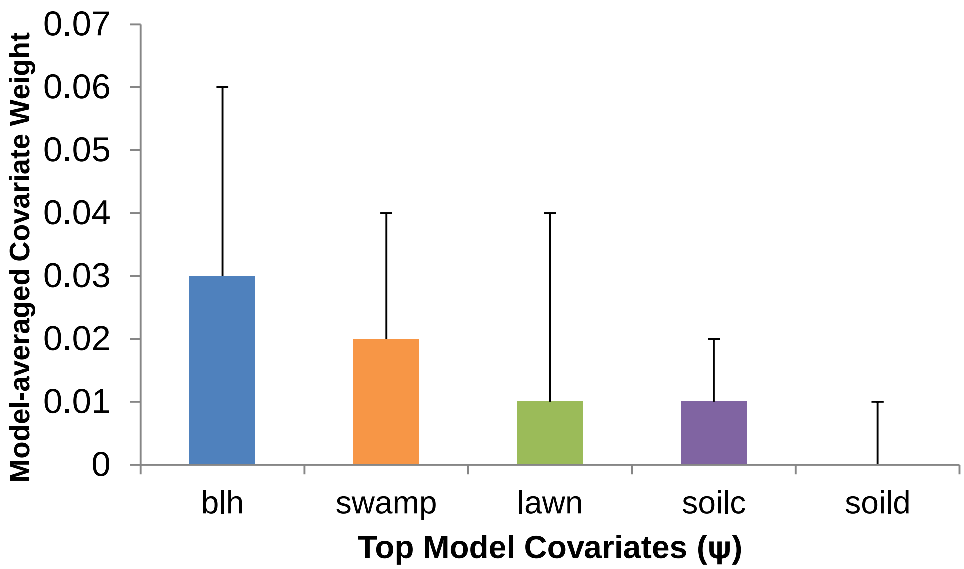

FIGURE 2.10. Absolute values of model averaged covariate weights ± SE for Louisiana Comprehensive Wildlife Conservation Strategy habitat classes (600 m) associated with occupancy at 36 sites/4 rounds. BLH = bottomland hardwood forest, swamp = cypress-tupelo-blackgum swamp, soilc = soil hydrologic group C, and soild = soil hydrologic group D.

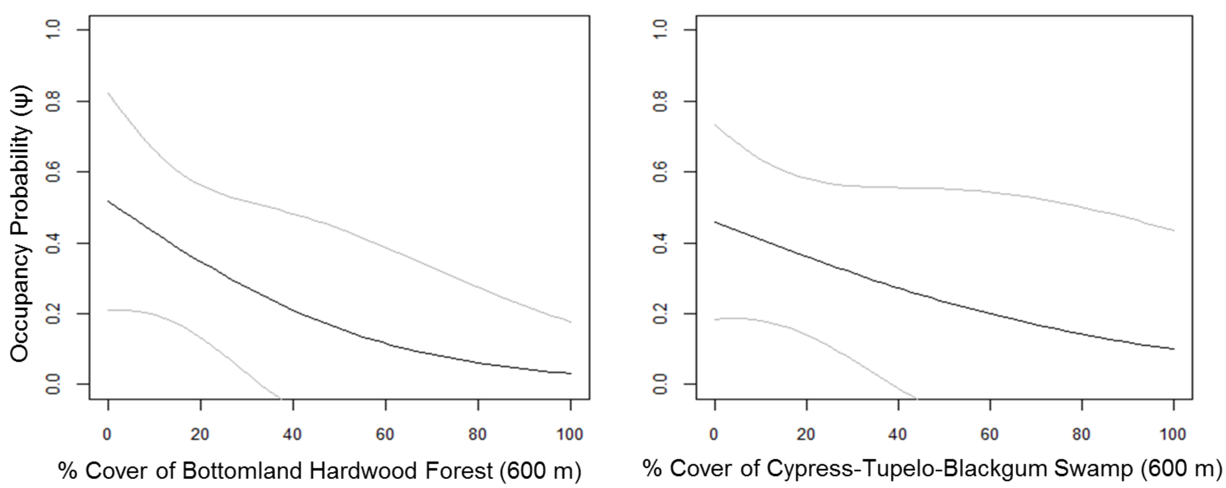

FIGURE 2.11. Relationship between occupancy probability and the most important Louisiana Comprehensive Wildlife Conservation Strategy landscape habitat covariates (based on models ψ(BLH) and ψ(swamp), respectively). Gray lines represent 95% confidence intervals of the estimates.

To ensure that the negative relationships between occupancy probability and forested wetlands were not simply due to inappropriately large scales or error associated with classifying remotely sensed imagery, I also constructed a set of candidate models using 100 m site-scale field-estimated habitat cover. Eight models accounted for 74% of the weight (Table 2.7). The top model showed adequate fit ($\chi^2 = 12.2$, $p = 0.20$, $\hat{c} = 1.34$).

TABLE 2.7. Top eight habitat association models for 36 sites/4 rounds (100 m fixed (ψ) field-estimated habitat covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AIC wt	k	n	-2log likelihood
1	ψ(soilc)	270.37	0.00	0.15	7	36	128.19
2	ψ(BLH100)	270.82	0.45	0.12	7	36	128.41
3	ψ(BLH100),γε(wetlitter)	271.09	0.71	0.10	9	36	126.54
4	ψ(BLH100+swamp100)	271.46	1.09	0.09	8	36	127.73
5	ψ(BLH100+swamp100),γε(wetlitter)	271.58	1.21	0.08	10	36	125.79
6	ψ(lawn100)	271.69	1.32	0.08	7	36	128.84
7	ψ(soild)	271.91	1.54	0.07	7	36	128.96
8	ψ(swamp100)	272.12	1.75	0.06	7	36	129.06
26	**ψγε(global)**	282.89	12.52	0.00	17	36	124.45
27	**ψγεp(null)**	288.70	18.33	0.00	4	36	140.35

*BLH = bottomland hardwood forest, soilc = soil hydrologic group C, soild = soil hydrologic group D

Again, the relationship between occupancy probability and forested wetland habitat types was negative, indicating that the direction of the relationship was not due to the larger scale in my original analysis. The most important variable associated with occupancy was soil hydrologic group C (Figure 2.12), which was positively associated with site occupancy (Figure 2.13). Original covariate weight estimates ± SE were -0.02 ± 0.02 for bottomland hardwood forest, -

0.01 ± 0.02 for cypress-tupelo-blackgum swamp, 0.01 ± 0.02 for lawn, 0.03 ± 0.02 for soil C, and -0.01 ± 0.01 for soil D. The model averaged parameter estimates for wet leaf litter, which appeared in two top models, were 0.07 ± 0.05 (γ, colonization) and -0.01 ±0.04 (ε, extinction).

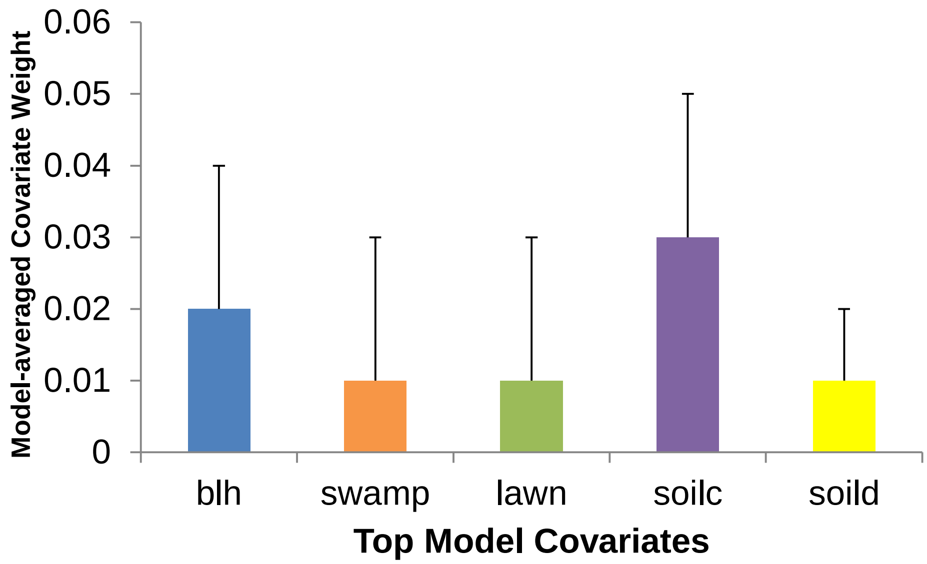

FIGURE 2.12. Absolute values of model averaged covariate weights ± SE for 100 m site-scale variables associated with occupancy at 36 sites/4 rounds. BLH = bottomland hardwood forest, swamp = cypress-tupelo-blackgum swamp, soilc = soil hydrologic group C, and soild = soil hydrologic group D

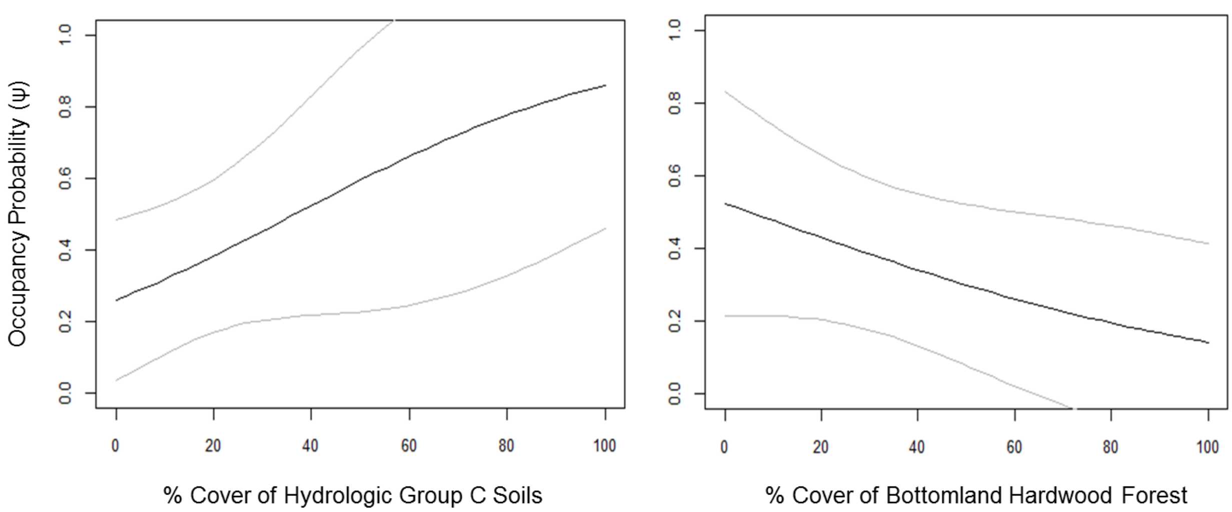

FIGURE 2.13. Predicted occupancy probability with the most important 100 m site-scale habitat variables (based on models ψ(soilc) and ψ(BLH100), respectively). Gray lines represent 95% confidence intervals of the estimates.

2.3.3 Relationship between Forested Wetlands and Deep Water

To determine whether the negative relationship between occupancy and deep water could be due to high cover of deep water at my sites, I used linearized power models to assess the relationship between natural log transformed deep water and forested wetland cover at 36 sites. For IRBWG 600 m habitat classes, there was a statistically significant ($\alpha = 0.05$) relationship between deep water and floodplain forest or woody wetland during all rounds except for round three (Figure 2.14).

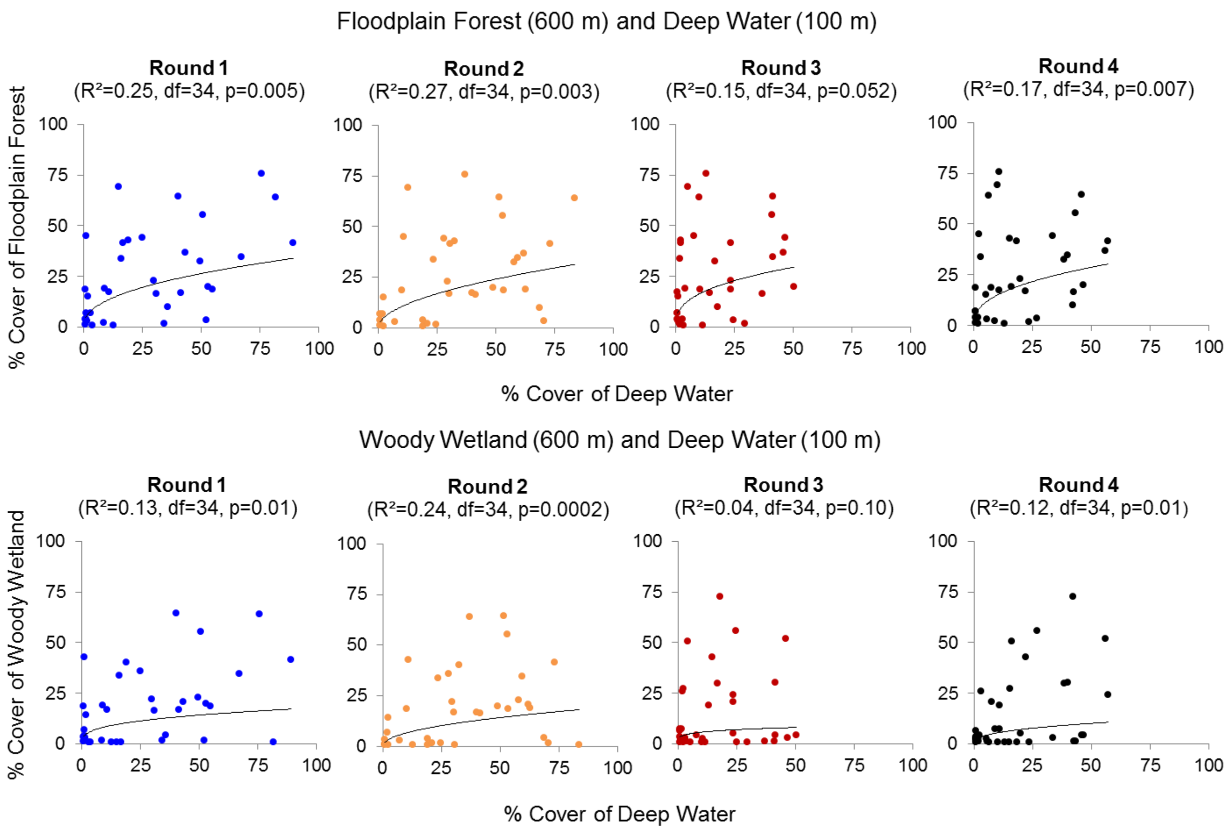

FIGURE 2.14. Linearized power models of percent cover of deep (≤ 5cm) water (100 m) and International Rusty Blackbird Working Group habitat classes (600 m) at sites. All models were significant ($\alpha = 0.05$) except for both habitat types during round 3.

Similarly, for LCWCS habitat classes, all models of the relationship between deep water and bottomland hardwood forest or cypress-tupelo-blackgum swamp were statistically significant (α = 0.05) for every round (Figure 2.15).

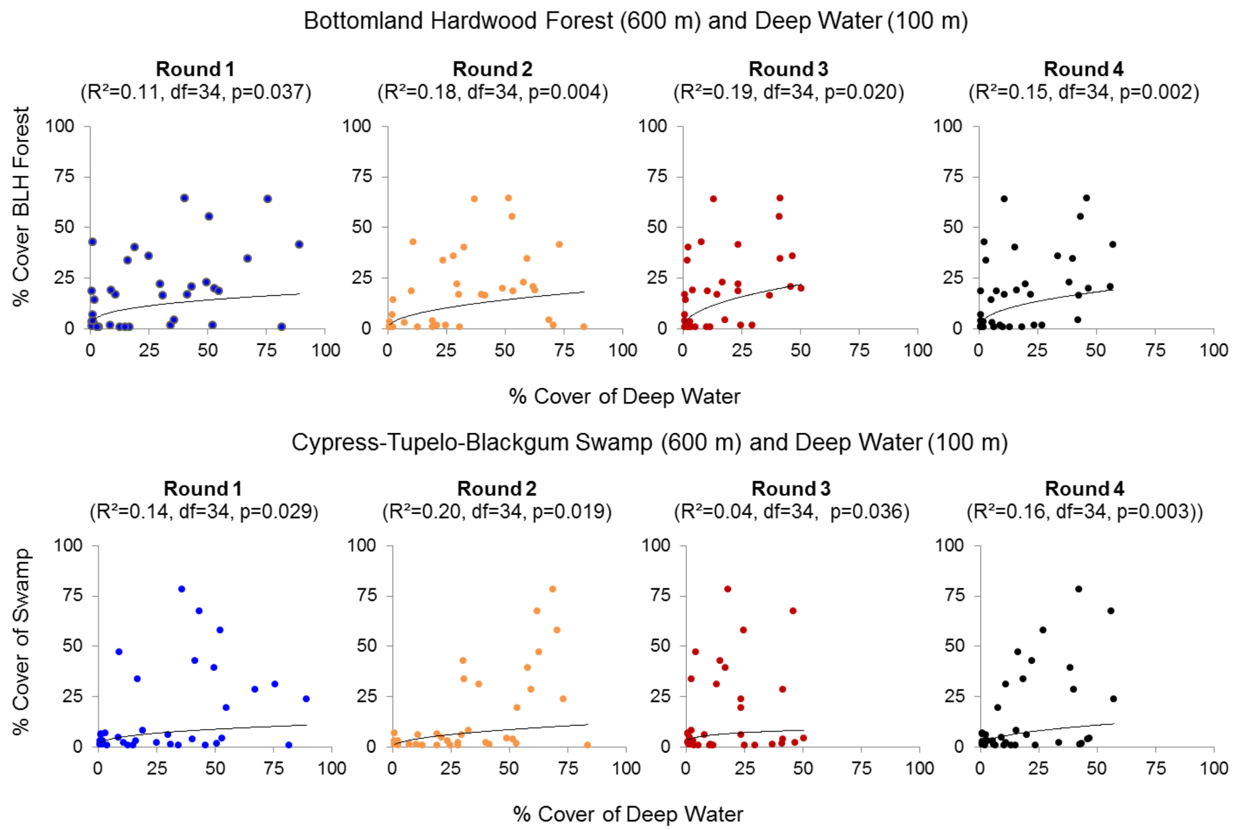

FIGURE 2.15. Linearized power models of percent cover of deep (≤ 5 cm) water (100 m) and Louisiana Comprehensive Wildlife Conservation Strategy habitat classes (600 m) at sites. Models were significant (α = 0.05) for every round.

I compared the relationship of deep water to site-scale (100 m) field estimated habitat and found the same trends, but they were weaker for bottomland hardwood forest. I only found a statistically significant (α = 0.05) relationship between the cover of site-scale bottomland hardwood forest and deep water during round two, although this relationship was significant for cypress-tupelo-blackgum swamp during all rounds (Figure 2.16).

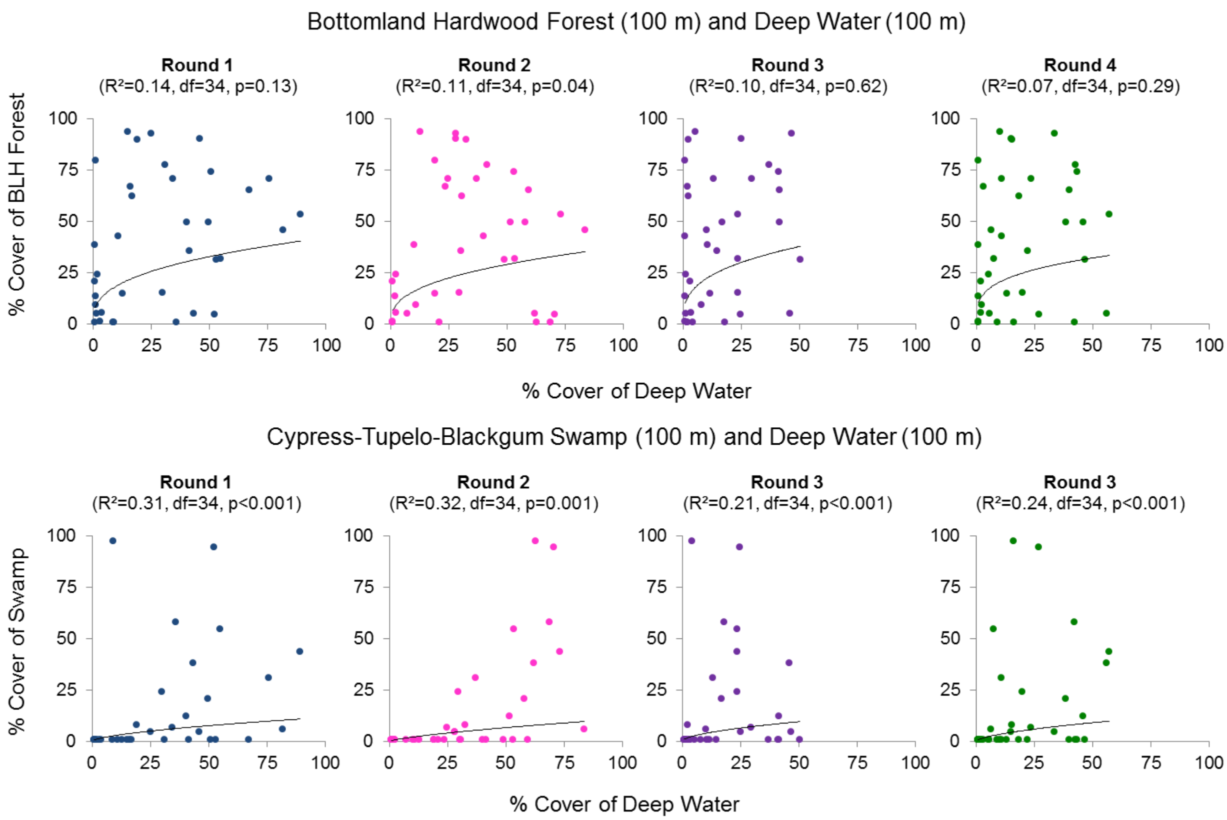

FIGURE 2.16. Linearized power models of percent cover of deep (≤ 5 cm) water (100 m) and field-estimated site-scale habitat classes (100 m). All models were significant (α = 0.05) except bottomland hardwood forest during rounds 1, 3, and 4.

2.3.4 Detection Probability in Forested Wetlands

I also wanted to examine whether there was a tendency for detection probability to decrease in forested wetlands, which could possibly produce lower occupancy estimates in these habitats. For IRBWG 600 m habitat classes, there was a statistically significant (α = 0.05) relationship between detection probability and floodplain forest, but not for woody wetland (Figure 2.17). For LCWCS habitat classes, there was no statistically significant relationship between detection probability and bottomland hardwood forest or cypress-tupelo-blackgum swamp (Figure 2.18). At the 100 m scale, there was a statistically significant (α = 0.05) relationship between field-estimated bottomland hardwood forest and detection probability, but not with cypress-tupelo-blackgum swamp (Figure 2.19).

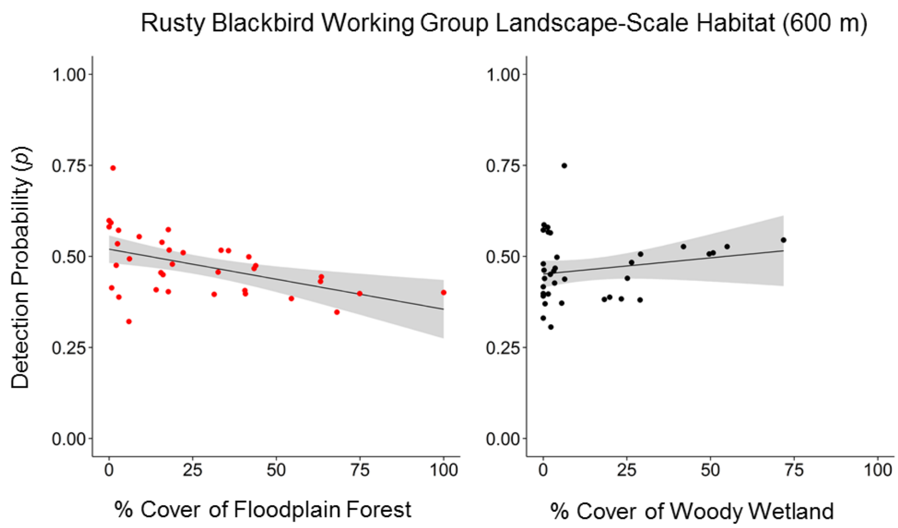

FIGURE 2.17. Intrinsically linear models of detection probability (averaged over 4 rounds) and percent cover of International Rusty Blackbird Working Group habitat classes (600 m) at 36 sites. Gray areas represent the 95% confidence intervals of the 36 estimates. The negative relationship between detection probability and floodplain forest cover was significant ($R^2 = 0.24$, df = 34, $p < 0.01$). There was no significant relationship between detection probability and woody wetland cover ($R^2 = 0.04$, df = 34, $p = 0.26$).

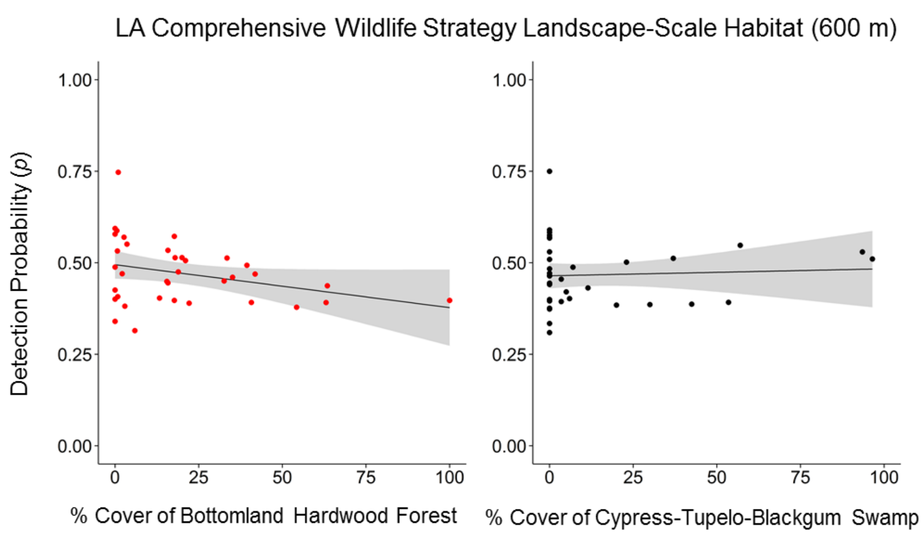

FIGURE 2.18. Intrinsically linear models of detection probability (averaged over 4 rounds) and percent cover of Louisiana Comprehensive Wildlife Conservation Strategy Landscape-Scale habitat (600 m) at 36 sites. Gray areas represent the 95% CI standard error of the 36 estimates. There was a negative trend between bottomland hardwood forest cover and detection probability, but it was not significant ($R^2 = 0.10$, df = 34, $p = 0.07$). There was no significant relationship between detection probability and cypress-tupelo-blackgum swamp cover ($R^2 = 0.02$, df = 34, $p = 0.47$).

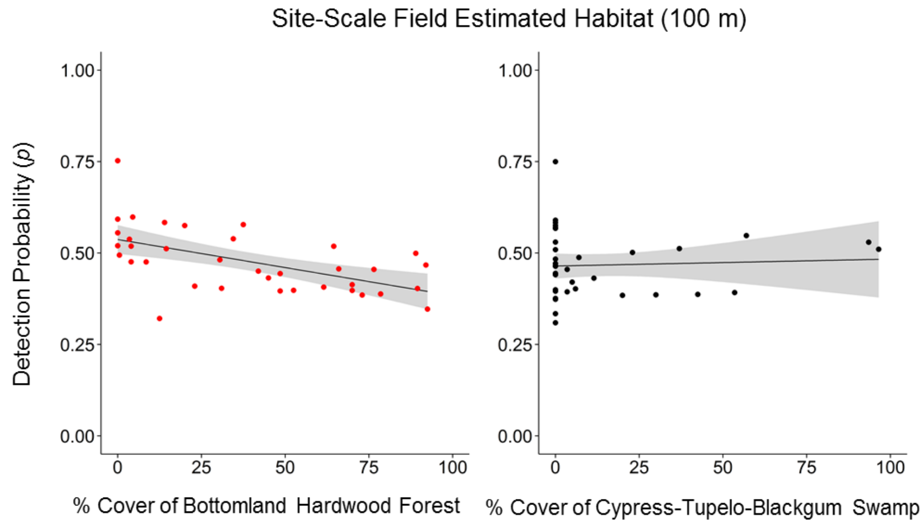

FIGURE 2.19. Intrinsically linear models of detection probability (averaged over the 4 rounds) and percent cover of 100 m field-estimated site-scale habitat classes for 36 sites. Gray areas represent the 95% CI standard error of the 36 estimates. The negative relationship between detection probability and bottomland hardwood forest cover was significant ($R^2 = 0.32$, df = 34, $p < 0.01$). There was no significant relationship between detection probability and cypress-tupelo-blackgum swamp cover ($R^2 = 0.003$, df = 34, $p = 0.75$).

2.3.5 Dynamic (Two Round, Three Survey) Habitat (100 m) and Invertebrate Biomass Analysis

Of the dynamic (two round, three survey) model set of 57 sites, which differed from previous model sets by the inclusion of a dynamic invertebrate biomass covariate, there were three top models that accounted for 49% of the available weight (Table 2.8). The top model showed adequate fit ($\chi^2 = 17.4$, $p = 0.10$, $\hat{c} = 1.49$).

Bottomland hardwood forest was the most important covariate associated with occupancy, although the error was within the range of cypress-tupelo-blackgum swamp, which exhibited a negligible relationship with occupancy (Figure 2.20). The percent cover of bottomland hardwood forest was negatively related to occupancy, but for the first time one of the forested wetland habitat types, swamp, was positively related to occupancy (Figure 2.21).

Original covariate weight estimates ± SE were -0.02 ± 0.01 for bottomland hardwood forest and 0.01 ± 0.02 for cypress-tupelo-blackgum swamp.

TABLE 2.8. Top six habitat association models for 57 sites/2 rounds (100 m field-estimated fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AIC wt	k	n	-2 log likelihood
1	ψ(BLH100),γε(wetlitter+biomass)	344.24	0.00	0.22	11	57	161.12
2	ψ(BLH100),γε(wetlitter)	344.54	0.30	0.19	9	57	163.27
3	ψ(BLH100+swamp100),γε(wetlitter+biomass)	346.15	1.91	0.08	12	57	161.08
4	ψ(swamp100),γε(wetlitter)	346.30	2.05	0.08	9	57	164.15
5	ψ(BLH100+swamp100),γε(wetlitter)	346.40	2.15	0.07	10	57	163.20
6	ψ(swamp100),γε(wetlitter+biomass)	346.98	2.74	0.06	11	57	162.49
29	**ψγεp(null)**	378.24	33.99	0.00	4	57	185.12

* BLH = bottomland hardwood forest
** global model was omitted due to nonconvergence

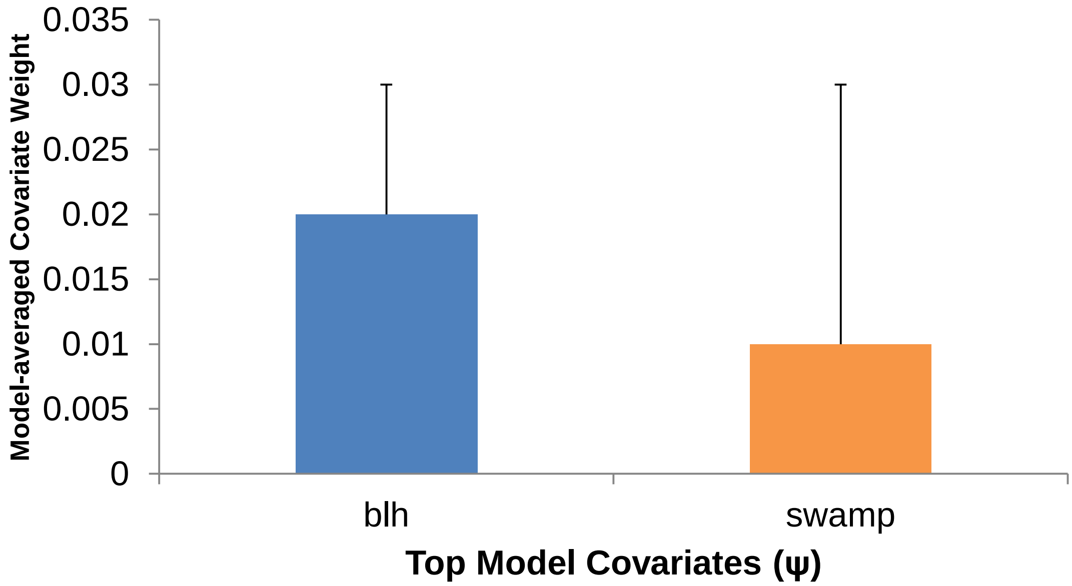

FIGURE 2.20. Absolute values of model averaged covariate weights ± SE for 100 m field-estimated habitat classes associated with occupancy at 57 sites/2 rounds.

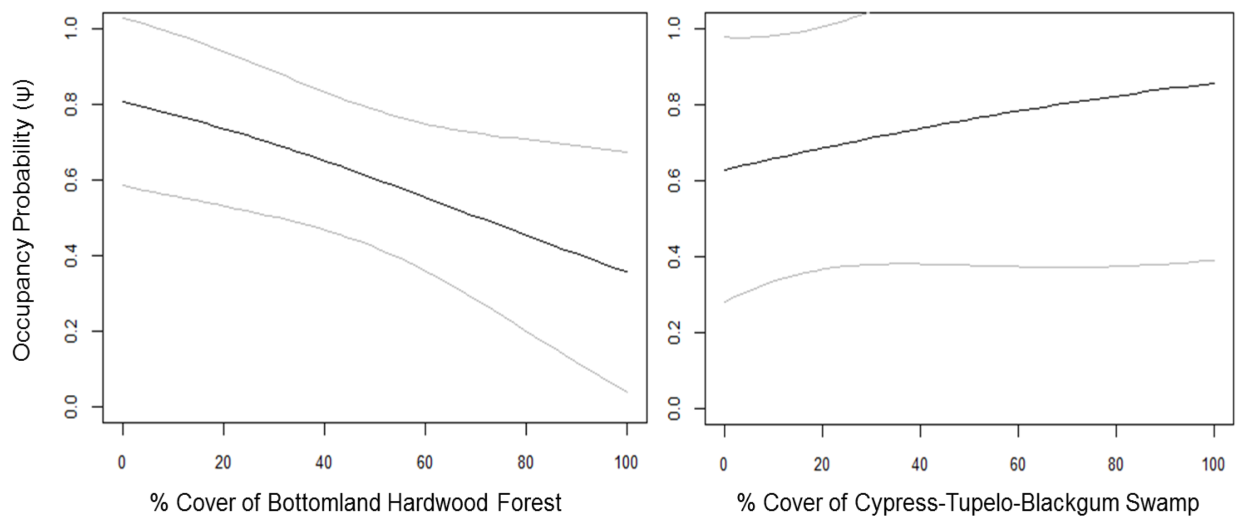

FIGURE 2.21. Predicted occupancy probability with 100 m field-estimated habitat covariates for 57 sites/2 rounds (based on models ψ(BLH100),γε(wetlitter+biomass) and ψ(swamp100),γε(wetlitter), respectively). Gray lines represent 95% confidence intervals of the estimates.

The covariates that most influenced colonization and occupancy probability were wet litter and invertebrate biomass, but the relationships were very weak (Figure 2.22). As expected, colonization was positively related and extinction negatively related to increasing invertebrate biomass and the cover of wet leaf litter (Figures 2.23 and 2.24). Although the magnitude of the estimates for invertebrate biomass were much larger than those for wet leaf litter, the relationship of wet leaf litter cover with colonization probability was the most reliable because the error of the estimate did not overlap zero.

For colonization, original covariate weight estimates ± SE were: wetlitter 0.24 ± 0.21 and invertebrate biomass 14.53 ± 21.54. For extinction, original covariate weight estimates ± SE were wetlitter -0.02 ± 0.05 and invertebrate biomass -10.12 ± 24.43 (Figure 2.22).

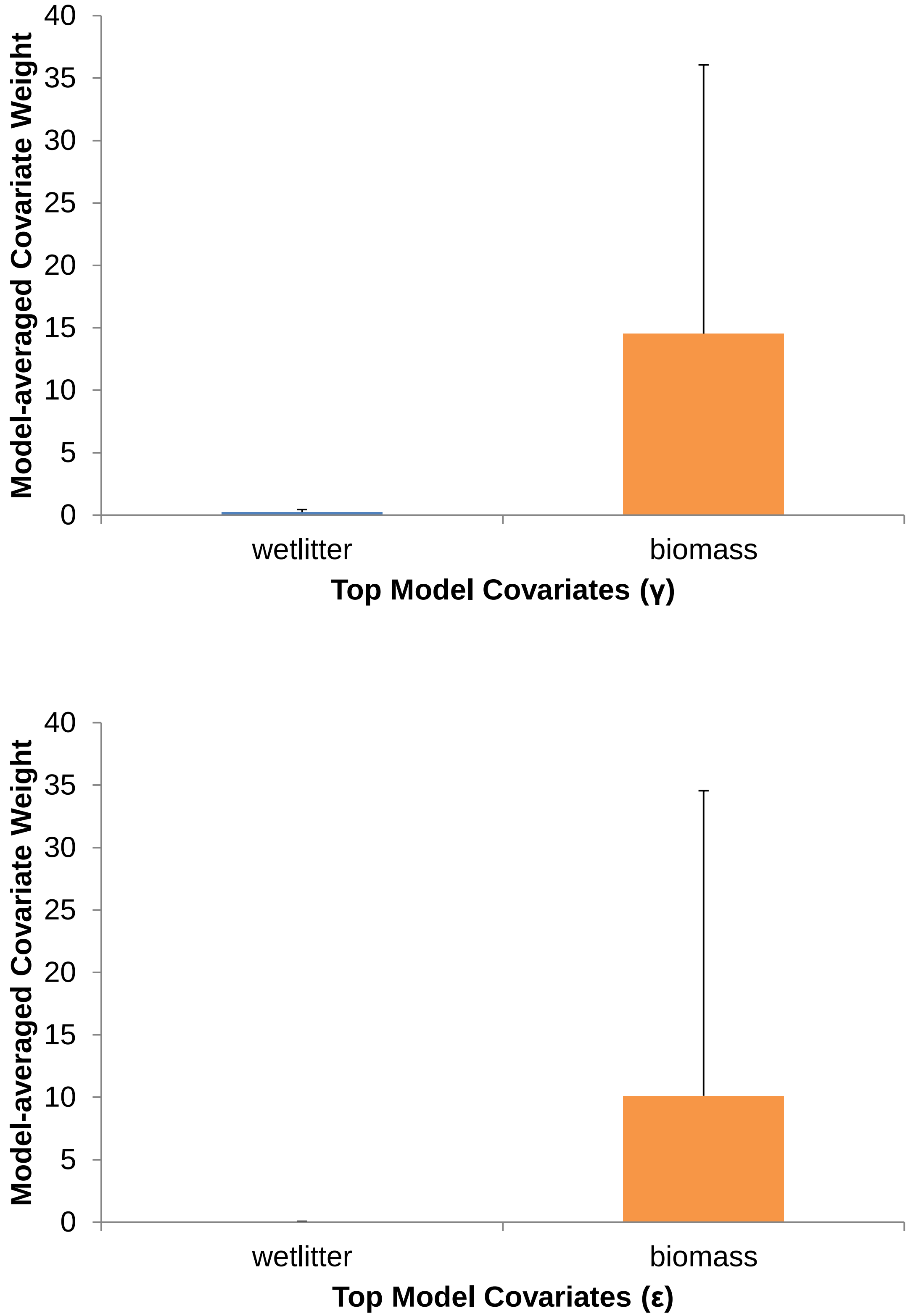

FIGURE 2.22. Absolute values of model averaged covariate weights ± SE for 100 m wet litter cover and invertebrate biomass associated with colonization and extinction at 57 sites/2 rounds.

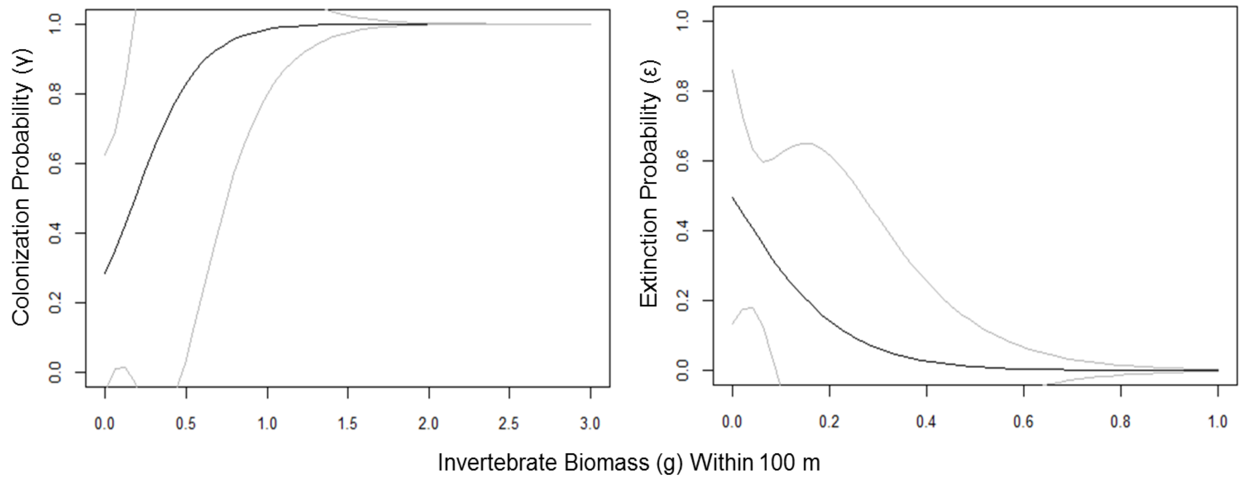

FIGURE 2.23. Predicted colonization and extinction probability with invertebrate biomass (100 m) at 57 sites/2 rounds (based on model ψ(BLH100+swamp100),γε(biomass)). Gray lines represent 95% confidence intervals of the estimates.

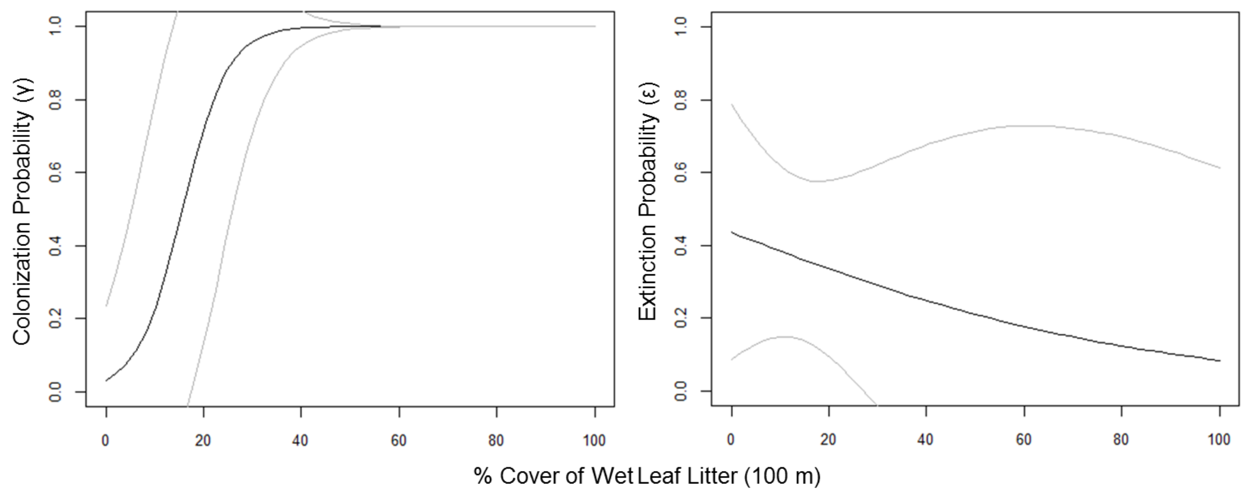

FIGURE 2.24. Predicted colonization and extinction probability with wet leaf litter (100 m) at 57 sites/2 rounds (based on model ψ(BLH100),γε(wetlitter)). Gray lines represent 95% confidence intervals of the estimates.

To determine the habitat covariates associated with presence of larger flocks (≥ 4 birds) of Rusty Blackbirds, I modeled occupancy with habitat for 30 sites/2 seasons. The top model showed adequate fit ($\chi^2 = 14.3$, $p = 0.16$, $\hat{c} = 1.39$). Data were too sparse to model flock occupancy during all four seasons. Again, bottomland hardwood forest and swamp were present in the two top models, which accounted for 75% of the available weight (Table 2.9), although bottomland hardwood forest had the strongest relationship with occupancy (Figure 2.25). Original covariate weight estimates ± SE were bottomland hardwood forest -0.07 ± 0.05 and cypress-tupelo-blackgum swamp -0.03 ± 0.05.

TABLE 2.9. Top three habitat association models abundance-adjusted for flocks (≥ 4 birds) at 30 sites/2 rounds (100 m field-estimated fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time+flock) for all models except the null. Models within Δ2AIC had substantial model support.

rank	**model**	**AIC**	**ΔAIC**	**AICwt**	**k**	**n**	**-2log likelihood**
1	ψ(BLH100)	212.49	0.00	0.31	8	30	98.25
2	ψ(BLH100+swamp100)	214.27	1.78	0.13	9	30	98.14
3	ψ(BLH100),γε(biomass)	215.00	2.51	0.09	10	30	97.50
29	**ψγε(global)**	226.32	13.83	0.00	18	30	95.16
30	**ψγεp(null)**	232.39	19.90	0.00	4	30	112.19

* BLH = bottomland hardwood forest

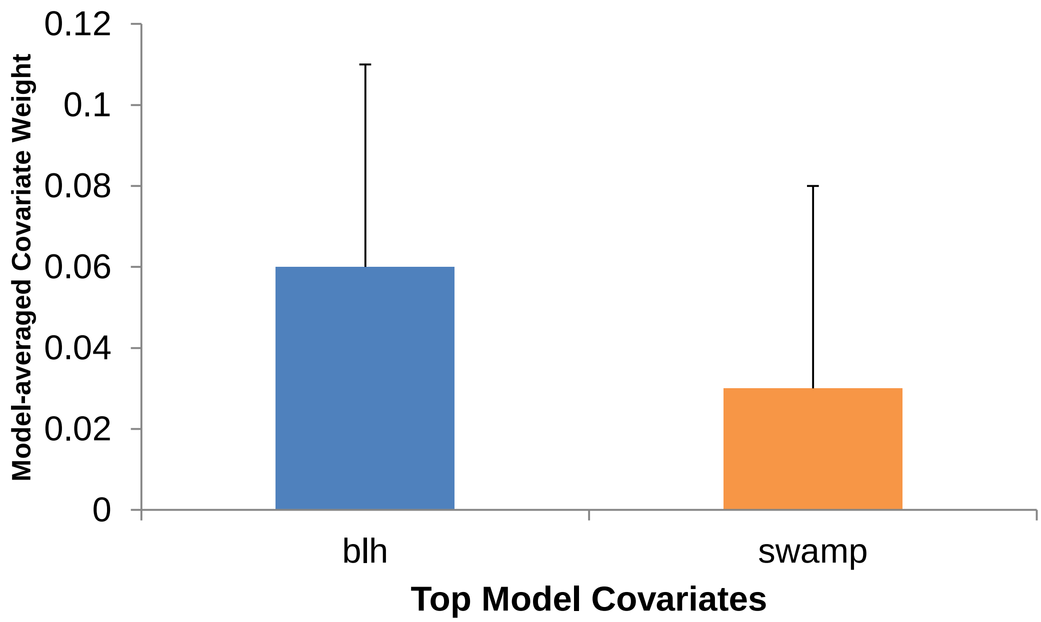

FIGURE 2.25. Model averaged covariate weights ± SE for 100 m field-estimated habitat classes associated with flock (≥ 4 birds) occupancy at 30 sites/2 rounds.

The relationship between occupancy and bottomland hardwood forest at this 100 m scale was again negative, although occupancy probability does not begin to decrease until about 40% cover of bottomland hardwood forest is reached (Figure 2.26). A plot of the bivariate relationship between swamp and occupancy probability yields a neutral relationship with a slightly positive trend (Figure 2.26).

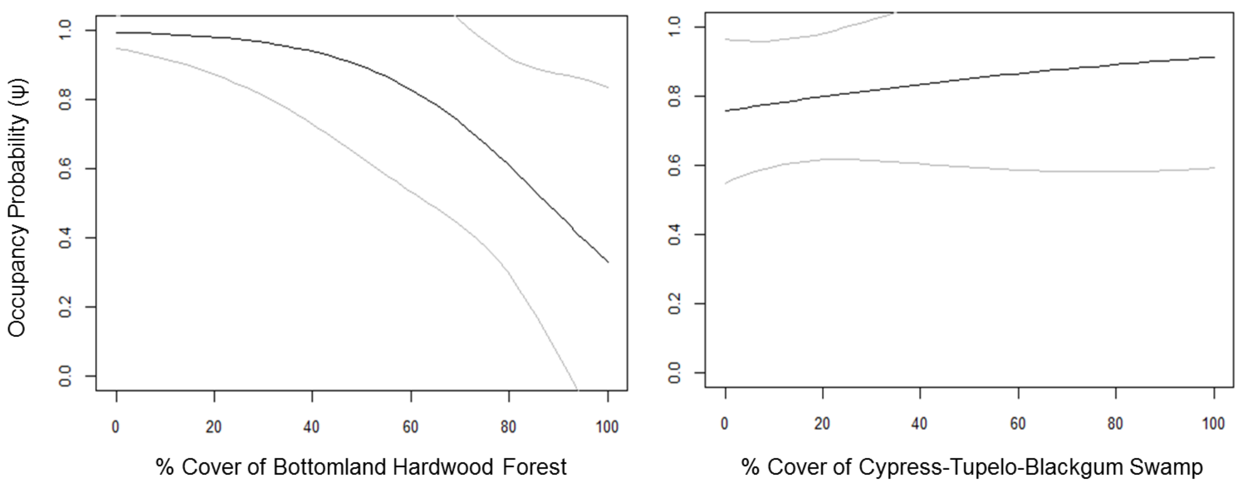

FIGURE 2.26. Abundance-adjusted (≥ 4 birds) predicted occupancy probability with 100 m field-estimated habitat covariates for 30 sites/2 rounds (based on models ψ(BLH100) and ψ(swamp100), respectively). Gray lines represent 95% confidence intervals of the estimates.

Total dry mass was similar between the first (4.45 g) and second (4.69 g) rounds in 2014. Of the total biomass, worms (Annelida; 23%), snails (Gastropoda; 21%), fingernail clams (Bivalvia; 14%), and adult and larval beetles (Coleoptera; 13%) accounted for the greatest percentages (Figure 2.27). When examining the fixed effects of habitat covariates with the random effect of round, two top models accounted for 73% of the model weight (Appendix IV, Table IV.14). The null model received the lowest AIC value, implying that none of my habitat variables adequately explained invertebrate biomass at the 100 m scale. The second top model included the fixed effect of percent cover of lawn within 100 m, which was statistically significant ($\beta = -0.01$, SE $\pm < 0.01$, df = 112, $p < 0.01$).

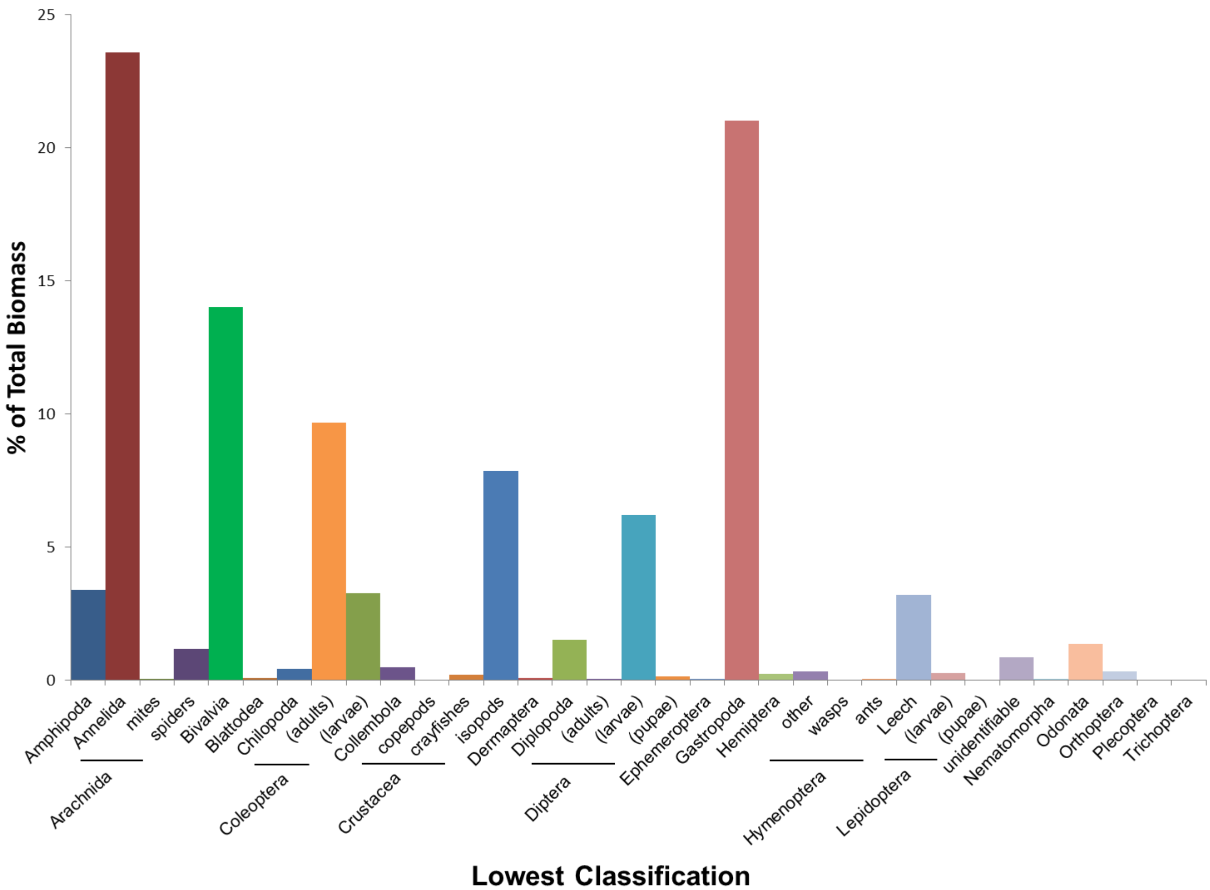

FIGURE 2.27. The composition of invertebrate biomass samples at 57 sites (5 samples/site/2 rounds) within 100 m during the year 2014. Invertebrates are grouped by Order or the lowest classification I could determine.

2.4 Discussion

Habitat Associations

At the 600 m landscape level, forested wetlands most influenced site occupancy. As predicted, floodplain forest/bottomland hardwood forest and woody wetland/cypress-tupelo-blackgum swamp were present in the top models. For the International Rusty Blackbird Working Group habitat classes, floodplain forest was more important than woody wetland, the latter of which had an estimate overlapping zero, indicating a weak effect on occupancy. When examining Louisiana-specific (LCWCS) habitat classifications, the most important variable was soil hydrologic group C, but its confidence intervals overlapped with those of less important variables, including lawn and soils of hydrologic groups C and D.

Despite appearing in the top models, forested wetland types had an unexpected negative relationship with occupancy at the 600 m landscape scale for the analysis of two years and four rounds of data. This is contrary to my initial predictions, as well as findings from Luscier (2009), in which there was a strong positive relationship between occupancy and wet bottomland hardwood forests. I also included a model set based on site scale habitat (100 m) to determine whether overly large scales could be producing a negative relationship, but I found the same relationships with forested wetlands at this smaller scale, with the addition of the soil hydrologic group C appearing in the top model (Table 2.7). The Rusty Blackbird is described as a forested wetland specialist (Avery 1995, Greenberg and Droege 1999) and its distribution in Louisiana seems to correspond to areas of forested wetland cover (Appendix IV, Figure IV.1). However, it is important to note that unlike Luscier (2009), I did not differentiate between wet and dry types of forest as occupancy covariates. Rusty Blackbirds require shallow water and wet substrates to forage in, thus it is possible that forested wetlands could be unsuitable if they lacked water or had too much deep water (DeLeon 2012). I modeled habitat occupancy with colonization and extinction covariates related to water (shallow, wet litter, wet grass, and rain) that could be indicative of wetness levels at my sites. If I had estimated moisture within habitat types, such as in Luscier's (2009) study, it is possible that there could have been a positive relationship between forested wetlands and occupancy at my sites.

Several scenarios could explain a negative relationship between forested wetlands and occupancy. Potentially, the forested wetlands may have been too dry, too wet, or too fragmented. It is possible that it is difficult to detect Rusty Blackbirds during dry periods or years of reduced flooding because of the patchiness of water on the landscape. Rusty Blackbirds have large average winter foraging ranges (5.08 km^2; Newell 2013, unpublished data analyzed by Borchert) and if water is unavailable at my survey sites, I may not be detecting them simply because they have traveled farther away to find it.

I also had several sites that had fragments of forested wetland embedded within a larger urban or agricultural landscape. Even though most of the surrounding landscape could be unsuitable for Rusty Blackbirds at these sites, I may still have reliably detected Rusty Blackbirds because they were concentrated in the only suitable habitat available. When I examined the relationship between detection probabilities with the top forested wetland habitat covariates, I found that detection probability decreases with increasing cover of floodplain forest (600 m scale) and bottomland hardwood forest (100 m scale), providing some support for these hypotheses. Rusty Blackbirds are more likely to be detected if there are Common Grackles present; I detected larger grackle flocks at low cover of bottomland hardwood forest, which may have lowered Rusty Blackbird detection probability at high cover. Although the relationship was significant, detection probability does not decrease very much between low cover and high cover sites (Figures 2.17, 2.18, 2.19). Additionally, there was no trend for detectability in bottomland hardwood forest (600 m scale), woody wetland (600 m scale), or cypress-tupelo-blackgum swamp (100 m and 600 m scales). With woody wetland and swamp, there was a neutral or even increasing trend with detection probability.

Forested wetlands could also be unsuitable for occupancy if there is too much deep water at a site, which is unusable for Rusty Blackbirds due to their inability to wade in deep water. To explore this possibility, I tested the relationship between percent cover of deep water and forested wetlands during each of the four rounds. With increasing cover of floodplain forest or woody wetland there was increasing cover of deep water, which was significant for all rounds except for the third in the second year (Figure 2.14). Likewise, this relationship was significant for bottomland hardwood forest and cypress-tupelo-blackgum swamp during all rounds (Figure 2.15). These results suggest that sites with high coverages of forested wetlands, which would presumably be preferred by a forested wetland specialist, could be unsuitable due to deep water constraining the usable foraging substrate. Supporting this trend, Luscier (2009) found that occupancy increased by 35% in bottomland hardwood greentree reservoirs when water levels were drawn down from 2.24 m to 1.14 m, likely because more foraging habitat was made available. Interannual temporal variability in flooding may influence Rusty Blackbird use of these habitats.

I thought it possible that Rusty Blackbirds would use an area as long as it could support shallow water, regardless of habitat or substrate. Soils with a better ability to retain surface water may be more important for Rusty Blackbirds, thus I classified soils by groups that describe their ability to transmit water. I hypothesized that soil hydrologic groups C and D would be the most important because they had the slowest rates of water transmission, but group C may be more important because its rate of water transmission would be slow enough to contribute to ponding but not deep water. Group D, which is associated with high water tables and clay soils, could be associated with deeply flooded areas that are less usable. As predicted, at the 600 m and 100 m scale, soil hydrologic group C appeared in the top model and was positively related to Rusty Blackbird occupancy. Soil hydrologic group D also appeared in the top models, but had a weakly negative relationship with occupancy. These findings may provide some indirect support that while surface water is important, areas that are deeply flooded for longer durations are not preferable to Rusty Blackbirds.

Wet leaf litter was the most important variable for site transience. Between rounds, a site was more likely to be colonized with increasing cover of wet leaf litter and more likely to go extinct with decreasing cover of wet leaf litter. However, the magnitude of the covariate weights for colonization and extinction were small and the confidence intervals overlapped zero for extinction, indicating weaker relationships. Wet leaf litter has been previously shown to be important for occupancy, particularly for larger flocks (DeLeon 2012). Although I observed negative relationships with forested wetlands and occupancy at the 600 m scale, local site conditions such as increased wet leaf litter could be making these sites important at different times within a season. The evidence that wet leaf litter contributes to colonization or abandonment of a site implies that flooded forest must be important by association, since leaf litter quantity is likely increasing with greater numbers of trees. In comparisons between bottomland hardwood forest and cypress-tupelo swamp plots, Conner and Day (1976) did not demonstrate any differences in leaf litter quantity. There may not be a difference in quantity of litter between the two habitat types, except in swamp where deep water or saline water intrusion has caused tree die-offs. However, leaf litter may decompose quickly in bottomland hardwood forest (Conner and Day 1976), which increases its nutritive value for invertebrates (Cummins 1973, Suberkropp et al. 1983). If litter in bottomland hardwood forests is of higher quality for invertebrates, and thus Rusty Blackbirds, that would help explain why bottomland hardwood forest was the most important habitat variable in most analyses.

I constructed a separate model set for the second year of my study because I incorporated a measure of invertebrate biomass specific to the substrates Rusty Blackbirds would be foraging in. Since invertebrates can form aggregations, I only examined the relationship between invertebrate biomass within 100 m and habitat at the same scale. Similarly to my landscape scale results, bottomland hardwood forest and cypress-tupelo-blackgum swamp appeared in the top models. Bottomland hardwood forest was again the most important covariate at this scale and displayed a negative relationship with Rusty Blackbird occupancy probability. The magnitude of the estimate for swamp was weak; however, there was a positive relationship between swamp and occupancy for this model set. Interestingly, when examining the abundance-adjusted occupancy set, the top models included bottomland hardwood and swamp but did not include any dynamic covariates associated with colonization or extinction. For large flocks, bottomland hardwood forest seems to be the most important, regardless of any dynamic covariates. The fitted values from the top model predicted that flocks ($\geq$ 4 birds) occupy bottomland hardwood forest at about 100% until around 40% cover of bottomland hardwood forest, when occupancy begins to decrease. At higher coverages of bottomland hardwood forest there was a trend of increasing deep water cover, which may be reducing Rusty Blackbird occupancy past certain levels. Although the estimate for swamp was not significantly different from zero, it was positive; therefore it could still be an important habitat.

Naïve occupancy was different in sites composed of primarily swamp versus bottomland hardwood forest for 100 m sites in the second year. However, teasing apart the effects of both separately is difficult because many sites were a mix of the two forested wetland types. For sites that had greater than 50 percent cover of the forested wetland of interest and less than 10 percent cover of the other type, average naïve occupancy in swamp over two rounds was 60% (n = 5), while it was 36% in bottomland hardwood forest (n = 18). If possible, future studies should incorporate more sites in cypress-tupelo-blackgum swamp to better investigate the relative importance of these habitats for Rusty Blackbirds. I used Rusty Blackbird locations provided by birders to establish my sites. The lack of swamp sites in this study likely reflects the inaccessibility and less appealing nature of flooded swamps for birding, rather than lack of use by the birds.

For the dynamic (two round, three survey) model set, the most important dynamic covariate was again wet leaf litter, as well as the additional invertebrate biomass covariate. The probability of a Rusty Blackbird colonizing a site increased with wet leaf litter and invertebrate abundance; conversely, the probability of a site going extinct increased with decreasing cover of wet leaf litter and biomass of invertebrates. However, these relationships were not strong because the standard errors of the covariate weights overlapped each other, as well as zero. For two of the top models, the combination of wet leaf litter and biomass had the greatest effect on movement into or out of a site. My sampling in primarily wet leaf litter likely skewed the relationship of invertebrate biomass towards leaf litter. However, leaf litter in forested wetlands is incredibly important for macroinvertebrate detritivores as a primary food source (Fredrickson and Batema 1992), thus wet leaf litter is likely important for Rusty Blackbirds in the context of providing a substrate for the invertebrates that they feed on.

At my sites, worms (Annelida), snails (Gastropoda), fingernail clams (Bivalvia), and beetle adults and larvae (Coleoptera) accounted for most of the invertebrate biomass. Rusty Blackbirds have been observed eating all of these groups (DeLeon 2012, Newell 2013); therefore it is likely that Rusty Blackbirds could consume the majority of the represented invertebrate biomass at my sites. The top model for invertebrate biomass as a response variable had no habitat covariates (null), therefore none of my ground cover or habitat variables explained invertebrate biomass well. The second top model, lawn cover, had a negative relationship with invertebrate biomass at my sites. The negative relationship could be due to invertebrate biomass truly being low in sites with high lawn cover (Pratt et al. 1981, Rogers et al. 2002), which is reflected in my occupancy models as lawn was not one of the most important covariates. However, my sampling in primarily wet leaf litter and shallow water, which could be sparser in lawns, may have biased my ability to draw meaningful inferences between habitat, ground cover, and invertebrate biomass at my sites.

Detectability

Time of day and the number of Common Grackles recorded during the point count were the most important covariates influencing Rusty Blackbird detection probability in both the 4 round/36 site and 2 round/57 site analyses. The confidence intervals of the estimates did not overlap, suggesting that both covariates are important. Time of day had a much stronger effect, with the probability of detection decreasing later in the day. Although detections supposedly decrease with time, the confidence intervals are wide throughout the day, indicating that there may not be a great difference in detection probability between morning and afternoon. My results suggest that studies should survey earlier in the day to maximize detections, if possible, but this is not a steadfast rule.

The number of Common Grackles had a much weaker effect on detectability than time, but Rusty Blackbirds were more likely to be detected with increasing numbers of Common Grackles. Particularly, predicted detection probability of Rusty Blackbirds increases to 100% after ten Common Grackles are detected. Luscier (2009) found that the probability of co-occurrence of Rusty Blackbirds and Common Grackles was greater than the probability of either occurring alone, suggesting that either Rusty Blackbirds or Common Grackles could be benefiting from flocking together. However, while DeLeon (2012) found that detectability of Rusty Blackbirds was conditional on Common Grackles, occupancy was only conditional on Red-winged Blackbirds.

Common Grackles have a diet high in acorns (Meanley 1972) and may be acting as intermediaries that enable Rusty Blackbirds to scavenge on the smaller mast pieces discarded by grackles (Luscier 2009, Newell 2013, S. Borchert personal observation). For many birds, flocking behavior can help decrease predation rates because there are more sentinels available to scan for predators. Rusty Blackbirds could historically be seen in large flocks, but with the dramatic decline in their numbers they may be seeking refuge in larger Common Grackle or mixed flocks to avoid the increased predation rates associated with smaller groups. However, this could also be forcing them into competition with these other blackbird species (Greenberg and Matsuoka 2010). When I had detections of large (> 300) Rusty Blackbird flocks, they were generally monospecific, with only small numbers of individuals of other blackbird species present. In areas where Rusty Blackbird concentrations are high, they may prefer to forage with other Rusty Blackbirds, but in areas of low concentrations they may be forced to flock with Common Grackles due to the lack of conspecifics.

Conclusions/Management Recommendations

In this study, forested wetland types were important variables that determined Rusty Blackbird occupancy, but my results suggest that they are only important under certain conditions. Rusty Blackbirds were more likely to occupy a site with increasing bottomland hardwood forest and colonize a site if there was wet leaf litter to forage on, which implies that they need forests to be moist. Higher wet leaf litter cover indicates that forested wetlands should be shallow enough to support high amounts of litter on the forest floor. There is a possible linkage between high invertebrate biomass at sites and wet leaf litter, but because my sampling was subjective and the covariate weights were weak, I cannot draw any conclusions related to these variables. High cover of forested wetlands was significantly related to high cover of deep water that Rusty Blackbirds are unlikely to use, which suggests that negative occupancy rates in these habitats could have been driven by deep water.

There are two major forested wetland systems in the southeast with separate management challenges: bottomland hardwood forests and cypress-tupelo-blackgum swamps. Hydrological alterations have affected the frequency, duration, and sources of floodwaters in these landscapes. Upstream bottomland hardwood forests tend to be drier as a result of channelization (Shankman 1997). Levees have disconnected the floodplain from freshwater inputs and in the absence of floodwaters, communities can transition from flood tolerant oaks to flood intolerant tree species over time (Gee et al. 2014), which could lead to a decrease in oak mast availability for Rusty Blackbirds. Additionally, macroinvertebrate densities and diversity can decrease in response to flood control measures (Kennedy and Turner 2011). Future research could address differences in Rusty Blackbird occupancy and invertebrate biomass in flood controlled systems versus systems that have not been extensively altered.

Coastal baldcypress-tupelo swamp faces similar challenges. Flood control affects these systems by preventing the flow of water and deposition of sediment so that natural rates of land subsidence are no longer being balanced out by accretion. Swamps tend to deepen over time or convert to open water or marsh in the absence of freshwater inputs and also due to saltwater intrusion (Chambers et al. 2005, Shaffer et al. 2009), which would make these habitats unsuitable for Rusty Blackbirds. Freshwater diversions could be beneficial if they contribute to shallower swamps and help prevent tree die-offs (Shaffer et al. 2003). A before and after diversion comparison of Rusty Blackbird occupancy in these areas could help determine the effectiveness of diversions in increasing habitat quality.

Programs such as the Conservation Reserve Program and the expired Wetland Reserve Program, which provided financial incentives to private landowners for restoring wetlands and retiring their eligible lands from agriculture, can help reclaim lost forested wetlands. However, hydrological alterations seem to be changing these landscapes dramatically and combining hydrological restoration with reclaimed land is imperative (King et al. 2006). Most wetland management has been targeted at waterfowl in the form of moist-soil impoundments (King et al. 2006). Green tree reservoirs, which are impounded areas of bottomland hardwood forest, could be valuable tools for Rusty Blackbird management if they are not flooded for extended time periods and the water is not too deep (Luscier 2009). Forested wetland invertebrates make up the largest proportion (an average of 73.5%) of wintering Rusty Blackbird diets (Newell 2013). These invertebrates are adapted to fluctuating water regimes; short-term flooding increases their abundance and biomass as they respond to nutrient release from flooded leaf litter (Fredrickson and Batema 1992). Fluctuating forested wetland water levels to maximize invertebrate production, keeping water levels low to increase foraging substrate, and maintaining trees on the wintering ground could be key for improving habitat for Rusty Blackbirds.

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CHAPTER 3. USING CITIZEN SCIENCE DATA FROM THE LOUISIANA WINTER BIRD ATLAS TO IDENTIFY LANDSCAPE-LEVEL HABITAT ASSOCIATIONS OF RUSTY BLACKBIRDS

3.1 Introduction

Over the last century, large-scale citizen science programs have successfully utilized public participation to collect ornithological data, including Audubon's long-running Christmas Bird Count, the more recently developed Cornell Lab of Ornithology's eBird (Sullivan et al. 2009), and various local and state-wide atlases. The data collected can be used to examine broad ecological patterns for species and communities, including trends in relative abundance, distribution, migratory timing, survival, and reproductive success (Dickinson et al. 2010). One major benefit of these programs, in contrast to small studies, is their ability to provide data on rare or declining species due to the high volume of data submitted by many volunteers (Dickinson et al. 2010).

Rusty Blackbirds (*Euphagus carolinus*) have had the sharpest decline of any North American landbird, with the fastest rate of decline occurring between the 1950s and 1970s (Greenberg and Droege 1999). Compared to other songbirds, the difficulty of monitoring them is compounded by their rarity, inconspicuous behavior, and the inaccessibility of the forested wetlands they use (Greenberg and Matsuoka 2010). Citizen science data were used to determine the cumulative 95% magnitude of the decline (Greenberg and Droege 1999, Niven et al. 2004, Sauer et al. 2005). Until we can identify the factors affecting the population at each stage of the annual cycle, the major causes of the decline will remain unknown, but wintering ground forested wetland loss coupled with hydrological alterations could play a large role (Greenberg et al. 2011, DeLeon 2012). Compared to the boreal breeding ground, southeastern forested wetlands have been cleared at a much faster rate, with 75 – 80% of bottomland hardwoods being converted to agriculture since European settlement (Hefner and Brown 1984, Hefner et al. 1994, Twedt and Loesch 1999). During the 1970s, clearing of southeastern wetland occurred at an even higher rate and leveled off as some land was reclaimed through programs such as the Conservation Reserve and Wetland Reserve Program easements given by the U.S. Department of Agriculture (King et al. 2006). These patterns of forested wetland change appear to roughly follow patterns in Rusty Blackbird Christmas Bird Count data from the same time period, providing support for wintering habitat change's role in the decline (Greenberg and Droege 1999, Hamel et al. 2009, Greenberg et al. 2011, DeLeon 2012).

The value of citizen science programs for identifying population changes for the Rusty Blackbird and other birds is undeniable, but given the large coverage of these programs, they can also be used to identify large-scale habitat associations. Previous research of Rusty Blackbirds has mainly focused on smaller scales, including site-specific habitat at 25 m and 100 m scales (DeLeon 2012) and the 11.3 m scale (Luscier 2009, Luscier et al. 2010). However, larger scales may be important because Rusty Blackbirds can be highly dispersed, fly long distances over short time periods, and have large winter foraging ranges (5.08 km^2; Newell (2013), unpublished data analyzed by Borchert). Additionally, these studies chose survey sites based on where Rusty Blackbirds were either previously observed or expected to be observed (Luscier 2009, Luscier et al. 2010, DeLeon 2012, Borchert Chapter 2). The drawback of studies that localize surveys to areas of known occurrence is that they may lack a true comparison between suitable and unsuitable habitats. To make inferences about habitat selection, studies need to determine habitat use relative to availability where possible (Johnson 1980); in practice, this has been difficult to apply for a rare species like the Rusty Blackbird.

I combined abundance data from the Louisiana Winter Bird Atlas (hereafter LWBA) with broad scale environmental data to determine Rusty Blackbird landscape habitat associations on the wintering ground. Citizen science data blend well with landscape ecology studies because biological and environmental data can be paired to understand how spatial and temporal patterns affect ecological processes (Zuckerberg and McGarigal 2012). In addition to habitat, I included rainfall data because precipitation could be increasing the appeal of certain habitats (Newell 2013) and influencing their movements within a winter (Hamel and Ozdenerol 2009). The LWBA provided high quantities of Rusty Blackbird data not possible in traditional field studies and also allowed me to measure distribution across the state, instead of being limited to areas where birds were previously seen. My objectives were to 1) determine Rusty Blackbird landscape-scale habitat associations over the state of Louisiana using spatial land cover, crop land cover, and soils datasets and 2) determine whether Rusty Blackbird abundance was correlated with variation in annual winter rainfall.

3.2 Methods

3.2.1 Rusty Blackbird Count Data from the Louisiana Winter Bird Atlas

The LWBA was conducted from January 10^{th} to February 20^{th} over a period of eight years from 2007 to 2014 (Remsen et al. 2012). Participants surveyed 7.5 minute U.S. Geological Survey quadrangles, which have an area of about 160 km^2, varying with latitude. The goal of the LWBA was to survey as many quadrangles as possible and accumulate a minimum of ten hours of effort per quadrangle, although this was not possible for all quadrangles. I was able to use data from 512 unique quadrangles (969 quadrangles surveyed over multiple years). Participants went out for a minimum of one hour and recorded the numbers of birds seen or heard, distance traveled, and the areas they covered. Effort was quantified as the number of party hours spent birding, with a party defined as one or more observers traveling together and detecting the same birds. To maximize detectability, observers avoided surveying in bad weather and began surveys as early as possible. Surveyors attempted to cover the area of the quad evenly and avoided repeatedly counting from a single location to avoid skewing the data towards habitats that may not be representative of the bird distribution in the entire quad. To minimize biases, I discarded count data from any survey that was less than 1 party-hour or if a survey was conducted at night.

3.2.2 Spatial Data Geoprocessing

To determine the habitat composition of each quadrangle, I used 30 m resolution land cover data from the National Gap Analysis Program (U.S. Geological Survey 2011), which incorporates the ecological classification system developed by NatureServe (NatureServe 2011). Within the classification system there are 590 land use classes which were identified by using dominant vegetation type, but also incorporated digital data on soils, topography, hydrology, and climate. I reclassed land cover types according to recommendations provided by members of the International Rusty Blackbird Working Group (IRBWG) to make them more biologically relevant for the Rusty Blackbird (Appendix IV, Table IV.3). To make Louisiana-specific management recommendations, I also reclassed land cover types based on the habitats outlined in the Louisiana Comprehensive Wildlife Conservation Strategy (Lester et al. 2005; LCWCS, Appendix IV, Table IV.3). Because I used land cover data that were last updated in 2011, I assumed that the degree of land cover change over the eight years of the LWBA was not significant enough to affect the number of birds observed in each quadrangle.

Pecans are an important resource for Rusty Blackbirds in winter (Newell 2013, Mettke-Hofmann et al. 2015); for this reason, I also used the National Agricultural Statistics Service (NASS) Cropland Data Later (CDL) to incorporate commercial pecan orchard cover (U.S.

Department of Agriculture 2014). Cropland data is produced annually, but a pecan class was not available prior to 2010. I used the 2010 CDL for quadrangles surveyed in 2010 and earlier. For quadrangles surveyed from 2011-2014, I chose the CDL of the same year and if a quadrangle was visited in multiple years I used the earliest CDL available.

In addition to vegetation communities, I was also interested in soils that were more likely to maintain shallow water on their surface. I combined the 315 soil series present in Louisiana into their associated hydrologic groupings, which describe the ability of water to transmit through a soil (Appendix IV, Table IV.4). I used 10 m resolution Natural Resources Conservation Service (NRCS) Soil Survey Geographic Database (SURGO) raster data to obtain soil hydrologic group coverages within a quadrangle (Soil Survey Staff 2014). I was only interested in soil hydrologic groups that could maintain surface water for foraging Rusty Blackbirds; soil hydrologic group C has a slow rate of water transmission, group D has a very slow rate of water transmission, and group C/D displays group C characteristics in drained areas and group D characteristics in undrained areas (U.S. Department of Agriculture 2009; Appendix IV, Table IV.4)

After reclassification I projected all raster layers to NAD83 UTM 15N before geoprocessing in ArcGIS 10.2 (ESRI 2013). I transformed the CDL from the WGS84 datum to the NAD83 datum before projection to match the other raster datasets. To insert pecan pixels from the CDL into the GAP data layer, I snapped the rasters together to align the processing extent and used the “Con” function in the Spatial Analyst toolbox. In an area where a pecan pixel from the CDL occurred, it replaced the overlapping GAP pixel and created a new raster dataset. Once the land cover rasters were combined I used “Spatial Analyst: Extract by Mask” to extract the land cover within each quadrangle, then calculated the percent cover of each land cover class from the number of pixels within each quadrangle. Repeating this process with the soils data, I calculated the percent cover of each soil hydrologic group within quadrangles.

To determine the total amount of rainfall over the survey periods during each year, I obtained monthly normal precipitation data for December – February of each survey season from Oregon State University's PRISM Climate Group (PRISM Climate Group 2014). The Parameter-elevation Relationships on Independent Slopes Model (PRISM) takes into account spatial climate patterns and adjusts precipitation in each grid cell (pixel) using its location, elevation, coastal proximity, topographic facet orientation, vertical atmospheric layer, topographic position, and the orographic effectiveness of the terrain (Daly et al. 2008). January and February precipitation data corresponded to the dates of the LWBA period, but I also included December rainfall because rainfall in the month prior could be contributing to the amount of standing water within quadrangles. I summed the rainfall values for each pixel by quadrangle for the three months (December of previous year – February of the LWBA year) to obtain the total precipitation for each survey season. The PRISM model only estimates precipitation over land and bodies of freshwater; for quadrangles partially positioned over the ocean I averaged the pixels with data to fill in values for the pixels missing data.

3.2.3 Data Selection and Statistical Analysis

I analyzed data from individual quadrangles with at least five hours of survey effort (party-hours) each year because I assumed five hours was enough time to adequately cover a quadrangle and detect a Rusty Blackbird. To account for varying effort, which would bias abundance estimates, I divided the number of Rusty Blackbirds observed in a quadrangle by the total amount of effort per year to obtain a rate (Rusty Blackbirds/party-hour, hereafter

RUBL/party-hr). Prior to analyzing the data, I removed extreme counts that were greater than three standard deviations from the mean count rate of each year (Pukelsheim 1994).

I used generalized linear mixed models (GLMMs) to relate Rusty Blackbird count rate data to fixed landscape covariates and random effects (year and rainfall nested within year) using PROC GLIMMIX for SAS Software (SAS Institute Inc. 2013). GLMMs are useful because they allow the response variable to have a distribution within the exponential family of distributions (e.g., normal, Poisson, negative binomial; Faraway 2006). I modeled Rusty Blackbird count rate data with a Poisson distribution and a log link, the latter of which allows the function of the response variable to vary linearly with predicted values, rather than the response variable itself. By including the random variables, year and rainfall nested within year, and specifying the variance-covariance matrix, these models can also account for inherent time correlations of the response variable in each quadrangle (Gbur et al. 2012), which was an issue because I had count rates for eight consecutive years of data. Sampling was imbalanced because quadrangles were not surveyed every year or even in multiple years. I ran models with a variety of different covariance structures before choosing the best-fit variance, by lowest AIC, variance-covariance (variance components) matrix specification, which allowed me to model a different variance component for each random effect (year and rainfall within year).

When building *a priori* models for Rusty Blackbird abundance, I only used habitat variables that I considered important for the birds based on my own observations and the literature. I had two candidate sets, one based on the IRBWG habitat reclassifications and another Louisiana-specific set based on the habitats outlined in the LCWCS (Lester et al. 2005). Prior to constructing all models, I tested variables for collinearity with other variables (Spearman Rank Correlation Test $|\rho| \geq 0.5$). The only correlations were between variables (e.g. floodplain forest and bottomland hardwood forest) used in different candidate sets, thus, multicollinearity was not an apparent issue. However, for the model set based on LCWCS habitat classifications I

TABLE 3.1. Covariates used to identify fixed and random effects in models of Rusty Blackbird abundance data from the Louisiana Winter Bird Atlas. One model set was based on International Rusty Blackbird Working Group land cover classifications and a different set was based on Louisiana-specific Comprehensive Wildlife Conservation Strategy (Lester et al. 2005) land cover classifications.

covariate	description	correlations
random		
year	year quadrangle was surveyed (2007-2014)	none
rain(year)*	total rainfall Dec-Feb (for each LWBA period from 2007-2014) nested within year	yes, with one or more LCWCS variables
fixed		
International Rusty Blackbird Working Group Landscape Cover		
floodplain forest	% cover of floodplain forest	none
woody wetland	% cover of woody wetland	none
developed	% cover of developed land (grassy areas, pavement, buildings, etc.)	none
pecan orchard	% cover of pecan orchards	none
soil C	% cover of soil hydrologic group C	none
soil D	% cover of soil hydrologic group D	none
soil C/D	% cover of dual soil hydrologic group C/D	none
Louisiana Comprehensive Wildlife Conservation Strategy Landscape Cover		
bottomland hardwood forest	% cover of bottomland hardwood forest	none
cypress-tupelo-blackgum swamp	% cover of cypress-tupelo-blackgum swamp	none
lawn	% cover of low intensity development (lawn)	none
pecan orchard	% cover of pecan orchards	none
soil C	% cover of soil hydrologic group C	none
soil D	% cover of soil hydrologic group D	none
soil C/D	% cover of dual soil hydrologic group C/D	none

* Only included in International Rusty Blackbird Working Group habitat class model set.

was unable to include rainfall as a random effect because there was collinearity between rain and one or more explanatory variables, which became apparent only after including multiple variables in the same model. I removed the effect of rain from that analysis because land and soil cover variables are likely to be more important for Rusty Blackbirds than annual variation in rainfall.

I used Laplace approximation to determine the log likelihood of each model, which later allowed me to perform likelihood ratio tests among models and compute likelihood based fit statistics (Schabenberger 2007, Lumley and Scott 2015). I then compared models to each other; top models had the lowest Akaike Information Criteria (AICc) and I considered models within $\Delta 2$ AICc to have the most support (Burnham and Anderson 2002). AICc is a bias correction term that accounts for small sample sizes when there are a large number of estimated parameters (Burnham and Anderson 2002).

I compared the relationship between annual cumulative RUBL/party-hr and 3 month total rainfall in surveyed quadrangles by cross-correlation using PROC ARIMA for SAS Software (SAS Institute Inc. 2013). Cross-correlation accounts for the serially correlated errors inherent to the measurement of birds and precipitation over time (Cryer and Chan 2010).

3.3. Results

Of 512 unique quadrangles included in the analysis, 201 (39%) had at least one Rusty Blackbird detected during one of the eight years (Figure 3.1). I analyzed 969 quadrangles total, which included instances where quadrangles were surveyed during multiple years. Excluding outliers, the cumulative abundance per quadrangle ranged from 0 – 23.3 RUBL/party-hr. The highest abundances by year in an individual quadrangle included 3.8 RUBL/party-hr in 2007, 4.7 RUBL/party-hr in 2008, 11 RUBL/party-hr in 2009, 9 RUBL/party-hr in 2010, 17.6

RUBL/party-hr in 2011, 19.8 RUBL/party-hr in 2012, 8.4 RUBL/party-hr in 2013, and 10.4 RUBL/party-hr in 2014. Cumulatively, the year 2011 had the most Rusty Blackbirds/party-hr (207 RUBL/party-hr/year) and also had the highest proportion (39%) of surveyed quadrangles with Rusty Blackbirds (Figure 3.3).

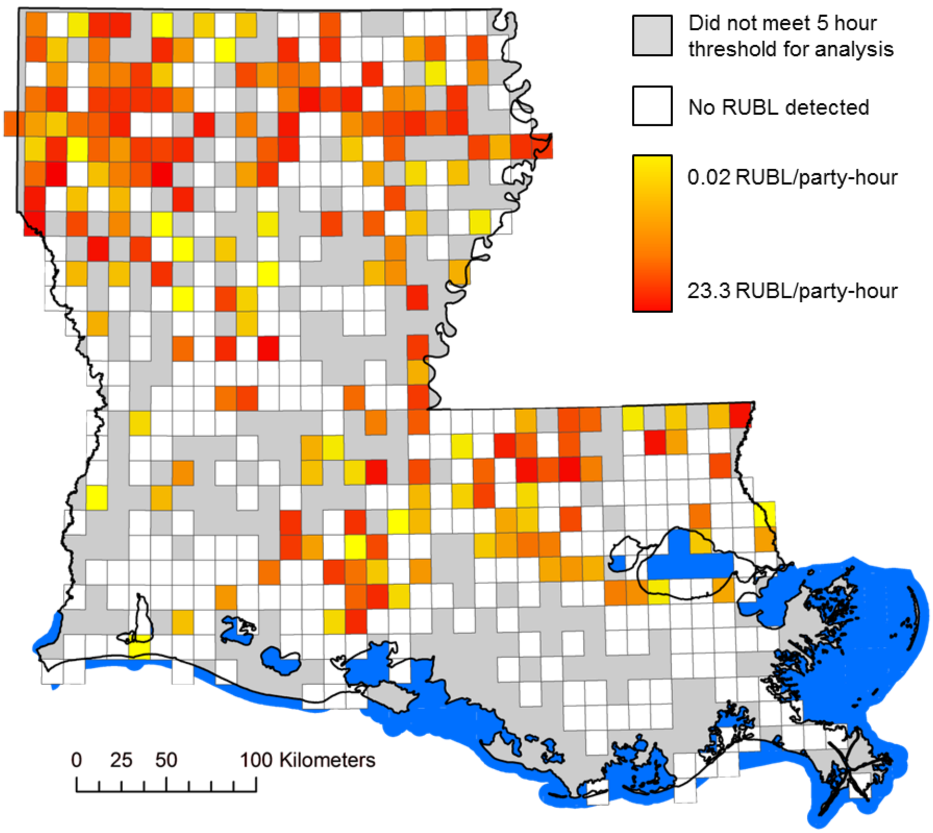

FIGURE 3.1. Cumulative Rusty Blackbirds/party-hour over the eight years (2007-2014) of the Louisiana Winter Bird Atlas. Gray areas contained quadrangles that did not meet the 5 hour minimum survey effort in at least one year to be included in analysis.

My landscape habitat association model set based on International Rusty Blackbird Working Group habitat classes yielded one top model, the global model, which accounted for 100% of the available weight (Table 3.2). None of the less parameterized models were important as judged by AICc. However, the variables I chose explained the number of Rusty

Blackbirds/party-hr per quadrangle better than the null model, which includes no explanatory variables.

TABLE 3.2. Generalized linear mixed model set for Rusty Blackbirds/party-hr based on International Rusty Blackbird Working Group habitat classifications. Year and rainfall nested within year were the random effects.

model	fixed effects	AICc	ΔAICc	likelihood	weight
1	**global**	2665.7	0	1	1
2	soilC+soilC/D	2723.9	58.4	0	0
3	soilC	2737.4	71.9	0	0
4	FF+WW+pecan+developed	2802.6	137.1	0	0
5	pecan+developed	2819.0	153.5	0	0
6	developed	2820.7	155.3	0	0
7	FF+developed	2822.6	157.2	0	0
8	WW+pecan	2823.8	158.4	0	0
9	FF+WW+pecan	2825.6	160.1	0	0
10	WW	2827.4	161.9	0	0
11	FF+WW	2829.1	163.7	0	0
12	soilD	2833.7	168.2	0	0
13	soilC/D	2836.5	171	0	0
14	pecan	2844.2	178.8	0	0
15	**null**	2845.8	180.3	0	0
16	FF+pecan	2846.2	180.8	0	0
17	FF	2847.7	182.3	0	0

* FF= floodplain forest, WW = woody wetland, soilC = soil hydrologic group C, soilC/D = soil hydrologic group C/D, soilD = soil hydrologic group D

Based on the magnitude of the parameter estimates, pecan orchards had the strongest positive relationship with Rusty Blackbird abundance (2.76 ± 0.72, df = 960, $p = 0.0001$). A one unit increase in pecan cover increased the odds of reporting a RUBL/party-hr by 2.76. After back-transforming the log-scale estimate, there is a predicted increase of 15.8 RUBL/party-hr for each 1% increase in pecan orchard cover of a quadrangle. Soil hydrologic groups C, C/D, and D were all mildly positive with abundance (soil C: 0.03 ± 0.003, soil C/D: 0.02 ± 0.003, soil D: 0.01 ± 0.003; df = 960, $p < 0.0001$). Higher cover of woody wetland (-0.03 ± 0.008, df = 960, p

< 0.0001) and developed land (-0.02 ± 0.005, df = 960, $p < 0.0001$) in a quadrangle decreased Rusty Blackbird abundance. Floodplain forest was the only variable with an estimate that contained zero, implying that there was no meaningful relationship with Rusty Blackbirds. Abundance varied positively or negatively with year but was only statistically significant for 2011 (-1.46 ± 0.71, df = 960, $p = 0.04$). The effect of annual rainfall was not meaningful for any of the eight years (Appendix V, Table V.1).

For my second model set (using LCWCS classified habitat), the global model again accounted for 100% of the available weight (Table 3.3).

TABLE 3.3. Generalized linear mixed model set for Rusty Blackbirds/party-hr based on Louisiana Comprehensive Wildlife Conservation Strategy habitat classifications. Year was the only random effect.

model	fixed effects	AICc	ΔAICc	likelihood	weight
1	**global**	2625.32	0	1	1
2	BLH+swamp+pecan+lawn	2755.25	130.1	0	0
3	soilC+soilC/D	2769.77	144.6	0	0
4	BLH+swamp	2771.78	146.6	0	0
5	BLH+swamp+pecan	2771.88	146.8	0	0
6	swamp+pecan	2777.66	152.5	0	0
7	swamp	2777.86	152.7	0	0
8	soilC	2789.22	164.1	0	0
9	pecan+lawn	2869.83	244.7	0	0
10	BLH+lawn	2870.46	245.3	0	0
11	lawn	2872.76	247.6	0	0
12	soil C/D	2887.86	262.7	0	0
13	BLH+pecan	2894.2	269.1	0	0
14	BLH	2896.59	271.5	0	0
15	soil D	2897.2	272.1	0	0
16	pecan	2899.01	273.9	0	0
17	**null**	2901.77	276.6	0	0

* BLH = bottomland hardwood forest, soil C = soil hydrologic group C, soil C/D = soil hydrologic group C/D, soil D = soil hydrologic group D

The relationships between abundance and landscape cover explanatory variables were similar to the previous model set. Pecan orchards again had the strongest positive relationship with Rusty Blackbird abundance (1.93 ± 0.74, df = 968, p = 0.009). Soil hydrologic groups C, C/D, and D were all mildly positive with abundance (soil C: 0.03 ± 0.003, soil C/D: 0.02 ± 0.003, soil D: 0.01 ± 0.003; df = 968, p < 0.0001). The negative relationships were similar to before; higher cover of cypress-tupelo-blackgum swamp (-0.10 ± 0.015, df = 968, p < 0.0001) and lawn (-0.02 ± 0.005, df = 968, p = 0.0002) in a quadrangle decreased Rusty Blackbird abundance. Bottomland hardwood forest, analogous to floodplain forest, was the only variable with an estimate that contained zero, implying that it may not have a relationship with Rusty Blackbirds. Abundance varied positively or negatively with year but the relationship was only statistically significant for the years 2010, 2011, and 2014 (Appendix V, Table V.2).

To explore whether Rusty Blackbird abundance varied with annual rainfall, I compared cumulative Rusty Blackbirds/party-hr with total rainfall in surveyed quadrangles. Cross-correlations between annual Rusty Blackbird abundance and 3 month total rainfall per quad were not statistically significant, suggesting the two measures are independent (Figure 3.2).

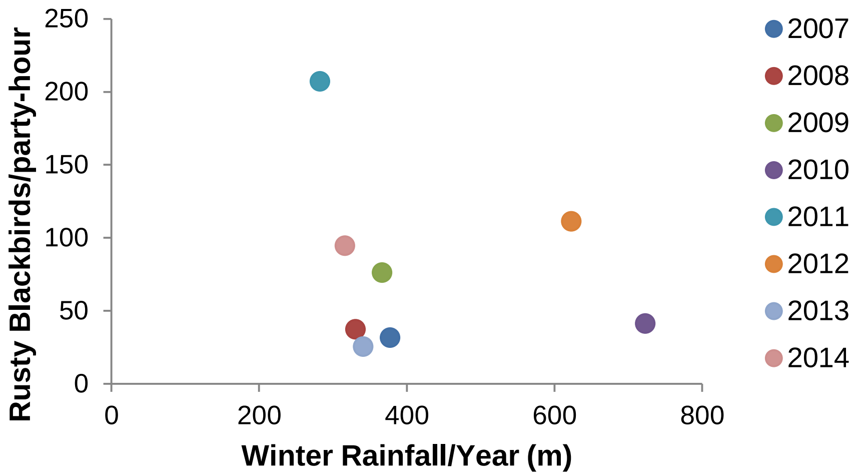

FIGURE 3.2. Relationship between total yearly Rusty Blackbirds/party-hr and total winter rainfall for all quadrangles that met the 5 hour threshold for analysis during the LWBA.

I predicted that there would be a positive relationship between the amount of forested wetland cover and Rusty Blackbird abundance in each quadrangle. My results for both model sets confirm a neutral relationship with floodplain forest/bottomland hardwood forest and a slightly negative relationship with woody wetland/cypress-tupelo-blackgum swamp. When comparing spatial patterns of Rusty Blackbird abundance to the distribution of forested wetlands within the state, there appears to be no association between areas of higher abundance and increased cover of forested wetlands (Figure 3.3).

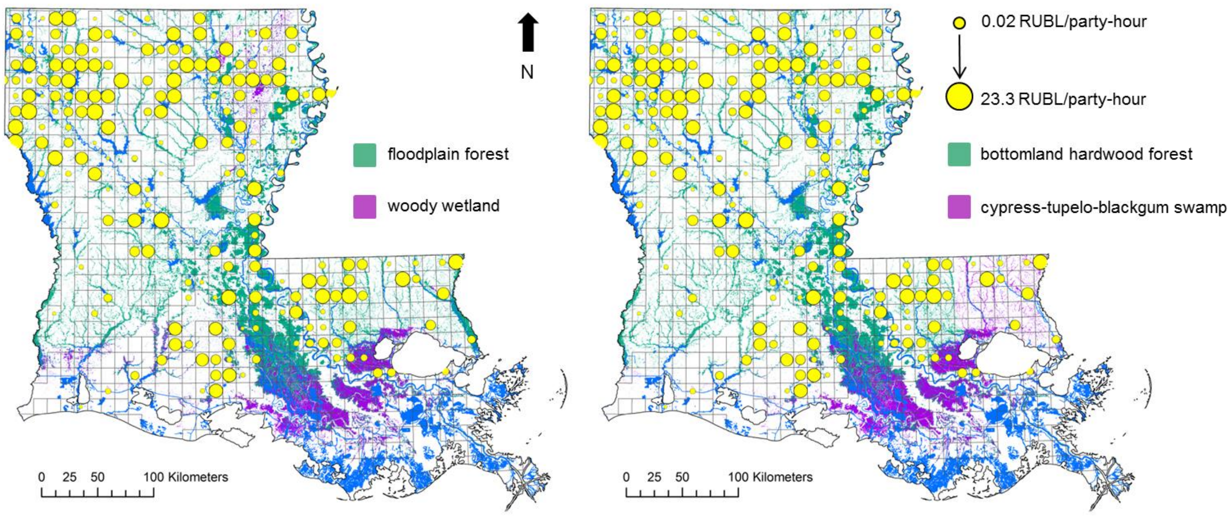

FIGURE 3.3. Cumulative Rusty Blackbirds/party-hr for each quadrangle (years 2007-2014) overlaid on forested wetlands. International Rusty Blackbird Working Group forested wetland classifications (left panel) are contrasted with those of the Louisiana Comprehensive Wildlife Conservation Strategy (right panel). Quadrangles with an outline met the 5 hour threshold to be included in the analysis; un-outlined areas were not included.

3.4 Discussion

Conclusions

The global model, which included all variables of interest, best modeled Rusty Blackbird abundance per party hour for both model sets (International Rusty Blackbird Working Group [IRBWG] and Louisiana Comprehensive Wildlife Conservation Strategy [LCWCS] reclassed habitat). Percent cover of pecan orchards was the most important variable; RUBL party-hr increased with increasing pecan cover. The magnitude of this effect was high for both model sets: for a 1% increase in pecan cover in a quadrangle you could expect an increase of 15.8 RUBL/party-hr (IRBWG) or 6.9 RUBL/party-hr (LCWCS). These are large predicted changes, however, an increase of 1% would be extremely unlikely as the average percent cover of pecan orchard was only 0.006% and the greatest cover in an individual quadrangle was 0.76%. Although pecan coverages were small, pecan orchards are probably more prevalent throughout the state because tree crops typically have low classification accuracy in the Cropland Data Layer (P. Willis, USDA National Agricultural Statistics Service, personal communication), which would underestimate cover. Satellite imagery lacks the resolution to identify small abandoned pecan groves such as those in residential areas, fallow orchards, or on small farms, which are used heavily by Rusty Blackbirds (Newell 2013). Even if increased cover could be accounted for, pecan orchards are still likely to have a positive effect on Rusty Blackbird abundance.

All soil hydrologic groups I included had a positive relationship with Rusty Blackbird abundance. Soil hydrologic groups C, C/D, and D have a slow rate of water transmission that would contribute to ponding (Appendix IV, Table IV.4). With a 1% unit increase in one of these soil types, RUBL/party-hr increases by 1.01 – 1.03. At larger landscape scales, such as in this study, soil hydrologic groups may be more important than any particular habitat because they promote shallow surface water regardless of habitat type or ground cover.

Unlike I predicted for a species described as a forested wetland specialist, no relationship existed between Rusty Blackbird abundance and floodplain forest or bottomland hardwood forest. There was a mildly negative relationship with woody wetland and cypress-tupelo-blackgum swamp habitat, which is similar to results from my field study (Chapter 2). To examine whether spatial patterns in abundance and forested wetlands might exist, I overlaid LWBA cumulative abundance data onto the distribution of forested wetlands in Louisiana (Figure 3.3). Areas of high Rusty Blackbird abundance do not seem to align with the distribution of forested wetlands throughout the state, but there were also tracts of forested wetlands that lacked hours of effort or were simply never surveyed (e.g. areas within the Atchafalaya River Basin; Figure 3.3). Some clusters of larger circles that are in the vicinity of urban areas (such as Shreveport in the northwest, Monroe in the northeast, and Baton Rouge in the southeast) may either reflect actual importance to Rusty Blackbirds or greater survey effort because these areas were easily accessible to birders (Figure 3.3).

In my previous chapter, I found that deep water cover increased with forested wetland cover during most of my survey rounds (Figures 2.14, 2.15, and 2.16). I may be seeing a neutral or even negative trend with forested wetlands because land cover datasets do not account for the cover of shallow water or wet leaf litter used for foraging in these habitats. Habitats can quickly change in suitability for Rusty Blackbirds depending on the amount of moisture at sites. Quadrangles with high forested wetland cover may be either chronically dry or have too much deep water to be used. To determine Rusty Blackbird habitat requirements, it may be more appropriate to account for the local ground cover conditions that are changing habitat suitability, which I cannot address with this dataset as I did in the previous chapter. However, combining land cover data with models of overbank flooding may be useful for determining the amount of surface water. A comparison of Rusty Blackbird abundance in flooded and unflooded quadrangles within floodplains could yield interesting results.

I indirectly addressed the issue of water cover in quadrangles by using total rainfall as a proxy. I summed the rainfall (December – February) in the month prior to the survey period and the two months the survey period occurred in. Newell (2013) found that Rusty Blackbirds increased their use of wetlands if there was more precipitation in the previous three days, which suggests that increased precipitation could be important for determining Rusty Blackbird abundance, especially in quadrangles with high forested wetland cover. However, I found no correlation between annual rainfall and Rusty Blackbird abundance. Precipitation may be important at the shorter time scale of days, rather than the three months I considered; yet, in my field study the total rainfall in the three days prior to a survey period did not have a meaningful effect on Rusty Blackbird occupancy compared to other variables. Hamel and Ozdenerol (2009) hypothesized that annual variation in weather could be affecting Rusty Blackbird movements. Radio-tracked Rusty Blackbirds in South Carolina and Georgia used the same collection of sites throughout the winter season and had an average home range of 5.08 km^2, implying that they were not traveling great distances within a season (Newell 2013; unpublished data analyzed by Borchert). Once Rusty Blackbirds have migrated to Louisiana they are likely staying within their home range for the winter, regardless of variability in rainfall. My results suggest that precipitation does not play a major role in determining their distribution, but precipitation could be acting at regional scales to influence migratory timing and the extent of travel.

Rusty Blackbirds can be difficult to detect, especially in forested wetlands, because they are secretive and can be camouflaged in the leaf litter where they forage. Surveying for Rusty Blackbirds may require an experienced birder familiar with their habits, which may not have been the case for all Louisiana Winter Bird Atlas volunteers. Inexperienced volunteers have different detection and identification abilities, which can lead to bias and error in abundance estimates (Dickinson et al. 2010). Additionally, 87% of Louisiana is comprised of private property (Twedt and Loesch 1999), which prevented access to some areas of the quadrangles. Because Rusty Blackbirds supposedly use forested wetlands, which may be inaccessible due to flooding or because they are privately owned, volunteers may have missed the highest Rusty Blackbird concentrations if they primarily surveyed from roads. If this is the case, large citizen science datasets that do not incorporate routes into forest interiors may be inappropriate for an analysis of abundance data for this rare species.

Last, I suspected that the five hour effort threshold I used for analysis may not have allowed enough time for birders to detect a Rusty Blackbird in a quadrangle, which would affect my results by underestimating abundance. To examine this possibility, I repeated the analysis using a threshold of ten or more hours of effort per year. The global model was again the top model and the estimates were similar, indicating that a lack of effort was not obscuring the relationship between abundance and landscape cover for this dataset.

Management Recommendations

Pecan orchards and soil hydrologic groups C, C/D, and D had positive relationships with Rusty Blackbird abundance. In reality, we cannot manage for soil type but we can incorporate pecan groves into current management practices. Pecans are an important food resource for Rusty Blackbirds, especially in the time period preceding cold weather (Newell 2013). Tree mast represented 19 – 34% of the diet of birds wintering in South Carolina and Georgia, depending on whether it was the eastern or western population (Newell 2013). Mettke-Hofmann et al. (2015) found that adult male Rusty Blackbirds were found more frequently in pecan groves and were in better body condition, presumably because of the higher nut biomass associated with these groves. For Rusty Blackbirds using a matrix of habitat patches in anthropogenically altered areas, maintaining pecan groves on the landscape is likely to be important. In groves that are not commercially harvested, landowners should avoid grooming the ground or collecting the entire crop to leave nuts available. When designing forest restoration projects, pecan trees (*Carya illinoinensis* and possibly *Carya aquatica*) planted alongside other trees would provide a food source for Rusty Blackbirds as well as other wildlife.

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CHAPTER 4. GENERAL DISCUSSION

4.1 Conclusions

At multiple scales, and even when using different habitat classifications, forested wetlands were the most important habitats related to Rusty Blackbird occupancy at my sites. Unexpectedly however, occupancy decreased with forested wetland cover (floodplain forest/bottomland hardwood forest and woody wetland/cypress-tupelo-blackgum swamp), which contrasts with my predictions and Rusty Blackbird natural history as described in the literature. The only scale where occupancy had an increasing trend with forested wetland cover was for cypress-tupelo-blackgum swamp at the 100 m scale during the year 2014. Analyses of habitat using Louisiana Winter Bird Atlas abundance data produced comparable results; I found no relationship with bottomland hardwood forest and a mildly negative relationship with swamp. Although Rusty Blackbirds have historically been considered forested wetland specialists (Avery 1995, Greenberg and Droege 1999, Greenberg and Matsuoka 2010), previous studies on the wintering ground have found that Rusty Blackbirds use a diversity of habitats. Suburban areas are frequently visited by Rusty Blackbirds, as well as pecan groves (DeLeon 2012, Newell 2013). However, wet bottomland hardwood forest was important for Rusty Blackbird occupancy in Arkansas (Luscier 2009, Luscier et al. 2010) and bottomland hardwood forest was the most supported variable associated with occupancy in my study.

The decreasing relationship between occupancy and forested wetland cover may have been observed because detecting a Rusty Blackbird at higher forested wetland cover is more difficult. I found that my ability to detect Rusty Blackbirds decreased during some rounds with increasing floodplain forest/bottomland hardwood forest cover, but not in woody wetland/swamp, which could be due to a number of reasons. The availability of water in drier forest areas could be patchy, causing birds to search for water outside of the point count circle. Alternatively, if the water in the general area of a site is too deep, they may be using the margins of my sites. Increased structural complexity and cover in forests, which could make birds harder to see, or increased predator abundance in these habitats, could also be potential factors that slightly decreased detection probability. Most of my sites were embedded in a landscape matrix of different habitat types and were rarely in pure forest. It is possible that Rusty Blackbirds could be more easily detected at low bottomland hardwood forest cover sites if they are forced to concentrate in small patches, which would increase detection probability at low cover. Luscier's (2009) study may have been able to emphasize the importance of wet bottomland hardwood forest because his sites were in pure stands, whereas research in Louisiana (DeLeon 2012), Georgia, and South Carolina (Newell 2013) studied Rusty Blackbirds using a matrix of habitats.

Forested wetlands may not actually lack importance for Rusty Blackbirds; rather, it may be that significant ecosystem alterations to forested wetland systems have diminished their quality. Extant forested wetland area has much decreased from what was available historically, but changing hydrological regimes in particular could have compounded the already decreased quality of forested wetlands for Rusty Blackbirds. Upstream bottomland hardwood forests are facing issues with drying associated with channelization and a lack of flooding (Shankman 1997, Gee et al. 2014). Cypress-tupelo swamps are becoming too deep without sediment input, and water salinity is increasing in coastal areas, leading to tree die-offs (Chambers et al. 2005, Shaffer et al. 2009, Day et al. 2012). Moreover, throughout the southeastern United States, increasing beaver (*Castor canadensis*) populations have led to a greater proportion and depth of floodplain inundation in headwater systems (Collen and Gibson 2001, Jakes et al. 2007). At my sites, I found that deep water cover generally increased with forested wetland cover. Deep water may have made primarily forested wetland sites unappealing to Rusty Blackbirds, which is especially apparent for flocks ($\geq$ 4 birds) in bottomland hardwood forest. Soil hydrologic type C, which is characterized by slow water transmission, appeared as an important variable positively related with occupancy. The less important soil hydrologic type D, characterized by impermeability and possibly associated with deeper water, was not as important, providing additional support for the decreased attractiveness of deep water. Additionally, there was a positive relationship between wet leaf litter and colonization and a negative relationship with extinction, which implies that Rusty Blackbirds use shallow areas that can provide access to leaf litter foraging substrate.

Rusty Blackbirds were also more likely to visit sites with greater invertebrate biomass and to leave sites with lower biomass. In my top models the wet leaf litter variable appeared with invertebrate biomass, which suggests a linkage between that substrate and the availability of invertebrates (Cummins 1973, Suberkropp et al. 1983). Forested wetlands in my study may only become important in the presence of wet leaf litter and when more food is available. The second most important model of invertebrate biomass at my sites, behind the null model, identified a negative relationship between lawn habitat and biomass. Although Rusty Blackbirds can use suburban lawns, overall invertebrate biomass in lawns may be too low for regular use. Lawns may be particularly important for Rusty Blackbirds immediately before and after precipitation events that make terrestrial worms more available as they come to the surface (Newell 2013). Another study of foraging Rusty Blackbirds found that forest, as opposed to grassy areas, had the highest invertebrate abundance (Mettke-Hofmann et al. 2015).

I also investigated landscape-scale Rusty Blackbird habitat associations in USGS 7.5-minute quadrangles using Louisiana Winter Bird Atlas data. The global model, which included all possible variables I tested, was the most important. Similar to my occupancy modeling results, there was a neutral or negative relationship with forested wetlands, which may be reflective of my inability to account for how much shallow water is in these wetland landscapes. Combining land cover with models of overbank flooding for the peak wintering period (January – February) could be more useful than a consideration of land cover alone.

At the quadrangle scale, the availability of water, rather than any particular habitat or ground cover could be more important. Soil hydrologic groups C, C/D, and D, which have a propensity to maintain surface water better than other soil groups, were positively associated with Rusty Blackbird abundance at this scale and may support this theory. Pecan orchard cover had the strongest positive relationship with Rusty Blackbird abundance; several studies have found pecan groves to be an important source of energy on the wintering ground (Newell 2013, Mettke-Hofmann et al. 2015). Abandoned pecan groves, where nuts can accumulate on the ground and are not harvested, could be particularly important. Allowing groves to persist near strategic areas (e.g. wildlife management areas, refuges, national forests, or National Audubon Society Important Bird and Biodiversity Areas [IBA]) could supplement Rusty Blackbirds' food supply during periods of low invertebrate abundance (Newell 2013).

4.2 Recommendations for Management and Future Research

Despite some conflicting results, I believe that my results support, rather than refute, that historically important wintering ground forested wetlands continue to be of value for Rusty Blackbirds. However, forested wetlands seem to be important only under certain conditions, including when shallowly flooded with above-water wet leaf litter and with increased invertebrate biomass. Impounded and leveed bottomland hardwood forests that lack riverine inputs may be too dry for Rusty Blackbirds. A lack of water can shift the trees in these communities from flood tolerant oaks, which are important sources of oak mast, to less flood tolerant trees (Gee et al. 2014). Additionally, flood control can affect invertebrate community composition and decrease invertebrate availability for foraging Rusty Blackbirds (Kennedy and Turner 2011). Future research could address how Rusty Blackbirds use floodplains that are flood controlled versus areas that lack major flood control measures. Comparisons of invertebrate biomass, mast, and water depth and availability between these sites would be valuable for assessing the effect of hydrological change on the decline.

Hydrologic alterations have also affected coastal swamps, which were not as prominent as bottomland hardwood forest in my models, but are where I saw some of the largest flocks of Rusty Blackbirds. Flood control has contributed to increasing depth in swamps because there are no longer inputs of sediment-laden river water to balance out subsidence rates. Swamps are gradually sinking, which stresses trees over time and leads to conversion of swamp to open water and marsh (Shaffer et al. 2009). Wet leaf litter is important for Rusty Blackbirds, but stressed swamps produce less aboveground leaf litter biomass (Hoeppner et al. 2008). For some coastal forested wetlands, river diversions have been proposed to reestablish the link between the river and the swamp (Shaffer et al. 2003). A study of pre and post river diversion habitat use, coupled with water depth measurements and ground cover estimation, could help determine whether diversions are effective at decreasing water depth over time, thereby increasing foraging substrate for Rusty Blackbirds.

Nearly half of Louisiana may have been composed of wetlands (Hefner et al. 1994). In light of drastic habitat changes in the southeast, forested wetland restoration can help reclaim lost habitat. Replanting of marginal farmland with bottomland trees (including pecan trees) could be beneficial, especially since Rusty Blackbirds have been observed to use these areas with increasing frequency as the trees age (Hamel et al. 2009). However, simply increasing habitat without hydrological restoration may not be enough (King et al. 2006). Much wetland restoration has been for waterfowl in the form of moist soil impoundments, but these open water wetlands lack the forest and shallow water that Rusty Blackbirds need. Luscier (2009) found that lowering water levels in greentree reservoirs, which are impounded areas of bottomland hardwood forest, increased Rusty Blackbird occupancy, presumably because of associated increases in shallow foraging habitat. Fluctuating the water levels in these reservoirs, rather than constant flooding, also promotes higher invertebrate abundance (Fredrickson and Batema 1992). Successful restoration for a variety of wildlife species, including Rusty Blackbirds, needs to strike a balance between the availability of many different wetland successional stages and hydroperiods. My results suggest that trees capable of providing leaf litter, shallow areas that support surface wet leaf litter for foraging, pecan trees, and higher invertebrate biomasses in wintering ground forested wetlands are all important for Rusty Blackbirds. Historic and continued wintering habitat loss and degradation may be the biggest threat to the Rusty Blackbird population, but targeted restoration practices could help slow the Rusty Blackbird decline.

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APPENDIX I. PROCESS USED FOR CHOOSING INVERTEBRATE SAMPLING LOCATIONS

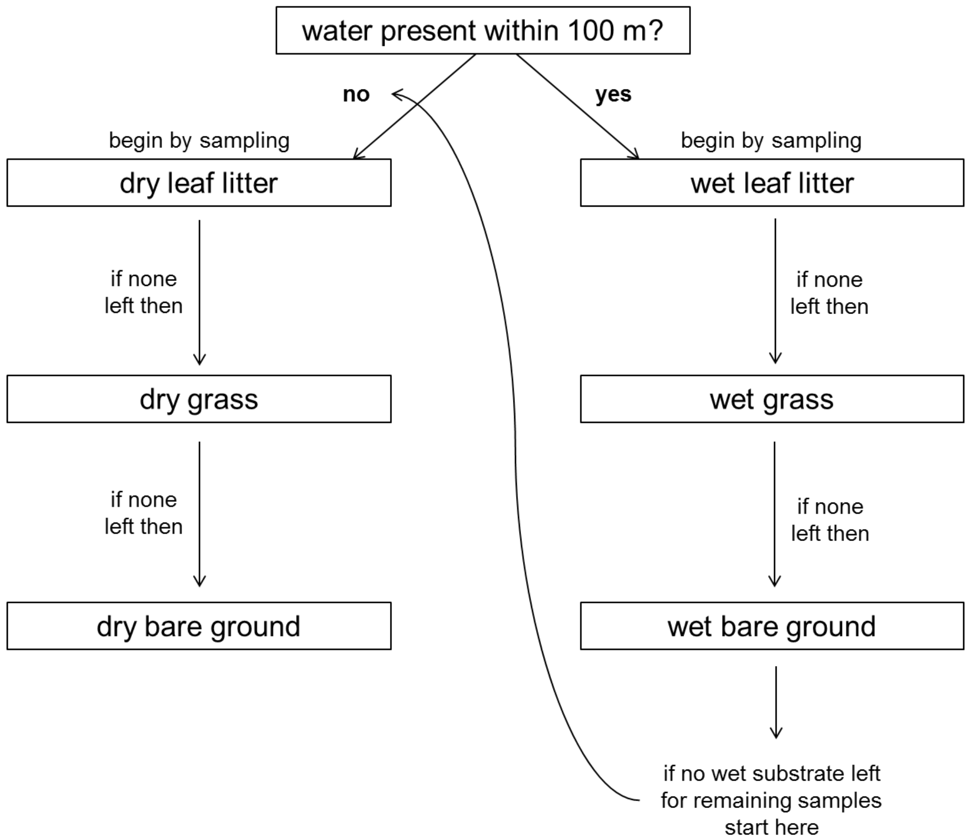

FIGURE I.1. Process for choosing substrate sample locations (5 samples/site/round) during the second year (2014). I chose sampling locations based on the substrates foraging Rusty Blackbirds favored during field observations in the first year.

APPENDIX II. PROBABILITY ESTIMATES FOR TOP MODELS

TABLE II.1. Probability estimates for the set of 36 sites/4 rounds using the International Rusty Blackbird Working Group landscape habitat classifications (600 m) as an example. The probability estimates ± SE are at the mean value of each covariate appearing in the top models.

top models	occupancy rate	detection rate	colonization rate	extinction rate
ψ(FF),p(cogr+time)	0.29 ± 0.11	0.79 ± 0.20	-	-
ψ(FF),$\gamma\varepsilon$(wetlitter),p(cogr+time)	0.29 ± 0.11	0.79 ± 0.20	0.47 ± 0.12	0.50 ± 0.14
ψ(FF+WW),p(cogr+time)	0.30 ± 0.12	0.79 ± 0.20	-	-

*FF = floodplain forest, WW = woody wetland, cogr = Common Grackles

APPENDIX III. DETECTABILITY COVARIATES AND MODELS

Table III.1. All detectability covariates collected. Covariates were eliminated if they were not biologically meaningful, correlated with other variables, or not consistently measured in the field. I retained some covariates (-) to model detection probability in the detection model sets.

covariate	description	reason if eliminated
year	year surveyed (2013 or 2014)	-
date	julian date of survey	-
time	standardized time of day	-
weather	measure of air moisture (sunny = 1, partly cloudy = 2, overcast = 3, rain = 4)	-
wind	Beaufort scale (1-5) wind speed	-
prior	Rusty Blackbirds previously detected within 200 m? (yes/no)	-
flock	# other blackbirds, American Robins, and European Starlings detected within 200 m	-
COGR	# Common Grackles detected within 200 m	-
RWBL	# Red-winged Blackbirds detected within 200 m	-
round	round surveyed (1 or 2)	structurally inherent to dynamic occupancy modeling
open	amount of grass averaged over rounds (estimates open space)	highly correlated with multiple habitat covariates
observer	person surveying	all surveys by one person
vocalization	birds vocalizing when found (yes/no)	related to detection of birds
temperature	temperature (°F)	not logically associated with detectability

TABLE III.2. Detectability model set for 36 sites/4 rounds.

model	**ψ**	**γ**	**ε**	**p**	**converged**
1	.	.	.	.	yes
2	.	.	.	year	yes
3	.	.	.	julian	yes
4	.	.	.	time	yes
5	.	.	.	prior	yes
6	.	.	.	weather	yes
7	.	.	.	wind	yes
8	.	.	.	rwbl	yes
9	.	.	.	cogr	yes
10	.	.	.	flock	yes
11	.	.	.	rwbl+time	yes
12	.	.	.	rwbl+weather	yes
13	.	.	.	rwbl+wind	yes
14	.	.	.	cogr+time	yes
15	.	.	.	cogr+weather	yes
16	.	.	.	cogr+wind	yes
17	.	.	.	flock+time	yes
18	.	.	.	flock+weather	yes
19	.	.	.	flock+wind	yes
20	.	.	.	julian+flock	yes
21	.	.	.	julian+prior	yes
22	.	.	.	julian+weather	yes
23	.	.	.	year+prior	yes
24	.	.	.	year+julian	yes
25	.	.	.	year+flock	yes
26	.	.	.	year+julian+flock	yes
27	.	.	.	year+julian+wind	yes
28	.	.	.	year+julian+weather	yes
29	.	.	.	year+julian+time+prior+weather+wind+rwbl+cogr+flock	yes

TABLE III.3. Detectability model set for 57 sites/2 rounds and abundance-adjusted 30 sites/2 rounds. Year was omitted because both rounds occurred in 2014.

rank	ψ	γ	ε	p	converged
1	.	.	.	.	yes
2	.	.	.	julian	yes
3	.	.	.	time	yes
4	.	.	.	prior	yes
5	.	.	.	weather	yes
6	.	.	.	wind	yes
7	.	.	.	cogr	yes
8	.	.	.	rwbl	yes
9	.	.	.	flock	yes
10	.	.	.	julian+flock	yes
11	.	.	.	julian+prior	yes
12	.	.	.	julian+weather	yes
13	.	.	.	rwbl+time	yes
14	.	.	.	rwbl+wind	yes
15	.	.	.	rwbl+weather	yes
16	.	.	.	cogr+time	yes
17	.	.	.	cogr+wind	yes
18	.	.	.	cogr+weather	yes
19	.	.	.	flock+time	yes
20	.	.	.	flock+wind	yes
21	.	.	.	flock+weather	yes
22	.	.	.	julian+time+prior+weather+wind+cogr+rwbl+flock	yes

TABLE III.4. Detectability results from package “unmarked” for R for 36 sites/4 seasons. Occupancy (ψ), colonization (γ), and extinction (ε) are held constant for all models.

rank	model	AIC	ΔAIC	AICwt	k	n	-2 log likelihood
1	p(cogr+time)	270.54	0.00	0.67	6	36	129.27
2	p(flock+time)	274.47	3.93	0.09	6	36	131.24
3	p(cogr+weather)	274.56	4.02	0.09	6	36	131.28
4	p(flock+weather)	276.10	5.55	0.04	6	36	132.05
5	p(cogr)	276.34	5.80	0.04	5	36	133.17
6	p(cogr+wind)	278.22	7.67	0.01	6	36	133.11
7	p(flock)	278.37	7.83	0.01	5	36	134.18
8	p(year+flock)	278.68	8.14	0.01	6	36	133.34
9	p(flock+wind)	279.34	8.79	0.01	6	36	133.67
10	p(rwbl+time)	279.62	9.08	0.01	6	36	133.81
11	p(rwbl+weather)	281.36	10.82	0.00	6	36	134.68
12	p(time)	283.74	13.20	0.00	5	36	136.87
13	p(rwbl+wind)	283.87	13.33	0.00	6	36	135.94
14	p(rwbl)	284.38	13.84	0.00	5	36	137.19
15	p(weather)	286.16	15.62	0.00	5	36	138.08
16	p(year)	287.60	17.06	0.00	5	36	138.80
17	p(wind)	287.91	17.37	0.00	5	36	138.96
18	**p(null)**	288.70	18.16	0.00	4	36	140.35
19	p(year+prior)	289.33	18.79	0.00	6	36	138.66
20	p(prior)	290.45	19.91	0.00	5	36	140.23
21	p(julian)	293.43	22.89	0.00	5	36	141.71
22	p(year+julian)	295.43	24.89	0.00	6	36	141.71
23	p(julian+flock)	295.43	24.89	0.00	6	36	141.71
24	p(julian+prior)	295.43	24.89	0.00	6	36	141.71
25	p(julian+weather)	295.43	24.89	0.00	6	36	141.71
26	p(year+julian+flock)	297.43	26.89	0.00	7	36	141.71
27	p(year+julian+wind)	297.43	26.89	0.00	7	36	141.71
28	p(year+julian+weather)	297.43	26.89	0.00	7	36	141.71
29	**p(global)**	309.43	38.89	0.00	13	36	141.71

TABLE III.5. Detectability results from package “unmarked” for R for 57 sites/2 seasons. Occupancy (ψ), colonization (γ), and extinction (ε) are held constant for all models.

rank	model	AIC	ΔAIC	AICwt	k	n	-2 log likelihood
1	p(cogr+time)	348.68	0.00	0.87	6	57	168.34
2	p(cogr+wind)	354.48	5.81	0.05	6	57	171.24
3	p(cogr+weather)	354.65	5.97	0.04	6	57	171.32
4	p(cogr)	354.76	6.08	0.04	5	57	172.38
5	p(flock+time)	366.75	18.07	0.00	6	57	177.37
6	p(flock+wind)	370.42	21.74	0.00	6	57	179.21
7	p(flock+weather)	370.60	21.92	0.00	6	57	179.30
8	p(flock)	370.69	22.01	0.00	5	57	180.35
9	p(time)	374.10	25.42	0.00	5	57	182.05
10	p(rwbl+time)	374.27	25.59	0.00	6	57	181.13
11	p(wind)	376.74	28.06	0.00	5	57	183.37
12	p(rwbl+wind)	377.04	28.36	0.00	6	57	182.52
13	p(rwbl+weather)	377.56	28.88	0.00	6	57	182.78
14	p(weather)	377.82	29.14	0.00	5	57	183.91
15	p(rwbl)	378.13	29.45	0.00	5	57	184.06
16	**p(null)**	378.24	29.56	0.00	4	57	185.12
17	p(prior)	380.23	31.55	0.00	5	57	185.11
18	p(julian)	381.93	33.25	0.00	5	57	185.97
19	p(julian+flock)	383.90	35.22	0.00	6	57	185.95
20	p(julian+prior)	383.90	35.22	0.00	6	57	185.95
21	p(julian+weather)	383.99	35.31	0.00	6	57	186.00
22	**p(global)**	395.90	47.22	0.00	12	57	185.95

TABLE III.6. Detectability results from package “unmarked” for R for abundance-adjusted 30 sites/2 seasons. Occupancy (ψ), colonization (γ), and extinction (ε) are held constant for all models.

rank	model	AIC	ΔAIC	AICwt	k	n	-2log likelihood
1	p(cogr+time)	213.99	0.00	0.58	6	30	100.99
2	p(flock+time)	215.74	1.75	0.24	6	30	101.87
3	p(cogr+wind)	218.94	4.95	0.05	6	30	103.47
4	p(cogr)	219.31	5.32	0.04	5	30	104.66
5	p(cogr+weather)	219.69	5.70	0.03	6	30	103.84
6	p(flock+wind)	221.08	7.09	0.02	6	30	104.54
7	p(flock)	221.14	7.15	0.02	5	30	105.57
8	p(flock+weather)	221.51	7.52	0.01	6	30	104.75
9	p(rwbl+time)	222.09	8.10	0.01	6	30	105.04
10	p(time)	226.96	12.98	0.00	5	30	108.48
11	p(rwbl+wind)	227.24	13.25	0.00	6	30	107.62
12	p(rwbl)	227.83	13.84	0.00	5	30	108.92
13	p(rwbl+weather)	228.01	14.02	0.00	6	30	108.00
14	p(wind)	230.49	16.50	0.00	5	30	110.24
15	**p(null)**	232.39	18.40	0.00	4	30	112.19
16	p(weather)	232.68	18.69	0.00	5	30	111.34
17	p(prior)	234.39	20.40	0.00	5	30	112.19
18	p(julian)	252.83	38.84	0.00	5	30	121.42
19	p(julian+flock)	254.83	40.84	0.00	6	30	121.42
20	p(julian+weather)	254.83	40.84	0.00	6	30	121.42
21	p(julian+prior)	254.83	40.84	0.00	6	30	121.42
22	**p(global)**	266.83	52.84	0.00	12	30	121.42

APPENDIX IV. HABITAT COVARIATES AND MODELS

TABLE IV.1. All site-scale (100 m) dynamic habitat covariates collected. These changed with each survey round and were thus modeled with site colonization (γ) and extinction (ε). Covariates were eliminated if they were not biologically meaningful, correlated with other variables, or not consistently measured in the field. I retained a few covariates (-) to model colonization and extinction in every habitat association model set.

covariate	description	reason if eliminated
shallow	% ground covered by shallow water	-
wetlitter	% ground covered by wet litter (damp and saturated categories)	-
wetgrass	% ground covered by wet grass	-
rain	total rainfall (mm) in the 3 days prior to a round	-
water	% ground covered by shallow and deep water	deep water not biologically meaningful
grass	% ground covered by dry and wet grass	dry grass not biologically meaningful
litter	% ground covered by leaf litter (dry, damp, and saturated)	correlated with more biologically meaningful wet litter
woody	% ground covered by dead woody debris (pulp, branches)	not measured consistently in field and pulp alone likely more important for invertebrates
leafy	% ground covered by leafy vegetation	not measured consistently in field
impervious	% impervious cover (pavement, buildings, etc.)	not biologically meaningful
toforest	average distance to nearest substantial tree cover	landscape covariates more meaningful

TABLE IV.2. All site (100 m) and landscape (600 m) scale land cover covariates used in habitat association model sets. These covariates were fixed every round and were only used to estimate site occupancy (ψ) in model sets. I developed separate candidate model sets for International Rusty Blackbird Working Group, Louisiana Comprehensive Wildlife Conservation Strategy, and field estimated cover classes.

covariate	description
International Rusty Blackbird Working Group Landscape Cover (600 m)	
floodplain forest	% cover of floodplain forest
woody wetland	% cover of woody wetland
developed	% cover of developed land (grassy areas, pavement, buildings, etc.)
soil C	% cover of soil hydrologic group C
soil D	% cover of soil hydrologic group D
Louisiana Comprehensive Wildlife Conservation Strategy Landscape Cover (600 m)	
bottomland hardwood forest	% cover of bottomland hardwood forest
cypress-tupelo-blackgum swamp	% cover of cypress-tupelo-blackgum swamp
lawn	% cover of low intensity development (lawn)
soil C	% cover of soil hydrologic group C
soil D	% cover of soil hydrologic group D
Field Estimated Habitat Cover (100 m)	
bottomland hardwood forest	% cover of bottomland hardwood forest
cypress-tupelo-blackgum swamp	% cover of cypress-tupelo-blackgum swamp
lawn	% cover of lawn
soil C*	% cover of soil hydrologic group C
soil D*	% cover of soil hydrologic group D

***** Cover of soil C and soil D data were obtained from the Soil Survey Geographic Database (USDA NRCS 2009) and included in the field estimated habitat cover model sets.

TABLE IV.3. Of the 590 GAP classes, 108 were present in Louisiana. Of those, I reclassed 37 into biologically meaningful categories for Rusty Blackbirds. The International Rusty Blackbird Working Group (IRBWG) grouped classes into meaningful types for the Rusty Blackbird on a regional scale. I used the habitats defined in the Louisiana Comprehensive Wildlife Conservation Strategy (LCWCS) to also reclass values into Louisiana-specific habitat types. Since analyses differed in scale, not all habitat types were present at smaller scales (Chapter 1 – 600 m analysis, Chapter 2 – Louisiana Winter Bird Atlas).

Reclass Value (IRBWG/LCWCS)	GAP Value	Scale Present At	GAP Class Name	NatureServe Description (Natureserve 2011)
developed /lawn	581	600 m atlas	Developed, Open Space	Includes areas with a mixture of some constructed materials, but mostly vegetation in the form of lawn grasses. Impervious surfaces account for less than 20 percent of total cover. These areas most commonly include large-lot single-family housing units, parks, golf courses, and vegetation planted in developed settings for recreation, erosion control, or aesthetic purposes.
developed /lawn	582	600 m atlas	Developed, Low Intensity	Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 20-49 percent of total cover. These areas most commonly include single-family housing units.
developed /lawn	583	600 m atlas	Developed, Medium Intensity	Includes areas with a mixture of constructed materials and vegetation. Impervious surfaces account for 50-79 percent of the total cover. These areas most commonly include single-family housing units.
developed	584	600 m atlas	Developed, High Intensity	Includes highly developed areas where people reside or work in high numbers. Examples include apartment complexes, row houses and commercial/industrial. Impervious surfaces account for 80 to100 percent of the total cover.
floodplain forest/bottomland hardwood forest	195	atlas	Central Appalachian Floodplain - Forest Modifier	This system encompasses floodplains from southern New England to Virginia. Mostly forested, these occur on floodplains of medium to large rivers where topography and process have resulted in the development of a relatively flat floodplain with a complex of upland and wetland temperate alluvial vegetation. This complex includes floodplain forests in which *Acer saccharinum, Populus deltoides*, and *Platanus occidentalis* are characteristic, as well as herbaceous sloughs and shrub wetlands. Most areas are underwater each spring; microtopography determines how long the various habitats are inundated. Depositional and erosional features may both be present depending on the particular floodplain, although there is a history of deposition in the floodplain formation.
floodplain forest	219	600 m atlas	East Gulf Coastal Plain Large River Floodplain Forest - Forest Modifier	This system represents a geographic subset of Kuchler's (1964) Southern Floodplain Forest. Examples may be found along large rivers of the East and Upper East Gulf Coastal Plain, especially the Apalachicola, Alabama, Tombigbee, Pascagoula, and Pearl rivers, all of which ultimately drain into the Gulf of Mexico. Several distinct plant communities can be recognized within this system that may be related to the array of different geomorphologic features present within the floodplain. Some of the major geomorphic features associated with different community types include natural levees, point bars, meander scrolls, oxbows, and sloughs (Sharitz and Mitsch 1993). Vegetation generally includes forests dominated by bottomland hardwood species and other trees tolerant of flooding. However, herbaceous and shrub vegetation may be present in certain areas as well.
floodplain forest/blh	220	600 m atlas	East Gulf Coastal Plain Small Stream and River Floodplain Forest	This is a predominantly forested system of the East Gulf Coastal Plain associated with small brownwater rivers and creeks. In contrast to East Gulf Coastal Plain Large River Floodplain Forest, it has fewer major geomorphic floodplain features typically associated with large river floodplains. Those features that are present tend to be smaller and more closely intermixed with one another, resulting in less obvious vegetational zonation. Bottomland hardwood tree species are typically important and diagnostic, although mesic hardwood species are also present in areas with less inundation, such as upper terraces and possibly second bottoms. As a whole, flooding occurs annually, but the water table usually is well below the soil surface throughout most of the growing season. Areas impacted by beaver impoundments are also included in this system.
floodplain forest/cypress-tupelo-blackgum swamp	221	600 m atlas	East Gulf Coastal Plain Tidal Wooded Swamp	This system encompasses the tidally flooded portions of river floodplains which flow into the northern Gulf of Mexico east of the Mississippi River. Large outflows of freshwater keep salinity levels at a minimum, and flooding is of short enough duration to allow survival of tree canopies. Bald cypress, tupelo, or ash generally dominate. These swamps may be regularly flooded at least twice daily (FNAI 1990).
floodplain forest/bottomland hardwood forest	223	atlas	East-Central Texas Plains Riparian Forest	This system occurs in various situations along small and intermittent streams of the "East Central Texas Plains" (sensu EPA; Griffith et al. (2004) Level III Ecoregion 33) and "Texas Blackland Prairie" (Level III Ecoregion 32), this comprising the terrain between the West Gulf Coastal Plain (or South Central Plains; Level III Ecoregion 35 sensu EPA) to the east and the Crosstimbers (EPA level III Ecoregion 27 and Edwards Plateau (EPA level III Ecoregion 30 etc. to the west). Some trees that may be present in

				stands of this system include sugarberry, netleaf hackberry, sycamore, little walnut, Arizona walnut, plateau live oak, water oak, willow oak, tall indigobush, swamp privet, silver maple, wingleaf soapberry, black willow, green ash, honey-locust, pecan, and cedar elm. The environment and vegetation of this system become generally and correspondingly drier from east to west with moister representatives (such as communities containing water oak) occurring in the eastern parts of the range. Representatives of this system typically occur in stream-scoured situations and vary in the openness of the habitat and physiognomy.
floodplain forest/cypress-tupelo-blackgum swamp	225	600 m atlas	Mississippi River Bottomland Depression	This system represents semipermanently flooded to saturated depressional areas of the lower Mississippi River Alluvial Valley, from southern Illinois south to Mississippi and Louisiana. These areas have a distinctly longer hydroperiod than other parts of the landscape. Typical and characteristic trees in examples of this system include swamp red maple, water hickory, pumpkin ash, water-locust, water tupelo, swamp blackgum, planertree, overcup oak, pin oak, black willow, and bald-cypress. Some characteristic shrubs include common buttonbush, stiff dogwood, swamp-loosestrife, swamp privet, Virginia-willow, and planertree. Herbs are uncommon, but floating water-primrose, lanceleaf arrowhead, hornwort spp., waterweed spp., pondweed spp., and lesser duckweed may be found.
floodplain forest/bottomland hardwood forest	226	600 m atlas	Mississippi River Floodplain and Riparian Forest	This systems group comprises floodplain forests in the Mississippi River Alluvial Plain of the southeastern United States, from far southeastern Missouri and extreme southern Illinois south to the Gulf of Mexico, including the floodplains and terraces of the Mississippi River and the Red River (in Louisiana and eastern Texas). Within this area, it includes broad gradients of river size, soil nutrient levels, and flood frequency, including smaller tributaries. Flooding ranges from semipermanent in the wettest areas to intermittent and short on the higher portions of the floodplain. Some of the major geomorphic features associated with different community types include natural levees, point bars, meander scrolls, oxbows, and sloughs. Small river floodplain forests have fewer major geomorphic floodplain features typically associated with large river floodplains. Those features that are present tend to be smaller and more closely intermixed with one another, resulting in less obvious vegetational zonation. Large rivers have greater variation in water levels and have flood regimes that integrate the effects of very large watersheds. Depositional landforms are larger, and communities can be more segregated. Along the Mississippi River, low bottomlands are characteristic. These are seasonally flooded backswamps, with flooding usually more frequent than every two years, generally by still water that may be impounded behind natural levees. Vegetation generally includes forests dominated by bottomland hardwood species and other trees tolerant of flooding. However, herbaceous and shrub vegetation may be present in certain areas, particularly on recently deposited bars and in oxbow lakes. Most examples are nearly contiguous over large areas, broken only by the river itself. Higher terraces may have a mosaic of floodplain and upland systems, and may include nonriverine wetland systems. Some of the most typical and characteristic tree species found in stands of this systems group include *Taxodium distichum, Nyssa aquatica, Acer saccharinum, Platanus occidentalis, Populus deltoides, Acer negundo, and Salix nigra. Other trees may include Celtis laevigata, Carya illinoinensis, Fraxinus pennsylvanica, Gleditsia triacanthos, Liquidambar styraciflua, Nyssa biflora, Quercus laurifolia, Quercus lyrata, Quercus michauxii, Quercus nigra, Quercus pagoda, Quercus phellos, Quercus similis, Quercus texana, Quercus virginiana, Salix nigra, Ulmus americana*, and *Ulmus crassifolia*. Three distinct groups of associations can be recognized. The lowest, wettest areas have some combination of *Taxodium distichum* and *Nyssa aquatica* dominating. Natural levees and riverfronts have a diverse mixture of trees that typically includes *Platanus occidentalis, Celtis laevigata, Fraxinus pennsylvanica, Acer saccharinum, Acer negundo,* and other species that benefit from the high light levels and heavy alluvial deposition of these sites. Soils are typically sandier than those of the lower bottomlands. *Arundinaria gigantea* (giant cane) is a common understory in these forests on natural levees and higher point bars, and may become dominant after thinning or removal of the overstory. Willow and cottonwood sandbars may have an open-canopy (woodland-type) structure. Moderate to high parts of the floodplain away from the levee are usually dominated by bottomland hardwoods, various mixtures of wetland oaks, including *Quercus laurifolia, Quercus michauxii, Quercus pagoda,* and sometimes a number of other oak species, along with *Liquidambar styraciflua* or other species. The wettest forests can be simple in structure, with an understory but little shrub or herb layer; others tend to have well-developed subcanopy, shrub, and herb layers. Woody vines are usually prominent. Shrubs and small trees include *Alnus serrulata, Arundinaria gigantea, Carpinus caroliniana, Cephalanthus occidentalis, Clethra alnifolia, Cornus foemina, Crataegus viridis, Forestiera acuminata, Ilex decidua, Itea virginica, Morella cerifera, Planera aquatica, Sabal minor,* and *Sebastiania fruticosa*. Vines may include *Berchemia scandens and Smilax bona-nox. Herbaceous species may include Boehmeria cylindrica, Carex complanata, Carex debilis, Carex intumescens, Carex joorii, Leersia virginica, Lycopus virginicus, Mikania scandens, Saccharum baldwinii, and Typha latifolia. Aquatic and*

				floating herbs include Lemna minor, Nelumbo lutea, Nuphar lutea ssp. advena, and *Nymphaea odorata.*
floodplain forest/bottomland hardwood forest	227	600 m atlas	Mississippi River Low Floodplain (Bottomland) Forest	Low bottomlands are usually seasonally flooded in backswamps, with flooding more frequent than every five years, usually more frequently than every two years, generally by still water that may be impounded behind natural levees, and are classed as Low Gradient Riverine Backwater wetlands in hydrogeomorphic classifications. Low bottomlands occur along the Mississippi River and its tributaries in the Mississippi River Alluvial Plain ecoregion. Prolonged flooding dominates this system, and its duration is greater than in the adjacent Mississippi River Riparian Forest. Overcup oak is the characteristic dominant species. Soils are clayey with poor internal drainage.
floodplain forest	228	600 m atlas	Mississippi River Riparian Forest	This system is comprised of "riverfront" Associations, generally temporarily (but rarely seasonally) flooded on point bars and natural levees adjacent to the river that formed them, with flooding more frequent than every five years, by flowing water directly from the stream. They occur along the lower Mississippi River and its tributaries in the Mississippi River Alluvial Plain ecoregion. They are classed as Low Gradient Riverine Overbank wetlands in a hydrogeomorphic classification. Flooding is of lower duration than on adjacent backswamps where water is impounded behind riverfront natural levees. Flooding is of longer duration than on adjacent high bottomlands that are typically temporarily flooded. Soils are typically sandier than those of low bottomlands. Giant cane (giant cane) is a common understory in these forests on natural levees and higher point bars, and may become dominant after thinning or removal of overstory. Willow and cottonwood sandbars may have an open-canopy (woodland-type) structure.
floodplain forest/bottomland hardwood forest	229	atlas	Red River Large Floodplain Forest	This system represents a geographic subset of Kuchler's (1964) Southern Floodplain Forest which is specifically restricted to the main stem of the Red River in southwestern Arkansas (partly bordering Texas) and Louisiana in the West Gulf Coastal Plain and Upper West Gulf Coastal Plain. Several distinct plant communities can be recognized within this system that may be related to the array of different geomorphic features present within the floodplain. Some of the major geomorphic features associated with different community types within the system include natural levees, point bars, meander scrolls, oxbows, and sloughs (Sharitz and Mitsch 1993). Vegetation generally includes forests dominated by bottomland hardwood species and other trees tolerant of flooding, including bald-cypress and water tupelo. However, herbaceous and shrub vegetation may be present in certain areas as well. This system is generally similar in concept to West Gulf Coastal Plain Large River Floodplain Forest but is distinct from it because of the difference in magnitude between the typical large rivers (such as the Trinity, Neches, and Sabine), on the one hand, and the Mississippi River on the other. Its range is conceptually coincident with the vast majority of Subsection 234Ai of Keys et al. (1995), excluding the portion of 234Ai within TNC Ecoregion 42 (Mississippi River Alluvial Plain). Its range is also coincident with Level IV Ecoregion 35g (red River Bottomlands) of Omernik.
floodplain forest/cypress-tupelo-blackgum swamp	230	600 m atlas	Southern Coastal Plain Blackwater River Floodplain Forest	This system occurs along certain river and stream drainages of the southern Coastal Plain of Florida, Alabama, Mississippi, and southwestern Georgia that are characterized by dark waters high in particulate and dissolved organic materials, and that generally lack floodplain development. In most cases these are streams that have their headwaters in sandy portions of the Outer Coastal Plain. Consequently, they carry little mineral sediment or suspended clay particles and are not turbid except after the heaviest rain events. The water is classically dark in color due to concentrations of tannins, particulates, and other materials derived from drainage through swamps or marshes (FNAI 1990). In comparison with spring-fed rivers and brownwater rivers of the region, this system tends to be much more acidic in nature and generally lacks extensive and continuous floodplain and levees; steep banks alternating with floodplain swamps are more characteristic (FNAI 1990). This system includes mixed rivers, with a mixture of blackwater and spring-fed tributaries such as the Suwannee River. Canopy species typical of this system are obligate to facultative wetland species such as bald-cypress (bald-cypress), water tupelo (water tupelo), and Atlantic white-cedar (Atlantic white-cedar).
floodplain forest/bottomland hardwood forest	233	600 m atlas	West Gulf Coastal Plain Large River Floodplain Forest	This system represents a geographic subset of Kuchler's (1964) Southern Floodplain Forest found west of the Mississippi River. Examples may be found along large rivers of the West Gulf Coastal Plain and Upper West Gulf Coastal Plain, especially the Trinity, Neches, Sabine, and others. Several distinct plant communities can be recognized within this system that may be related to the array of different geomorphic features present within the floodplain. Some of the major geomorphic features associated with different community types include natural levees, point bars, meander scrolls, oxbows, and sloughs (Sharitz and Mitsch 1993). Vegetation generally includes forests dominated by bottomland hardwood species and other trees tolerant of flooding, including bald-cypress and water tupelo. However, herbaceous and shrub vegetation may be present in certain areas as well.
floodplain forest/cypress-tupelo-blackgum swamp	234	600 m atlas	West Gulf Coastal Plain Near-Coast	These swamp forests are found along rivers flowing through the Gulf Coast Prairies and Marshes region of the Outer Coastal Plain of western Louisiana and adjacent Texas. Included are areas where the rivers enter bays and estuaries along the northern Gulf of

			Large River Swamp	Mexico that are somewhat tidally influenced.
floodplain forest/bottomland hardwood forest	235	600 m atlas	West Gulf Coastal Plain Small Stream and River Forest	This is a predominantly forested system of the West Gulf Coastal Plain associated with small rivers and creeks. In contrast to West Gulf Coastal Plain Large River Floodplain Forest, examples of this system have fewer major geomorphic floodplain features. Those features that are present tend to be smaller and more closely intermixed with one another, resulting in less obvious vegetational zonation. Bottomland hardwood tree species are typically important and diagnostic, although mesic hardwood species are also present in areas with less inundation, such as upper terraces and possibly second bottoms. As a whole, flooding occurs annually, but the water table usually is well below the soil surface throughout most of the growing season. Areas impacted by beaver impoundments are also included in this system.
floodplain forest	512	600 m atlas	East Gulf Coastal Plain Large River Floodplain Forest - Herbaceous Modifier	This system represents a geographic subset of Kuchler's (1964) Southern Floodplain Forest. Examples may be found along large rivers of the East and Upper East Gulf Coastal Plain, especially the Apalachicola, Alabama, Tombigbee, Pascagoula, and Pearl rivers, all of which ultimately drain into the Gulf of Mexico. Several distinct plant communities can be recognized within this system that may be related to the array of different geomorphologic features present within the floodplain. Some of the major geomorphic features associated with different community types include natural levees, point bars, meander scrolls, oxbows, and sloughs (Sharitz and Mitsch 1993). Vegetation generally includes forests dominated by bottomland hardwood species and other trees tolerant of flooding. However, herbaceous and shrub vegetation may be present in certain areas as well.
pecan orchard	74*	atlas	Pecan Orchard	Commercially cultivated pecan orchards.
woody wetland	1	atlas	South Florida Bayhead Swamp	This system consists of predominately broad-leaved hardwoods emergent amidst marshes of the south Florida Everglades region. These areas are often called tree islands as they occur on slightly elevated sites above the low-relief marshes and have been considered "perhaps the most striking botanical feature in the Everglades" (Loveless 1959). Individual islands often have a characteristic shape depending upon the size; large islands are often teardrop-shaped, smaller islands are circular (Loveless 1959, Gunderson and Loftus 1993). Patches range in size from ¼ acre to exceeding 300 or more acres. These islands often form an abrupt ecotone with adjacent fire-prone marshes. Fires enter bayhead swamps only under extreme drought conditions and may kill much of the bayhead vegetation and heavily reduce peat accumulation. If left long unburned, bayheads may succeed to hardwood hammocks.
woody wetland/cypress-tupelo-blackgum swamp	2	atlas	South Florida Cypress Dome	This system is found primarily in the Everglades and Big Cypress regions. This system consists of small forested wetlands in poorly drained depressions which are underlain by an impervious layer that impedes drainage and traps precipitation. They receive their common name from the unique dome-shaped appearance in which trees in the center are higher than those around the sides (Monk and Brown 1965). Pond-cypress is the dominant tree, with the oldest and largest individuals characteristically occupying the center, and smaller and younger individuals around the margins. Pools of stagnant, highly acid water may stand in the center of these depressions ranging from 1-4 feet in depth, but becoming increasingly shallow along the margins. The understory flora is typified by species with tropical affinities.
woody wetland	3	atlas	South Florida Dwarf Cypress Savanna	The scrub or dwarf cypress system covers extensive areas of south Florida, especially in the Big Cypress Swamp region of southwest Florida. These stunted stands of pond-cypress grow on shallow sands or marl soils above limestone bedrock. Individual trees are usually quite small and widely scattered, with canopy coverage ranging from 30-45% (Flohrschutz 1978). The understory shares much overlap with wet prairies of the region (Drew and Schomer 1984) and is dominated by the following genera: beaksedge, flatsedge, muhly, and sawgrass. The open, stunted aspect is maintained in part by stresses imposed by extreme seasonal water level changes and low-nutrient soils (Anonymous 1978). Ewel (1990b) suggests a hydroperiod of approximately 6 months for this type.
woody wetland	4	atlas	South Florida Mangrove Swamp	This swamp system occurs along intertidal and supratidal shorelines in southern Florida. The primary species comprising this system are red mangrove, black mangrove, white mangrove, and buttonwood, each with essentially tropical affinities and poor survival in cold temperatures. This system attains best development in low wave-energy, depositional environments. Examples occur on soils generally saturated with brackish water at all times and which become inundated during high tides. The brackish environment tends to limit competition from other species. Although at least three broad variants of this system can be recognized, i.e., riverine mangrove forests, fringe mangrove forests, and basin mangrove forests (Lugo et al. 1988), all are included here for now.
woody wetland/cypress-tupelo-blackgum swamp	237	600 m atlas	Gulf and Atlantic Coastal Plain Swamp Systems	This systems group consists of poorly drained, organic or mineral soil flats and basins of the Atlantic and Gulf coastal plains. These areas are saturated by rainfall and seasonal high water tables. Most are not associated with river floodplains, although one component system is a tidal swamp. Dominant tree species vary with geography. South

				of Virginia, *Taxodium distichum* and *Nyssa* spp. are the most characteristic trees in many of these swamps. In the North Atlantic Coastal Plain, *Chamaecyparis thyoides, Acer rubrum, Liquidambar styraciflua, Nyssa sylvatica, Quercus phellos,* and *Fraxinus pennsylvanica* are characteristic dominants. Tidal wooded swamps from Virginia to Florida are dominated by *Taxodium, Nyssa*, or *Fraxinus*. In the Mississippi River Valley, along with *Taxodium distichum* and Nyssa spp., characteristic trees include *Acer rubrum, Carya aquatica, Fraxinus profunda, Gleditsia aquatica, Planera aquatica, Quercus lyrata, Quercus palustris*, and *Salix nigra.* At the southern edge of this group's range, hydric hammocks in northern to central Florida are characterized by *Chamaecyparis thyoides* and Sabal palmetto. Important wetland oaks throughout much of the range include *Quercus michauxii, Quercus pagoda, Quercus phellos*, and *Quercus laurifolia.*
woody wetland/bottomland hardwood forest	238	atlas	Southern Coastal Plain Hydric Hammock	This system occupies flat lowlands along the southern and outermost portions of the Coastal Plain of the southeastern United States, usually over limestone substrates. Vegetation is characterized by mixed hardwood species (FNAI 1997), often with hydric oak species common (A. Johnson pers. comm.). In Florida examples of this system are often found adjacent to the floodplain of spring-fed rivers with relatively constant flows. In some areas, such as the Big Bend region, they occupy large areas of broad, shallow, mucky or seepy wetlands but generally do not receive overbank flooding (A. Johnson pers. comm.). In Alabama, this system is apparently confined to floodplains of the Mobile-Tensaw (A. Schotz pers. comm.), where examples are topographically higher than the surrounding floodplains.
woody wetland/cypress-tupelo-blackgum swamp	239	atlas	Southern Coastal Plain Seepage Swamp and Baygall	This wetland system consists of forested wetlands in acidic, seepage-influenced habitats of the East Gulf Coastal Plain, extending into central Florida. These are mostly evergreen forests generally found at the base of slopes or other habitats where seepage flow is concentrated. Resulting moisture conditions are saturated or even inundated. The vegetation is characterized by sweetbay and swamp blackgum. Examples occur in the outer portions of the Coastal Plain within the range of swampbay, and where sweetbay is an important or even dominant species. To the north this system grades into East Gulf Coastal Plain Northern Seepage Swamp, where evergreen species are largely replaced by deciduous species in the canopy. Due to excessive wetness, these habitats are normally protected from fire except those which occur during extreme droughty periods. These environments are prone to long-duration standing water, and tend to occur on highly acidic, nutrient-poor soils.
woody wetland/cypress-tupelo-blackgum swamp	240	atlas	West Gulf Coastal Plain Seepage Swamp and Baygall	This West Gulf Coastal Plain ecological system consists of forested wetlands (often densely wooded) in acidic, seepage influenced wetland habitats. These wetlands may occur in poorly developed upland drainages, toe-slopes, and small headwaters stream bottoms. These environments are prone to long duration standing water, and tend to occur on highly acidic, nutrient-poor soils. The vegetation is characterized by sweetbay, blackgum, swamp blackgum, and red maple, although there is some variation according to latitude. Understory vegetation throughout the region consistently supports an abundance of ferns, such as cinnamon fern, royal fern, and netted chainfern. In most cases, these wetlands are embedded in uplands with deep sandy soils. When these communities are associated with streams, they tend to be low gradient, with narrow, often braided channels and diffuse drainage patterns. Due to excessive wetness, these habitats are normally protected from fire except those which occur during extreme droughty periods. The limited examples in Oklahoma are somewhat depauperate and lack some of the more southern and eastern taxa (e.g., sweetbay, swamp blackgum).
woody wetland/cypress-tupelo-blackgum swamp	241	atlas	Atlantic Coastal Plain Nonriverine Swamp and Wet Hardwood Forest - Taxodium/Nyssa Modifier	This system consists of poorly drained, organic or mineral soil flats of the outer Atlantic Coastal Plain. These areas are saturated by rainfall and seasonal high water table without influence of river or tidal flooding. Fire is generally infrequent, but may be important for some associations. Vegetation consists of hardwood or mixed forests of Taxodium distichum, Nyssa spp., bottomland oaks, or other wetland trees of similar tolerance. The lower strata have affinities with pocosin or baygall systems rather than the river floodplain systems that have affinities with the canopy. The combination of canopy dominants and nonriverine, non-seepage hydrology distinguishes this system from other Coastal Plain systems.
woody wetland	243	600 m atlas	East Gulf Coastal Plain Southern Loblolly-Hardwood Flatwoods	This forested system occurs on broad upland flats in the East Gulf Coastal Plain of Alabama and Mississippi, as well as western parts of the lower terraces of the East Gulf Coastal Plain ("Florida Parishes"; 74d of EPA) of Louisiana, and likely occurs in other parts of the region as well. Its status and extent in this intervening terrain is unknown. Known examples in the Alabama/Mississippi parts of the range include a mosaic of open forests dominated by loblolly pine interspersed with patches of willow oak and sometimes other tree species. The ground surface displays an evident microtopography of alternating mounds and swales occurring in a tight local mosaic. These mounds are most likely "gilgai" (R. Wieland pers. comm.) resulting from vertic or shrink-swell properties of the Luinn soil series. Known examples display a range of moisture conditions from dry to wet. The wettest examples trap significant moisture from local rainfall events. These areas have ponded water for a minimum of several days at an interval and potentially for long periods of the year, especially when evapotranspiration

				is lowest. The vegetation of this system supports a relatively low vascular plant diversity and thus may appear floristically similar to other pine-hardwood vegetation of the region. The dry portion of this vegetational mosaic is dominated by grassy ground cover (longleaf wood-oats) with scattered emergent greenbriars (greenbrier spp.) underneath a nearly pure loblolly pine overstory. The historical composition of this type is unknown, but it seems likely that loblolly pine was a natural and even dominant component of this system, as it is in related systems in the West Gulf Coastal Plain (R. Evans pers. obs., T. Foti pers. comm.). Wetter areas are dominated by an overstory of willow oak with an abundance of dwarf palmetto in the understory. Although the specific role of fire in this system is unknown, low-intensity ground fires may have been ecologically important. Such fires could have originated in the surrounding East Gulf Coastal Plain Interior Shortleaf Pine-Oak Forest. In the western parts of the lower terraces of the East Gulf Coastal Plain ("Florida Parishes") of Louisiana (74d and adjacent 75a of EPA), the flatwoods vegetation tends to be dominated primarily by hardwoods in the most western portion, and a mixture of spruce pine and loblolly pine in the intermediate portion to the east of this (Smith 1996b). In this "Louisiana Florida Parishes Spruce Pine Flatwoods Forest" some characteristic species include spruce pine, diamondleaf oak, swamp chestnut oak, water oak, cherrybark oak, live oak, loblolly pine, and southern magnolia. Some important understory trees and shrubs include western mayhaw, dwarf palmetto (which may often be very abundant or dominant), and switchcane.
woody wetland	245	atlas	Lower Mississippi River Flatwoods	This system is comprised of forests, prairies and woodlands on Pleistocene terraces in the Mississippi Alluvial Plain of Arkansas, Missouri and Louisiana. It occurs primarily west of Crowley's Ridge on Pleistocene glacial outwash deposits in Arkansas and Missouri, and on Macon Ridge in Louisiana and adjacent Arkansas. The sites are above modern floodplains, but have poor internal drainage and are flat with poor runoff, leading to very wet conditions in winter and spring. They also often have a claypan that restricts both internal drainage and, later in the year, water availability. Therefore, they are very wet in the winter/spring and very dry in the summer, a moisture regime termed hydroxeric. Because of this moisture regime, the communities are variable, ranging from willow oak flats to post oak flats to prairies. In the 1940s, the Arkansas Game and Fish Commission produced a wildlife habitat map of Arkansas in which these sites were classified as "terrace hardwood forests." These communities have a large variety of upland and lowland tree species, ranging from post oak to overcup oak in a small area. Such species diversity may be explained by regeneration of species with dramatically different moisture tolerances on the same site in dry and wet years on these hydroxeric sites. Because the sites are above current floodplains and susceptible to being drained, they have been cleared at an even greater rate than nearby floodplain forests.
woody wetland/cypress-tupelo-blackgum swamp	247	600 m atlas	Southern Coastal Plain Nonriverine Basin Swamp	This system occupies large, seasonally inundated basins with peaty substrates in the southern and outermost portions of the Coastal Plain of the southeastern United States. These basins are nonriverine and do not receive overbank flooding. The southern range of this system extends into central Florida especially along the Atlantic Coast in Volusia and Brevard counties (A. Johnson pers. comm.). Examples are generally forested; the vegetation is characterized by bald-cypress, swamp blackgum, evergreen "bay" shrubs and/or mixed hardwoods. Emergent slash pine may also be present. Some characteristic shrubs include black titi, titi, shining fetterbush, and blaspheme-vine.
woody wetland/bottomland hardwood forest	251	atlas	West Gulf Coastal Plain Nonriverine Wet Hardwood Flatwoods	This system represents predominantly wet hardwood flatwoods of the West Gulf Coastal Plain of southern Arkansas, eastern Texas, and western Louisiana. Examples may be somewhat more common in the inland portions of the region but are also found in the Outer Coastal Plain as well. These areas are usually found on nonriverine, Pleistocene high terraces (EPA 35c). Soils are fine-textured, and hardpans may be present in the subsurface. The limited permeability of these soils contributes to perched water tables during fairly substantial portions of the year (when precipitation is greatest and evapotranspiration is lowest). Saturation occurs not from overbank flooding but typically whenever precipitation events occur. The local landscape is often a complex of ridges and swales, usually occurring in close proximity. There is vegetation variability related to soil texture and moisture and disturbance history. Most examples support hardwood forests or swamps, which are often heavily oak-dominated. Important species are tolerant of inundation. They include swamp chestnut oak, willow oak, diamondleaf oak, and sweetgum, with sparse coverage of wetland herbs such as southern waxy sedge. Some swales support unusual pockets of water ash and hawthorn spp. Some examples can contain loblolly pine.
woody wetland	252	600 m atlas	West Gulf Coastal Plain Pine-Hardwood Flatwoods	This system represents predominantly mesic to dry flatwoods of limited areas of inland portions of the West Gulf Coastal Plain. These areas are usually found on nonriverine, Pleistocene high terraces. Soils are fine-textured, and hardpans may be present in the subsurface. The limited permeability of these soils contributes to shallowly perched water tables during portions of the year when precipitation is greatest and evapotranspiration is lowest. Soil moisture fluctuates widely throughout the growing season, from saturated to very dry, a condition sometimes referred to elsewhere as xerohydric. Saturation occurs not from overbank flooding but typically whenever

				precipitation events occur. Local topography is a complex of ridges and swales, often in close proximity to one another. Ridges tend to be much drier than swales, which may hold water for varying periods of time. Within both ridges and swales, there is vegetation variability relating to soil texture and moisture and disturbance history. The driest ridges support loblolly pine and post oak; more mesic ridges have loblolly pine with white oak and species such as horsesugar and southern arrow-wood. Fire may have been an important natural process in some examples of this system (T. Foti pers. comm.).
woody wetland	259	600 m atlas	East Gulf Coastal Plain Near-Coast Pine Flatwoods	This system of open forests or woodlands occupies broad, sandy flatlands in a relatively narrow band along the northern Gulf of Mexico coast east of the Mississippi River [see map in Peet and Allard (1993)]. This range corresponds roughly to Ecoregion 75a (EPA 2004). These areas, often called "flatwoods" or "flatlands," are subject to high fire-return intervals even though they are subject to seasonally high water tables. Overstory vegetation is characterized by longleaf pine and to a lesser degree by slash pine. Understory conditions range from densely shrubby to open and herbaceous-dominated, based largely upon fire history. Fire is naturally frequent, with a fire-return time of from one to four years.
woody wetland/cypress-tupelo-blackgum swamp	263	atlas	Southern Coastal Plain Nonriverine Cypress Dome	This system consists of small forested wetlands, typically dominated by pond-cypress, with a characteristic and unique dome-shaped appearance in which trees in the center are higher than those around the sides (Monk and Brown 1965). Examples are known from the Southern Coastal Plain (Omernik Ecoregion 75 and adjacent 65) (EPA 2004) of Florida and Georgia, extending into Alabama, Mississippi and Louisiana. Examples occupy poorly drained depressions which are most often embedded in a matrix of pine flatwoods. The oldest and largest individual trees typically occupy the center of these domed wetlands, with smaller and younger individuals around the margins. Pools of stagnant, highly acidic water may stand in the center of these depressions ranging from 1-4 feet in depth, but becoming increasingly shallow along the margins. These sites are underlain by an impervious clay pan which impedes drainage and traps precipitation. Some examples may have thick (50-100 cm) organic layers. In addition to pond-cypress, other woody species may include swamp blackgum, Chapman's St. John's-wort, myrtleleaf St. John's-wort, myrtle dahoon, swamp doghobble, wax-myrtle, common buttonbush, sweetgum, coastal sweet-pepperbush, shining fetterbush, and downy snowbell.
woody wetland	562	600 m atlas	Introduced Riparian and Wetland Vegetation	Vegetation dominated (typically >60% canopy cover) by introduced species. These are spontaneous, self-perpetuating, and not (immediately) the result of planting, cultivation, or human maintenance. Land occupied by introduced vegetation is generally permanently altered (converted) unless restoration efforts are undertaken. Specifically, land cover is significantly altered/disturbed by introduced riparian and wetland vegetation.

* Pecan class is from the USDA NASS Cropland Data Layer (2010-2014). Pixels from the GAP layer were replaced by value 74 pixels from the CDL where they overlapped.

TABLE IV.4. I combined the 315 soil series present in Louisiana into their associated hydrologic group (U.S. Department of Agriculture 2009) before geoprocessing. I discarded biologically uninformative groups, indicated by (-).

Soil Hydrologic Group	**Covariate in**	**Description (U.S. Department of Agriculture 2009)**
A	-	Soils having a high infiltration rate (low runoff potential) when thoroughly wet. These consist mainly of deep, well drained to excessively drained sands or gravelly sands. These soils have a high rate of water transmission.
B	-	Soils having a moderate infiltration rate when thoroughly wet. These consist chiefly of moderately deep or deep, moderately well drained or well drained soils that have moderately fine texture to moderately coarse texture. These soils have a moderate rate of water transmission.
C	600 m* atlas	Soils having a slow infiltration rate when thoroughly wet. These consist chiefly of soils having a layer that impedes the downward movement of water or soils of moderately fine texture. These soils have a slow rate of water transmission.
D	600 m atlas	Soils having a very slow infiltration rate (high runoff potential) when thoroughly wet. These consist chiefly of clays that have a high shrink-swell potential, soils that have a high water table, soils that have a claypan or clay layer at or near the surface, and soils that are shallow over nearly impervious material. These soils have a very slow rate of water transmission.
dual group (A/D, B/D, C/D)	atlas**	If a soil is assigned to a dual hydrologic group (A/D, B/D, or C/D), the first letter is for drained areas and the second is for undrained areas. Only the soils that in their natural condition are in group D are assigned to dual classes.

***** The dual group C/D was combined with group C for the 600 m modeling to reduce the number of covariates.

** I kept the dual group C/D as a separate covariate in the atlas analysis because it was present at high cover at the state scale, meriting its own interpretation.

TABLE IV.5. Candidate model set for International Rusty Blackbird Working Group habitat types at the 600 m scale for 36 sites/4 rounds.

model	ψ	γ	ε	p	converged
1	.	.	.	.	yes
2	FF	.	.	cogr+time	yes
3	WW	.	.	cogr+time	yes
4	developed	.	.	cogr+time	yes
5	soilc	.	.	cogr+time	yes
6	soild	.	.	cogr+time	yes
7	FF+WW	.	.	cogr+time	yes
8	FF	wetlitter	wetlitter	cogr+time	yes
9	FF	shallow	shallow	cogr+time	yes
10	FF	rain	rain	cogr+time	yes
11	FF	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
12	WW	wetlitter	wetlitter	cogr+time	yes
13	WW	shallow	shallow	cogr+time	yes
14	WW	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
15	developed	wetgrass	wetgrass	cogr+time	yes
16	developed	shallow	shallow	cogr+time	yes
17	developed	rain	rain	cogr+time	yes
18	soilc	shallow	shallow	cogr+time	yes
19	soilc	rain	rain	cogr+time	yes
20	soilc	rain+shallow	rain+shallow	cogr+time	yes
21	soild	shallow	shallow	cogr+time	yes
22	soild	rain	rain	cogr+time	yes
23	soild	rain+shallow	rain+shallow	cogr+time	yes
24	FF+WW	wetlitter	wetlitter	cogr+time	yes
25	FF+WW	shallow	shallow	cogr+time	yes
26	FF+WW	rain	rain	cogr+time	yes
27	FF+WW	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
28	FF+developed	shallow	shallow	cogr+time	yes
29	FF+developed	rain	rain	cogr+time	yes
30	FF+WW+ developed+soilc	wetlitter+shallow +wetgrass+rain	wetlitter+shallow +wetgrass+rain	cogr+time	yes

TABLE IV.6. Candidate model set for Louisiana Comprehensive Wildlife Conservation Strategy habitat types at the 600 m scale for 36 sites/4 rounds.

model	**ψ**	**γ**	**ε**	**p**	**converged**
1	.	.	.	.	yes
2	BLH	.	.	cogr+time	yes
3	swamp	.	.	cogr+time	yes
4	lawn	.	.	cogr+time	yes
5	soilc	.	.	cogr+time	yes
6	soild	.	.	cogr+time	yes
7	BLH+swamp	.	.	cogr+time	yes
8	BLH	wetlitter	wetlitter	cogr+time	yes
9	BLH	shallow	shallow	cogr+time	yes
10	BLH	rain	rain	cogr+time	yes
11	BLH	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
12	swamp	wetlitter	wetlitter	cogr+time	yes
13	swamp	shallow	shallow	cogr+time	yes
14	swamp	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
15	lawn	wetgrass	wetgrass	cogr+time	yes
16	lawn	shallow	shallow	cogr+time	yes
17	lawn	rain	rain	cogr+time	yes
18	soilc	shallow	shallow	cogr+time	yes
19	soilc	rain	rain	cogr+time	yes
20	soilc	rain+shallow	rain+shallow	cogr+time	yes
21	soild	shallow	shallow	cogr+time	yes
22	soild	rain	rain	cogr+time	yes
23	soild	rain+shallow	rain+shallow	cogr+time	yes
24	BLH+swamp	wetlitter	wetlitter	cogr+time	yes
25	BLH+swamp	shallow	shallow	cogr+time	yes
26	BLH+swamp	rain	rain	cogr+time	yes
27	BLH+swamp	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
28	BLH+lawn	shallow	shallow	cogr+time	yes
29	BLH+lawn	rain	rain	cogr+time	yes
30	BLH+swamp+ lawn+soilc	wetlitter+shallow +wetgrass+rain	wetlitter+shallow +wetgrass+rain	cogr+time	yes

* Global model could not include all variables due to multicollinearity

TABLE IV.7. Candidate model set for field-estimated habitat at the 100 m scale for 36 sites/4 rounds.

model	ψ	γ	ε	p	converged
1	.	.	.	.	yes
2	BLH	.	.	cogr+time	yes
3	swamp	.	.	cogr+time	yes
4	lawn	.	.	cogr+time	yes
5	soilc	.	.	cogr+time	yes
6	soild	.	.	cogr+time	yes
7	BLH+swamp	.	.	cogr+time	yes
8	BLH	wetlitter	wetlitter	cogr+time	yes
9	BLH	shallow	shallow	cogr+time	yes
10	BLH	rain	rain	cogr+time	yes
11	BLH	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
12	swamp	shallow	shallow	cogr+time	yes
13	swamp	wetlitter	wetlitter	cogr+time	yes
14	swamp	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
15	lawn	shallow	shallow	cogr+time	yes
16	lawn	rain	rain	cogr+time	yes
17	soilc	shallow	shallow	cogr+time	yes
18	soilc	rain	rain	cogr+time	yes
19	soilc	rain+shallow	rain+shallow	cogr+time	yes
20	soild	shallow	shallow	cogr+time	yes
21	soild	rain	rain	cogr+time	yes
22	soild	rain+shallow	rain+shallow	cogr+time	yes
23	BLH+swamp	shallow	shallow	cogr+time	yes
24	BLH+swamp	wetlitter	wetlitter	cogr+time	yes
25	BLH+swamp	rain	rain	cogr+time	yes
26	BLH+swamp	wetlitter+shallow	wetlitter+shallow	cogr+time	yes
27*	BLH+swamp+ soild	wetlitter+wetgrass+ rain+shallow	wetlitter+wetgrass+ rain+shallow	cogr+time	yes

* Global model could not include all variables due to multicollinearity

TABLE IV.8. Candidate model set for 100 m field-estimated habitat for 57 sites/2 rounds and abundance adjusted 30 sites/2 rounds. Detection covariates included cogr+time+flock for the ≥ 4 Rusty Blackbirds abundance adjusted set

model	**ψ**	**γ**	**ε**	**p**	**converged**	**converged (≥ 4 adj)**
1	.	.	.	.	yes	yes
2	BLH	.	.	cogr+time	yes	yes
3	swamp	.	.	cogr+time	yes	yes
4	lawn	.	.	cogr+time	yes	yes
5	soilc	.	.	cogr+time	yes	yes
6	soild	.	.	cogr+time	yes	yes
7	BLH+swamp	.	.	cogr+time	yes	yes
8	BLH	wetlitter	wetlitter	cogr+time	yes	yes
9	BLH	shallow	shallow	cogr+time	yes	yes
10	BLH	rain	rain	cogr+time	yes	yes
11	BLH	biomass	biomass	cogr+time	yes	yes
12	BLH	wetlitter+biomass	wetlitter+biomass	cogr+time	yes	yes
13	BLH	shallow+biomass	shallow+biomass	cogr+time	yes	yes
14	swamp	shallow	shallow	cogr+time	yes	yes
15	swamp	wetlitter	wetlitter	cogr+time	yes	yes
16	swamp	biomass	biomass	cogr+time	yes	yes
17	swamp	wetlitter+biomass	wetlitter+biomass	cogr+time	yes	yes
18	swamp	shallow+biomass	shallow+biomass	cogr+time	yes	yes
19	lawn	shallow	shallow	cogr+time	**no**	yes
20	lawn	rain	rain	cogr+time	yes	yes
21	lawn	biomass	biomass	cogr+time	yes	yes
22	lawn	shallow+biomass	shallow+biomass	cogr+time	yes	yes
23	soilc	shallow	shallow	cogr+time	yes	yes
24	soilc	rain	rain	cogr+time	yes	yes
25	soild	rain	rain	cogr+time	yes	yes
26	BLH+swamp	shallow	shallow	cogr+time	yes	yes
27	BLH+swamp	wetlitter	wetlitter	cogr+time	yes	yes
28	BLH+swamp	biomass	biomass	cogr+time	yes	yes
29	BLH+swamp	shallow+biomass	shallow+biomass	cogr+time	yes	yes
30	BLH+swamp	wetlitter+biomass	wetlitter+biomass	cogr+time	yes	yes
31*	BLH+swamp +soilc	wetlitter+wetgrass +rain+biomass	wetlitter+wetgrass +rain+biomass	cogr+time	**no**	yes

*Global model could not include all variables due to multicollinearity

Table IV.9. Occupancy model results from package “unmarked” for R for 36 sites/4 rounds (600 m fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). I reclassed landscape values according to recommendations by the International Rusty Blackbird Working Group. Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	**model**	**AIC**	**ΔAIC**	**AICwt**	**k**	**n**	**-2 log likelihood**
1	ψ(FF)	267.36	0.00	0.25	7	36	126.68
2	ψ(FF),γε(wetlitter)	267.84	0.48	0.20	9	36	124.92
3	ψ(FF+WW)	269.13	1.77	0.10	8	36	126.57
4	ψ(FF+WW),γε(wetlitter)	269.61	2.25	0.08	10	36	124.81
5	ψ(FF),γε(shallow)	270.88	3.52	0.04	9	36	126.44
6	ψ(FF+developed),γε(shallow)	271.07	3.70	0.04	10	36	125.53
7	ψ(FF),γε(rain)	271.17	3.81	0.04	9	36	126.59
8	ψ(FF+developed),γε(rain)	271.43	4.06	0.03	10	36	125.71
9	ψ(FF),γε(wetlitter+shallow)	271.70	4.34	0.03	11	36	124.85
10	ψ(soilc)	272.11	4.74	0.02	7	36	129.05
11	ψ(WW)	272.22	4.86	0.02	7	36	129.11
12	ψ(developed)	272.53	5.16	0.02	7	36	129.26
13	ψ(soild)	272.54	5.17	0.02	7	36	129.27
14	ψ(FF+WW),γε(shallow)	272.64	5.28	0.02	10	36	126.32
15	ψ(WW),γε(wetlitter)	272.73	5.37	0.02	9	36	127.37
16	ψ(FF+WW),γε(rain)	272.95	5.58	0.02	10	36	126.47
17	ψ(FF+WW),γε(shallow+wetlitter)	273.46	6.10	0.01	12	36	124.73
18	ψ(developed),γε(wetgrass)	274.05	6.68	0.01	9	36	128.02
19	ψ(soilc),γε(shallow)	275.73	8.37	0.00	9	36	128.87
20	ψ(WW),γε(shallow)	275.90	8.54	0.00	9	36	128.95
21	ψ(soilc),γε(rain)	275.91	8.55	0.00	9	36	128.96
22	ψ(developed),γε(shallow)	276.21	8.84	0.00	9	36	129.10
23	ψ(soild),γε(shallow)	276.22	8.85	0.00	9	36	129.11
24	ψ(developed),γε(rain)	276.33	8.97	0.00	9	36	129.16
25	ψ(soild),γε(rain)	276.34	8.98	0.00	9	36	129.17
26	ψ(WW),γε(wetlitter+shallow)	276.62	9.26	0.00	11	36	127.31
27	**ψγε(global)****	277.41	10.04	0.00	18	36	120.70
28	ψ(soilc),γε(rain+shallow)	279.58	12.21	0.00	11	36	128.79
29	ψ(soild),γε(rain+shallow)	280.05	12.69	0.00	11	36	129.03
30	**ψγεp(null)**	288.70	21.33	0.00	4	36	140.35

*FF = floodplain forest, WW = woody wetland, soild = soil hydrologic group D, soilc = soil hydrologic group C

** soil D was left out of the global model due to correlations with soil C

TABLE IV.10. Occupancy model results from package "unmarked" for R for 36 sites/4 rounds (600 m fixed landscape (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). I reclassed landscape values according to the habitats outlined in the Louisiana Comprehensive Wildlife Conservation Strategy (Lester et al. 2005). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	n	-2 log likelihood
1	ψ(BLH)	270.59	0.00	0.14	7	36	128.29
2	ψ(BLH),γε(wetlitter)	271.06	0.47	0.11	9	36	126.53
3	ψ(BLH+swamp)	271.47	0.88	0.09	8	36	127.73
4	ψ(swamp)	271.52	0.94	0.09	7	36	128.76
5	ψ(BLH+swamp),γε(wetlitter)	271.96	1.37	0.07	10	36	125.98
6	ψ(swamp),γε(wetlitter)	272.02	1.43	0.07	9	36	127.01
7	ψ(soilc)	272.11	1.52	0.07	7	36	129.05
8	ψ(lawn)	272.35	1.76	0.06	7	36	129.17
9	ψ(soild)	272.54	1.95	0.05	7	36	129.27
10	ψ(lawn),γε(wetgrass)	273.88	3.29	0.03	9	36	127.94
11	ψ(BLH),γε(shallow)	274.16	3.57	0.02	9	36	128.08
12	ψ(BLH),γε(rain)	274.37	3.78	0.02	9	36	128.18
13	**ψγε(global)** **	274.43	3.84	0.02	19	36	118.21
14	ψ(BLH),γε(wetlitter+shallow)	274.94	4.35	0.02	11	36	126.47
15	ψ(BLH+swamp),γε(shallow)	275.01	4.43	0.02	10	36	127.51
16	ψ(BLH+lawn),γε(shallow)	275.13	4.54	0.01	10	36	127.56
17	ψ(swamp),γε(shallow)	275.19	4.61	0.01	9	36	128.60
18	ψ(BLH+swamp),γε(rain)	275.26	4.67	0.01	10	36	127.63
19	ψ(BLH+lawn),γε(rain)	275.35	4.77	0.01	10	36	127.68
20	ψ(soilc),γε(shallow)	275.73	5.15	0.01	9	36	128.87
21	ψ(BLH+swamp),γε(shallow+wetlitter)	275.82	5.24	0.01	12	36	125.91
22	ψ(swamp),γε(wetlitter+shallow)	275.90	5.32	0.01	11	36	126.95
23	ψ(soilc),γε(rain)	275.91	5.33	0.01	9	36	128.96
24	ψ(lawn),γε(shallow)	276.04	5.45	0.01	9	36	129.02
25	ψ(lawn),γε(rain)	276.15	5.56	0.01	9	36	129.07
26	ψ(soild),γε(shallow)	276.22	5.63	0.01	9	36	129.11
27	ψ(soild),γε(rain)	276.34	5.75	0.01	9	36	129.17
28	ψ(soilc),γε(rain+shallow)	279.58	8.99	0.00	11	36	128.79
29	ψ(soild),γε(rain+shallow)	280.05	9.47	0.00	11	36	129.03
30	**ψγεp(null)**	288.70	18.11	0.00	4	36	140.35

*BLH = bottomland hardwood forest, soild = soil hydrologic group D, soilc = soil hydrologic group C

**soil D was left out of the global model due to correlations with soil C

TABLE IV.11. Occupancy model results from package “unmarked” for R for 36 sites/4 rounds (100 m fixed field-estimated habitat (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	n	-2log likelihood
1	ψ(soilc)	270.37	0.00	0.15	7	36	128.19
2	ψ(BLH100)	270.82	0.45	0.12	7	36	128.41
3	ψ(BLH100),γε(wetlitter)	271.09	0.71	0.10	9	36	126.54
4	ψ(BLH100+swamp100)	271.46	1.09	0.09	8	36	127.73
5	ψ(BLH100+swamp100),γε(wetlitter)	271.58	1.21	0.08	10	36	125.79
6	ψ(lawn100)	271.69	1.32	0.08	7	36	128.84
7	ψ(soild)	271.91	1.54	0.07	7	36	128.96
8	ψ(swamp100)	272.12	1.75	0.06	7	36	129.06
9	ψ(swamp100),γε(wetlitter)	272.59	2.22	0.05	9	36	127.30
10	ψ(soilc),γε(shallow)	273.83	3.45	0.03	9	36	127.91
11	ψ(soilc),γε(rain)	274.08	3.70	0.02	9	36	128.04
12	ψ(BLH100),γε(shallow)	274.47	4.10	0.02	9	36	128.23
13	ψ(BLH100),γε(rain)	274.52	4.15	0.02	9	36	128.26
14	ψ(BLH100),γε(wetlitter+shallow)	275.00	4.63	0.01	11	36	126.50
15	ψ(BLH100+swamp100),γε(shallow)	275.05	4.67	0.01	10	36	127.52
16	ψ(BLH100+swamp100),γε(rain)	275.11	4.74	0.01	10	36	127.56
17	ψ(lawn100),γε(shallow)	275.33	4.96	0.01	9	36	128.67
18	ψ(lawn100),γε(rain)	275.44	5.07	0.01	9	36	128.72
19	ψ(BLH100+swamp100),γε(wetlitter+shallow)	275.48	5.11	0.01	12	36	125.74
20	ψ(soild),γε(shallow)	275.58	5.21	0.01	9	36	128.79
21	ψ(soild),γε(rain)	275.64	5.27	0.01	9	36	128.82
22	ψ(swamp100),γε(shallow)	275.77	5.40	0.01	9	36	128.89
23	ψ(swamp100),γε(wetlitter+shallow)	276.48	6.11	0.01	11	36	127.24
24	ψ(soilc),γε(rain+shallow)	277.44	7.07	0.00	11	36	127.72
25	ψ(soild),γε(rain+shallow)	279.23	8.86	0.00	11	36	128.61
26	**ψγε(global)**	282.89	12.52	0.00	17	36	124.45
27	**ψγεp(null)**	288.70	18.33	0.00	4	36	140.35

*BLH = bottomland hardwood forest, soild = soil hydrologic group D, soilc = soil hydrologic group C

TABLE IV.12. Occupancy model results from package "unmarked" for R for 57sites/2 rounds (100 m fixed field-estimated habitat (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	n	-2log likelihood
1	ψ(BLH100),γε(wetlitter+biomass)	344.24	0.00	0.22	11	57	161.12
2	ψ(BLH100),γε(wetlitter)	344.54	0.30	0.19	9	57	163.27
3	ψ(BLH100+swamp100),γε(wetlitter+biomass)	346.15	1.91	0.08	12	57	161.08
4	ψ(swamp100),γε(wetlitter)	346.30	2.05	0.08	9	57	164.15
5	ψ(BLH100+swamp100),γε(wetlitter)	346.40	2.15	0.07	10	57	163.20
6	ψ(swamp100),γε(wetlitter+biomass)	346.98	2.74	0.06	11	57	162.49
7	ψ(BLH100),γε(shallow+biomass)	347.66	3.41	0.04	11	57	162.83
8	ψ(BLH100+swamp100)	347.70	3.45	0.04	8	57	165.85
9	ψ(BLH100)	348.99	4.75	0.02	7	57	167.50
10	ψ(BLH100+swamp100),γε(biomass)	349.30	5.06	0.02	10	57	164.65
11	ψ(soilc)	349.36	5.12	0.02	7	57	167.68
12	ψ(swamp100),γε(shallow+biomass)	349.40	5.16	0.02	11	57	163.70
13	ψ(BLH100),γε(rain)	349.46	5.22	0.02	9	57	165.73
14	ψ(swamp100)	349.49	5.25	0.02	7	57	167.75
15	ψ(BLH100+swamp100),γε(shallow+biomass)	349.56	5.32	0.02	12	57	162.78
16	ψ(lawn100),γε(shallow+biomass)	349.75	5.51	0.01	11	57	163.88
17	ψ(BLH100),γε(shallow)	350.10	5.86	0.01	9	57	166.05
18	ψ(soild)	350.27	6.02	0.01	7	57	168.13
19	ψ(BLH100),γε(biomass)	350.49	6.24	0.01	9	57	166.24
20	ψ(lawn100)	350.67	6.43	0.01	7	57	168.34
21	ψ(soilc),γε(rain)	350.75	6.51	0.01	9	57	166.38
22	ψ(swamp100),γε(biomass)	350.95	6.70	0.01	9	57	166.47
23	ψ(soild),γε(rain)	351.17	6.92	0.01	9	57	166.58
24	ψ(soilc),γε(shallow)	351.35	7.10	0.01	9	57	166.67
25	ψ(swamp100),γε(shallow)	351.58	7.34	0.01	9	57	166.79
26	ψ(lawn100),γε(rain)	351.88	7.64	0.00	9	57	166.94
27	ψ(BLH100+swamp100),γε(shallow)	351.96	7.72	0.00	10	57	165.98
28	ψ(lawn100),γε(biomass)	352.08	7.84	0.00	9	57	167.04
29	**ψγεp(null)**	378.24	33.99	0.00	4	57	185.12

*BLH = bottomland hardwood forest, soild = soil hydrologic group D, soilc = soil hydrologic group C

**global model was omitted due to nonconvergence

TABLE IV.13. Abundance adjusted (≥ 4 Rusty Blackbirds) occupancy model results from package "unmarked" for R for 30sites/2 rounds (100 m fixed field-estimated habitat (ψ) covariates and 100 m dynamic (γ, ε) ground cover covariates). Detectability was p(cogr+time+flock) for all models except the null. Models within Δ2AIC had substantial model support.

rank	model	AIC	ΔAIC	AICwt	k	n	-2log likelihood
1	ψ(blh100)	212.28	0.00	0.31	8	30	98.14
2	ψ(blh100+swamp100)	214.02	1.74	0.13	9	30	98.01
3	ψ(blh100),γε(biomass)	214.76	2.48	0.09	10	30	97.38
4	ψ(soild)	216.18	3.90	0.04	8	30	100.09
5	ψ(swamp100)	216.23	3.95	0.04	8	30	100.11
6	ψ(blh100),γε(shallow)	216.23	3.95	0.04	10	30	98.12
7	ψ(blh100),γε(wetlitter)	216.28	3.99	0.04	10	30	98.14
8	ψ(blh100),γε(rain)	216.28	4.00	0.04	10	30	98.14
9	ψ(lawn100)	216.43	4.15	0.04	8	30	100.22
10	ψ(soilc)	216.56	4.28	0.04	8	30	100.28
11	ψ(blh100+swamp100),γε(biomass)	216.64	4.35	0.04	11	30	97.32
12	ψ(blh100+swamp100),γε(shallow)	217.97	5.69	0.02	11	30	97.99
13	ψ(blh100+swamp100),γε(wetlitter)	218.02	5.73	0.02	11	30	98.01
14	ψ(blh100),γε(wetlitter+biomass)	218.50	6.22	0.01	12	30	97.25
15	ψ(blh100),γε(shallow+biomass)	218.66	6.38	0.01	12	30	97.33
16	ψ(swamp100),γε(biomass)	219.03	6.75	0.01	10	30	99.52
17	ψ(lawn100),γε(biomass)	219.22	6.93	0.01	10	30	99.61
18	ψ(swamp100),γε(wetlitter)	220.10	7.81	0.01	10	30	100.05
19	ψ(soild),γε(rain)	220.15	7.87	0.01	10	30	100.08
20	ψ(swamp100),γε(shallow)	220.16	7.88	0.01	10	30	100.08
21	ψ(lawn100),γε(shallow)	220.37	8.09	0.01	10	30	100.19
22	ψ(blh100+swamp100),γε(wetlitter+biomass)	220.40	8.12	0.01	13	30	97.20
23	ψ(lawn100),γε(rain)	220.41	8.12	0.01	10	30	100.20
24	ψ(soilc),γε(shallow)	220.50	8.21	0.01	10	30	100.25
25	ψ(soilc),γε(rain)	220.53	8.25	0.01	10	30	100.26
26	ψ(blh100+swamp100),γε(shallow+biomass)	220.55	8.26	0.01	13	30	97.27
27	ψ(swamp100),γε(shallow+biomass)	222.96	10.68	0.00	12	30	99.48
28	ψ(swamp100),γε(wetlitter+biomass)	223.03	10.74	0.00	12	30	99.51
29	ψ(lawn100),γε(shallow+biomass)	223.16	10.87	0.00	12	30	99.58
30	**ψγε(global)**	225.89	13.61	0.00	18	30	94.95
31	**ψγεp(null)**	232.39	20.10	0.00	4	30	112.19

*BLH = bottomland hardwood forest, soild = soil hydrologic group D, soilc = soil hydrologic group C

TABLE IV.14. Linear mixed-effects model set for invertebrate biomass as the response variable and field-estimated 100 m habitat and 100 m ground cover as the fixed predictor variables. Round (two rounds) was the random effect within the single year (2014). I ranked models by lowest AIC value. Models within Δ2 AIC had the most model support.

rank	fixed effects	random effect	df	AICc	ΔAICc	logLik	weight
1	**null**	round	3	377.55	0.00	-185.66	0.53
2	lawn	round	4	379.48	1.94	-185.56	0.20
3	wetgrass	round	4	380.74	3.19	-186.19	0.11
4	wetlitter	round	4	380.84	3.30	-186.24	0.10
5	swamp	round	4	383.15	5.61	-187.39	0.03
6	shallow	round	4	383.54	5.99	-187.59	0.03
7	BLH	round	4	387.67	10.13	-189.65	0.00
8	lawn+shallow	round	5	388.47	10.93	-188.96	0.00
9	swamp+wetlitter	round	5	388.62	11.08	-189.03	0.00
10	swamp+shallow	round	5	391.72	14.18	-190.58	0.00
11	BLH+wetlitter	round	5	392.26	14.72	-190.85	0.00
12	BLH+shallow	round	5	394.07	16.53	-191.76	0.00
13	**global**	round	9	423.30	45.76	-201.79	0.00

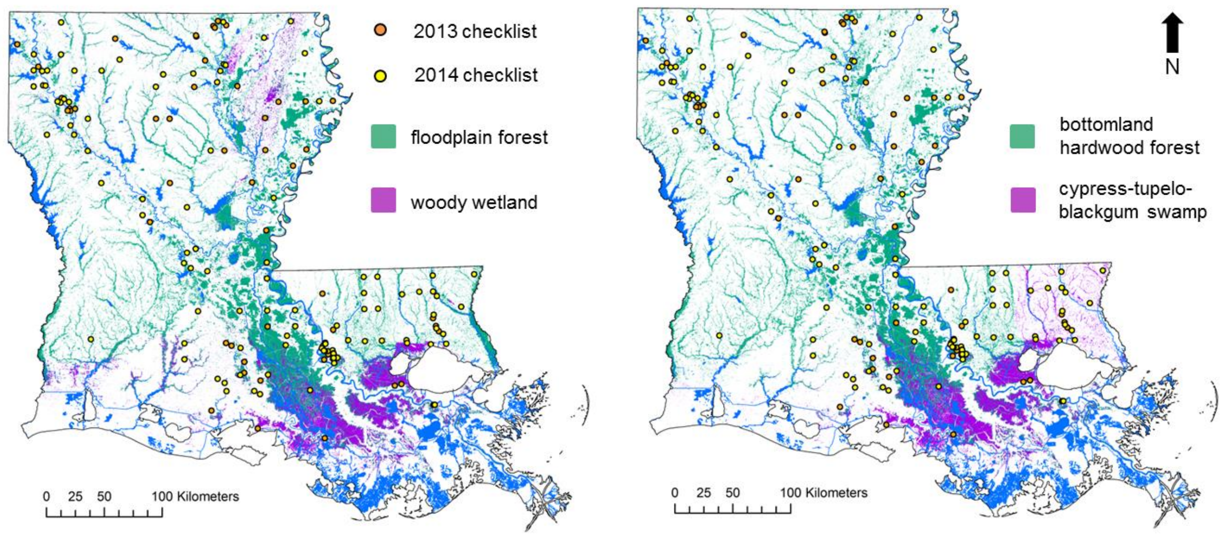

FIGURE IV.1. Locations of Rusty Blackbird checklists from 2013-2014, corresponding to the same years I conducted field surveys, overlaid onto the distribution of forested wetlands. International Rusty Blackbird Working Group (left panel) and Louisiana Comprehensive Wildlife Conservation Strategy (right panel) forested wetland types are depicted. Blue areas represent water.

APPENDIX V. PARAMETER ESTIMATES FOR TOP LOUISIANA WINTER BIRD ATLAS MODELS

TABLE V.1. Generalized linear mixed-effects model parameter estimates for Rusty Blackbirds/party-hour as the response variable and landscape cover (International Rusty Blackbird Working Group habitat classes) within USGS 7.5-minute topographic quadrangles as the fixed predictor variables. The random effects were year of the Louisiana Winter Bird Atlas and yearly total rainfall (December of the previous year – February of the survey year) nested within year.

Fixed Effects	Estimate	SE	df	t Value	Pr > |t|
intercept	-1.93	0.493	7	-3.92	0.0057
floodplain forest	0.00	0.003	960	-1.17	0.2414
pecan	2.76	0.721	960	3.83	0.0001
woody wetland	-0.03	0.008	960	-3.36	0.0008
developed	-0.02	0.005	960	-4.19	<0.0001
soil C	0.03	0.003	960	10.81	<0.0001
soil C/D	0.02	0.003	960	6.48	<0.0001
soil D	0.01	0.003	960	5.05	<0.0001

Random Effects	Year	Estimate	SE Predicted	df	t Value	Pr > |t|
year	2007	0.157	0.8828	960	0.18	0.8592
year	2008	-0.212	0.7461	960	-0.28	0.7761
year	2009	0.931	0.6298	960	1.48	0.1395
year	2010	1.177	0.7252	960	1.62	0.1049
year	2011	-1.464	0.7079	960	-2.07	0.0389
year	2012	-0.182	0.5661	960	-0.32	0.7483
year	2013	-0.872	0.8511	960	-1.02	0.3059
year	2014	0.492	0.6114	960	0.81	0.421
rain(year)	2007	0.000	0.0002	960	-0.44	0.6575
rain(year)	2008	0.000	0.0002	960	0.05	0.9619
rain(year)	2009	0.000	0.0002	960	-1.79	0.0745
rain(year)	2010	0.000	0.0002	960	-2.71	0.0068
rain(year)	2011	0.001	0.0003	960	4.03	<0.0001
rain(year)	2012	0.000	0.0001	960	1.16	0.2454
rain(year)	2013	0.000	0.0002	960	0.85	0.3982
rain(year)	2014	0.000	0.0002	960	0.31	0.7593

TABLE V.2. Generalized linear mixed-effects model parameter estimates for Rusty Blackbirds/party-hour as the response variable and landscape cover (Louisiana Comprehensive Wildlife Conservation Strategy habitat classes) within USGS 7.5-minute topographic quadrangles as the fixed predictor variables. The random effects were years of the Louisiana Winter Bird Atlas.

Fixed Effects	**Estimate**	**SE**	**df**	**t Value**	**Pr > |t|**
intercept	-1.719	0.266	7	-6.46	0.0003
BLH	0.003	0.003	968	0.87	0.3825
swamp	-0.097	0.015	968	-6.35	<.0001
pecan	1.937	0.742	968	2.61	0.0092
lawn	-0.019	0.005	968	-3.73	0.0002
soil C	0.027	0.003	968	9.99	<.0001
soil C/D	0.018	0.003	968	6.95	<.0001
soil D	0.011	0.002	968	4.5	<.0001

Random Effects	**Year**	**Estimate**	**SE Predicted**	**df**	**t Value**	**Pr > |t|**
year	2007	-0.321	0.2419	968	-1.33	0.1853
year	2008	-0.269	0.2358	968	-1.14	0.2536
year	2009	-0.088	0.2168	968	-0.41	0.6853
year	2010	-0.704	0.2321	968	-3.03	0.0025
year	2011	0.988	0.2038	968	4.85	<.0001
year	2012	0.201	0.2105	968	0.96	0.3391
year	2013	-0.296	0.2509	968	-1.18	0.2391
year	2014	0.554	0.2139	968	2.59	0.0098

APPENDIX VI. PERMISSIONS

Gmail by Google

Sinéad Borchert <sineadborchert@gmail.com>

home range size

3 messages

Sinéad Borchert <sineadborchert@gmail.com> Thu, Mar 26, 2015 at 12:07 PM
To: Patricia Wohner <pjwohner@gmail.com>

Hey - quick update. I calculated a 95% KDE using 17 of your birds with a cut off of 25 or greater points. The average home range size was 5.08 km2, so the circular buffer (radius) I'm using around my sites is 1.272 km.

Just thought you'd be curious! That's smaller than I previously thought. I'll let you know what the top models are when I'm done.

Thanks for the data,
Sinead

Patricia Wohner <pjwohner@gmail.com> Mon, Mar 30, 2015 at 7:34 AM
To: Sinéad Borchert <sineadborchert@gmail.com>

Sweet, you can publish that info if you want somewhere and just put me in the acknowledgements as I won't likely do anything with home ranges as there are so few sites but if you included it in your methods, at least it would be out there, Patti

[Quoted text hidden]

Sinéad Borchert <sineadborchert@gmail.com> Mon, Mar 30, 2015 at 5:49 PM
To: Patricia Wohner <pjwohner@gmail.com>

Alright, cool, thanks Patti. I'll include this in my thesis then, crediting you, and I'll let you know if it ultimately goes into a manuscript.

Sinead

[Quoted text hidden]

VITA

Sinéad Mary Jessie Burke Borchert was born in Sacramento, CA in 1986. As an only child with plenty of solitary time, she developed a love of nature while playing outside. She graduated from the University of California Santa Barbara in 2008 with a Bachelor of Science in Zoology. After graduation, she worked as a field biologist on many different projects located throughout North and South America, as well as on the island of Rota in the Northern Mariana Islands. She developed an interest in conservation after working with rare species and habitats, such as the Mariana Crow (*Corvus kubaryi*), Island Scrub-Jay (*Aphelocoma insularis*), and the western Yellow-billed Cuckoo (*Coccyzus americanus*), the latter of which depends on sparse riparian habitat. She returned to school to join Dr. Philip Stouffer's lab at Louisiana State University, where she was offered the chance to study the declining Rusty Blackbird and its wintering habitat for her master's thesis. She is expected to graduate in 2015.