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Evaluation of anuran richness in restored wetlands of central Louisiana

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EVALUATION OF ANURAN RICHNESS IN RESTORED WETLANDS OF CENTRAL LOUISIANA

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ABSTRACT

Bottomland hardwood forests and associated fauna, including frogs, are disappearing. The 1990 Farm Bill created a wetland restoration program on private lands called the Wetland Reserve Program (WRP) that has the potential to reverse the declines in species associated with bottomland hardwood forests. As of September 2005, nearly 85,000 ha had been enrolled in Louisiana, but the structure and value of these wetlands to frogs is not known. I evaluated 22 restored and 8 reference wetlands from January through May in 2004 and 2005 to determine the effects of local and landscape scale habitat characteristics on frog species richness and occurrence. I used chorus count surveys, egg mass searches, and dipnet surveys to detect frog species each season. Vegetation characteristics at each wetland were determined seasonally. I evaluated landscape influences by using aerial photography and satellite imagery of the sites to determine the surrounding land use. I used multiple linear and logistic regression analysis and t-tests to evaluate the effects of local and landscape variables on species richness and individual species occurrence. I detected 12 of the 13 species expected to occur. Frog species richness did not differ between restored and natural wetlands, but species richness was higher in 2004 than 2005 ($P < 0.0001$), presumably due to much greater amounts of rainfall in 2004. Species richness in 2004 was positively influenced by median water depth and canopy cover ($P = 0.0011$). In 2005, permanent flooding, median water depth, emergent and floating vegetation, and canopy cover positively influenced species richness ($P < 0.0001$). Species richness also increased with forest in the surrounding landscape. Bullfrogs and bronze frogs were associated with canopy closure, herbaceous vegetation, and nearby forest. Northern cricket frogs were associated

with shallow wetlands with floating vegetation, litter, and nearby forest. Gray tree frogs were found in wetlands with canopy cover, low emergent vegetation, and nearby agriculture. Restored wetlands in this study provided suitable frog habitat and supported similar frog species comparable to reference wetlands; however, additional frog and vegetation monitoring should be continued to evaluate restored sites throughout maturation.

CHAPTER I: INTRODUCTION AND METHODS FOR LOCAL AND LANDSCAPE SCALES

INTRODUCTION

The Mississippi Alluvial Valley (MAV) begins in southern Illinois and extends through parts of Missouri, Kentucky, Tennessee, Arkansas, Mississippi, and Louisiana. Glaciation was the single most important event that shaped this region (Saucier 1994). The response to glaciation included meanderings of the Mississippi River, formation of the floodplain, and sediment deposition throughout the region (Saucier 1994). More recently, the MAV has undergone severe anthropogenic alterations including widespread timber harvesting, channelization, and disconnection of the Mississippi River and its floodplain via the levee system (Rudis 1995).

Vegetation communities have changed through time as a result of changing climatic conditions (Delcourt and Delcourt 1984, King et al. 2005). The MAV, including Louisiana, was once covered with 10 million ha of bottomland forest; however, only 2.8 million ha remain (MacDonald et al. 1979, Dahl 1990, Rudis 1995). Much of this loss was due to extensive clearing for agriculture (MacDonald et al. 1979). Agriculture in the MAV not only cleared the forests, but leveled the land as well (Fredrickson 1997). Land leveling removed microtopography, which was responsible for a myriad of temporary wetlands in this landscape. The loss and alteration of bottomland hardwoods and associated wetlands is of international concern because of the number of species dependent on bottomland hardwood forests (Twedt and Loesch 1999).

Today, over 50% of forested wetlands remaining in the MAV are located in Louisiana (Twedt and Loesch 1999). Twedt and Loesch (1999) also found that approximately 12% of the MAV in Louisiana is bottomland hardwood forest with 87% in private ownership. Thus, conservation, restoration, and management of bottomland hardwood forests should include mechanisms to enhance these activities on private lands.

The Wetlands Reserve Program (WRP), administered by the United States Department of Agriculture (USDA) Natural Resources Conservation Service (NRCS), is a voluntary program designed to assist in the restoration of wetlands on private property. The landowners agree to restrict development and place a conservation easement on their property for 15 years, 30 years, or permanently. In exchange for the easement, WRP pays part or all of the restoration costs as well as an easement payment. WRP has approximately 595,292 ha enrolled in the program nationwide, including approximately 84,983 ha in Louisiana (NRCS 2006).

The WRP selection process is based on a number of rankings of property features. According to NRCS guidelines, one of the more important features of the property is the potential to support migratory birds, including waterfowl, songbirds, and other wetland birds. The potential of the property to provide habitat for declining species, as well as the water quality enhancement, floodwater retention, also results in higher rankings. Properties are also assessed for location, operational and maintenance issues, the extent of hydrology restoration needed to restore wetland functions, and the potential for the restoration to be achieved (NRCS 2006).

Initially, wetland restoration efforts in the WRP consisted of planting trees and plugging ditches with little regard for reconstructing microtopography that would support temporary wetlands. This was the most cost effective way to restore wetlands to a heavily modified agricultural landscape. Some wetlands were referred to as “walkaways” where ditches were plugged and revegetation was left for natural processes (Stratman

2000). Other than plugging ditches, most of these sites had little, if any, hydrologic restoration. At other sites, oak (*Quercus spp*.) trees were planted to replace the historic bottomland hardwood forests (Mark LaBourde pers. comm.). One survey stated that 77,698 ha had been reforested in the LMAV as of 1999, but the lack of hydrologic restoration at that time limited wetland functions as well as the composition of the wetland habitat (King and Keeland 1999). Stanturf et al. (2001) concluded WRP had largely failed in Mississippi because restoration did not restore wetland functions and it did not account for site-specific variability. More recently, WRP efforts feature a greater diversity of patterns in wetland design as well as a multitude of water depths, hydroperiods, and habitats for a greater suite of wildlife species (Stratman 2000).

Figure 1. A WRP property in Louisiana (photo courtesy of NRCS).

Presently, hydrologic restoration, or hydrologic rehabilitation, has become more common in WRP through the creation or restoration of macro- and microtopographic features. Macrotopography consists of large scale changes, whereas microtopography consists of small scale changes. Macrotopography techniques can involve establishing forests or creating impoundments capable of moist-soil management (Stratman 2000). Microtopography techniques usually add hydrologic features to the macrotopography design. Within impoundments/moist soil units, microtopography can be created by adding sloughs or deeper enhancements which hold water throughout the year. This technique creates more diverse hydroperiods and sources of water for various parts of the year. In reforested tracts, topography can be created such that the land ponds water, or trees may simply be planted in low areas.

Impoundments/moist-soil units are constructed by placing levees around an area that has a low slope and relatively impervious soils (Fredrickson and Taylor 1982). Water-control structures are placed in the levees to allow water-level manipulation to facilitate germination and growth of desirable plant species. Water levels vary based on objectives, although a major focus of many landowners is to provide habitat for wintering waterfowl. As seasonal wetlands are destroyed by various land use practices, moist-soil management may also provide habitat for displaced marsh birds, wading birds, and other wildlife (Fredrickson 1996).

The increase in hydrologic restoration in WRP is encouraging as improved wetland functions are expected. However, there is little evaluation of the effectiveness of this process in providing suitable wildlife habitat and other wetland functions. Amphibians, particularly frogs, are a group of species often used as an indicator of wetland restoration success (Semlitsch 2003). Some species of amphibians are sensitive to water quality parameters like pollution levels and dissolved oxygen in a wetland system (Semlitsch 2003). Salamanders can account for twice the biomass of birds and the same biomass as small mammals in certain ecosystems (Burton and Likens 1975b). Moreover, amphibians are an essential component of most wetlands systems due to their crucial role in the food web, serving as both predator and prey (Kline 1998).

Amphibians have become a topic of much concern due to worldwide declines (Phillips 1990, Wake 1991). However, due to their long life cycles and natural fluctuations in breeding populations, it is not clear whether observed declines are natural or whether they are negative responses due to human influence, particularly habitat loss (Pechmann et al. 1991, Blaustein et al. 1994, Lehtinen et al. 1999, Bosch et al. 2004). It is thought that wetland loss and degradation, diseases, and habitat fragmentation have played a large role in the decline of amphibians (Kolozsvary and Swihart 1999, Findlay and Bourdages 2000, Hels and Buchwald 2001, Beebee and Griffiths 2005). Furthermore, the worldwide declines mentioned suggest that these species could potentially benefit from programs such as WRP.

Frogs often require terrestrial as well as wetland habitat at varied scales to complete their annual cycle (Dundee and Rossman 1989). At the local scale, hydroperiod, vegetation structure, and wetland size influence frog populations. Hydroperiod is defined by the duration of flooding on any particular site over a period of time. Numerous studies indicate that hydroperiod can be the single most important determinant of frog community structure (Rowe and Dunson 1995, Dodd and Cade 1998). Brown (1974) suggested that naturally impounded water provided the best breeding sites for amphibians. In Pennsylvania, Rowe and Dunson (1995) found hydroperiod to be the greatest factor in determining what amphibian species were present and the reproductive success of those species. Also, they also found that longer hydroperiods are not necessarily beneficial to amphibians (Rowe and Dunson 1995). On the contrary, smaller wetlands with shorter hydroperiods are crucial for many species of amphibians because they rarely sustain fish populations, a main predator of many amphibian species (Semlitsch and Bodie 1998, Tiner 2003). Babbitt and Tanner (2000) demonstrated that amphibians in wetlands with intermediate hydroperiods had the highest overall reproductive success and at some point, all species of anurans present bred in these ponds. Few amphibian species in their study bred in wetlands with particularly short or long hydroperiods (Babbitt and Tanner 2000). In a similar study, intermediate pools provided the highest survival of tadpoles due to lower predation and sufficient time for development of larvae (Smith 1983). Larval amphibians have also been shown to be the most sensitive to hydroperiod disruption and their survival is linked to hydroperiod (Babbitt and Tanner 2000, Pechmann et al. 2001).

Hydroperiod can directly influence wetland vegetation as well. Vegetation structure within the wetland and surrounding lands is important to many species of frogs (Dundee and Rossman 1989, Semlitsch 2003). The emergent zones support different vegetation communities depending on the age of the wetland and timing of water level manipulation (Harris and Marshall 1963). In northwestern Minnesota, a slow, 5-yr drawdown with a long hydroperiod allowed undesirable species such as willow (*Salix spp.*) and even aspen (*Populoides spp.*), with rapid establishment traits, to take over a wetland and overwhelm many of the moist soil plants (Harris and Marshall 1963). Rapid drawdowns with short hydroperiods over a few days can raise the temperature of the soil so as to allow species of low value to some wildlife like coffeeweed (*Sesbania spp.*) and cocklebur (*Xanthium spp*.) to become established (Fredrickson and Taylor 1982). In Louisiana, moist soil plants have varied habitat characteristic requirements, particularly with wetland soils and temperatures. For example, smartweeds (*Polygonum spp*.) require an early drawdown with lower soil temperatures whereas millets (*Echinochloa spp*.)

require mid-season drawdowns with intermediate temperatures (Fredrickson and Taylor 1982). However, little is known about vegetation structure and composition in moist soil units and the influence on amphibian communities.

Semlitsch (2003) linked wetland vegetation structure and composition at the wetland edge to abundance and diversity of amphibians. Most tadpoles consume vegetable matter and may benefit from aquatic plants and zooplankton (Dundee and Rossman 1989). The emergent vegetation at the wetland edge may provide areas for egg sac attachment for some species of amphibians as well as calling sites for several species of tree frogs (Dundee and Rossman 1989). Some anurans such as cricket frogs (*Acris crepitans*) and several toads (*Bufo spp.*) rely on edge emergent vegetation for cover as well as for locating prey such as insects, spiders, and other invertebrates (Stumpel and van der Voet 1998, Dundee and Rossman 1989).

In addition to vegetation within the wetland, the terrestrial zone surrounding a wetland is also important to amphibians (Gibbons 2003, Semlitsch and Bodie 2003, Trenham and Shaffer 2005). A large portion of many frog life cycles is spent in non-breeding habitats (Dundee and Rossman 1989). Frogs also have relatively small home ranges (Gibbons 2003); thus, adjacent upland habitat near breeding wetlands is important for frogs to complete their life cycle (Trenham and Shaffer 2005). Large terrestrial buffer zones protect the migration of amphibians to upland habitat as well as secure forest input to the wetlands (Dodd 1996, Guerry and Hunter 2002). The literature varies on the effects of size of terrestrial buffer zones on frogs (Semlitsch and Bodie 2003); however, terrestrial zones in general provide many benefits, such as foraging habitat and routes to nearby wetlands (Gibbons 2003). In turn, reforested wetlands are heavily used by amphibians (Petranka et al. 2003) and restored wetlands have been used specifically by frogs as well (Stevens et al. 2002).

Conservation and value of wetlands is often determined by size; however, amphibian richness has not been entirely linked to wetland size (Babbitt and Tanner 2000, Snodgrass et al. 2000). Babbitt and Tanner (2000) noted that amphibian species richness increased with size, but Snodgrass et al. (2000) observed that species found at smaller, isolated wetlands are not necessarily a subset of the richness associated with larger wetlands. In addition, one study on the Ordway Preserve in Florida found small, isolated wetlands to host more total species as well as more species per site of amphibians (Moler and Franz 1987). A study in Minnesota showed that newly restored wetlands were colonized by some anurans but there were nearly 25% fewer species than in natural control wetlands (Lehtinen and Galatowitsch 2000). Also, this study showed that species richness was linked to distance to a source pond and that no species of amphibian was found only in restored wetlands (Lehtinen and Galatowitsch 2000).

Connectivity should be evaluated because without wetland connectivity, particularly to newly created or restored wetlands, the low dispersal rates of many species could slow or prevent colonization of new wetlands (Dodd 1996, Semlitsch and Bodie 1998, Skelly et al. 1999). When wetland connectivity is low, the probability of amphibian dispersers encountering small and disconnected wetlands is also low (Lehtinen and Galatowitsch 2000). A multi-scale study indicated species richness of amphibians decreased as wetland connectivity decreased in both urban and agricultural landscapes (Lehtinen et al. 1999). The success of juvenile dispersal is the key for many populations of amphibians to survive (Semlitsch 2000, Guerry and Hunter 2002).

The response of amphibians, and particularly frogs, to varied connectivity has been mixed. Long-toed salamanders (*Ambystoma macrodactylum*) and Pacific treefrogs (*Hyla regilla*) were not sensitive to landscape connectivity of artificial ponds in Idaho, but these two species have high dispersal rates and general habitat needs (Monello and Wright 1999). However, tiger salamanders (*Ambystoma tigrinum*) and western toads (*Bufo boreas*) never consistently utilized these artificial ponds, likely due to the low connectivity of the ponds as well as their more specific local habitat requirements (Monello and Wright 1999). Wetland connectivity and surrounding land use on the landscape scale can also influence success of frog populations in restored and created wetlands (Knutson et al. 1999, Pope et al. 2000). A study showed that common frogs and toads efficiently colonized newly constructed ponds within agricultural fields whereas great crested newts (*Triturus cristatus*) and smooth newts (*Triturus vulgaris*) occupied them at a significantly lower rate (Baker and Halliday 1999). More specifically, several species of frogs have been shown to avoid certain landscape features such as fields, pastures, clearcuts, and roads, as these features lower wetland connectivity (Rothermel and Semlitsch 2002, Marsh et al. 2004). Also, frogs suffered higher mortality rates from predation and desiccation while emigrating to new ponds through the unconnected landscape (Rothermel and Semlitsch 2002). Many studies have shown a positive association with presence and abundance of amphibians and the area of forest surrounding the breeding habitat as well as the proximity to forest (Knutson et al. 1999, Guerry and Hunter 2002). Landscape level composition and distribution can greatly affect amphibian populations although a recent review paper suggested that more research is needed at this scale (Cushman 2006). Without wetland connectivity, amphibian species with low dispersal rates and specific habitat needs may not benefit from pond construction (Baker and Halliday 1999).

Connectivity to terrestrial zones is a key aspect for amphibian conservation as well. However, the common restoration technique of creating corridors has not been indicated to be effective for amphibians. Wide forested corridors were not shown to support a greater number of reptiles and amphibians (Burbrink et al. 1998). Some research has even demonstrated that forested corridors were not used consistently by amphibians (Dodd and Cade 1998).

Evaluating restored and rehabilitated wetlands and their ability to provide functions similar to natural wetlands can improve restoration techniques and the overall success of wetland restoration. The objectives of this study were to: 1) determine the effects of local and landscape level habitat factors on frog species richness and individual frog species occurrence, and 2) compare habitat characteristics influential to frogs among restored and reference wetlands in east central Louisiana.

METHODS

Study Area

This study examined WRP sites in east central Louisiana. The historic landscape of this area was almost entirely bottomland hardwood forest. Backwater flooding from the Red River was the dominant flooding source, although wetlands within the bottomland hardwood forests had diverse hydroperiods and vegetation structure. Similar to other regions of the MAV, agricultural clearing and land leveling erased many historic lowlands and natural sloughs (Fredrickson 2002). Because of diverse agricultural impacts and landscape settings, a wide range of techniques have been used to restore wetlands in this region. Techniques used in Louisiana, however, are similar to those in other regions of the MAV.

I selected 22 wetlands enrolled in WRP (Table 1) and 8 reference wetlands in the Lake Ophelia National Wildlife Refuge (LONWR) (Table 2). The WRP tracts are located in Avoyelles Parish, south of LONWR. Restored wetlands were chosen based on landowner permission, location, and local and landscape habitat characteristics. The eight reference wetlands were chosen because of diverse local and landscape habitat characteristics and proximity to WRP tracts. This experimental design was ad hoc and not random due to constraints imposed by the WRP and reference wetlands available for study in Avoyelles Parish.

Restored wetlands ranged in size from 0.9 ha to 173.5 ha and in age from 1 year to 18 years. Restored wetlands were broadly classified into 1 of the following 4 categories: 1) reforested tracts; 2) impoundments/moist soil units; 3) dredged natural wetlands; and 4) created wetlands.

There were 4 reforested easements in which oaks were planted and some microtopography was restored on the landscape. Reforested wetlands were chosen across a broad age class (6 to 18 yrs) and varied water-holding capabilities. Two reforested sites held water through the frog breeding season and two reforested sites held water for only a short time (1-3 weeks) after a heavy rain

Table 1. Summary of WRP sites. Wetland categories are as follows: R (reforested), I (impoundment), DN (dredged natural), and C (created). Flooding is T (temporary) or P (permanent). Vegetation refers to dominant wetland vegetation.

Wetland Name	Wetland Category	Flooding	Vegetation	Size (ha)	Age (yrs)
Dupuy 1	I	P	Willow; aquatic	0.9	1
Dupuy 2	I	P	Nearly absent	0.6	1
Dupuy 3	C	P	Aquatic	0.6	1
Jimmy lake	DN	P	Thick aquatic, floating	173.5	8
Juneau lake	DN	P	Thick aquatic, floating	158.0	5
Juneau 1	C	P	Floating	4.0	5
Juneau 2	C	P	Floating	2.0	5
LONWR 1	C	T	Emergent	0.4	8
LONWR 2	R	T	Emergent in low spots	0.5	8
McCann 1	DN	P	Heavy floating, some emergent	5.2	18
McCann 2	R	T	Emergent in low spots	2.0	18
McCann 3	C	T	Little floating	0.6	18
Roseau 1	I	P	Thick emergent	17.5	8
Roseau 2	R	T	Emergent in low spots	3.5	8
Roseau 3	C	P	Emergent	0.5	8
Smith	I	P	Thick floating, aquatic	8.0	6
Steele 1	C	P	Emergent, floating	0.3	8
Steele 2	C	P	Emergent, floating	0.3	8
Steele 3	I	P	Thick emergent	1.5	8
Steele 4	I	P	Emergent	48.0	8
Steele 5	R	T	Emergent in low spots	2.0	8
Wolf prairie	C	P	Thick emergent, aquatic	9.0	8

Table 2. Summary of reference sites. Flooding is T (temporary) or P (permanent). Vegetation refers to dominant wetland vegetation.

Wetland Name	Flooding	Vegetation	Size (ha)
Doom's lake	P	Emergent	3.6
Duck lake	P	Thick emergent, aquatic	29.5
Narrow band	T	Nearly absent	11.4
Point Bosse	P	Thick emergent, floating	18.3
Temp pond 1	T	Thick emergent	1.6
Temp pond 2	T	Emergent, floating	0.3
Westcut lake	P	Thick emergent, floating	14.2
Willow slough	T	Emergent	11.2

. Six large impoundments with a 5:1 slope were studied. Most of these wetlands were rectangular and enclosed by levees (≤ 6 m tall), limiting connectivity to the Red River as well as suppressing backwater flooding. There was a wide range of variability among impoundments in terms of hydrologic management and impoundment size and structure. Some impoundments incorporated enhancements, which are borrow pits permanently holding water and creating a small mound of higher ground. Water levels in all impoundments could have been manipulated seasonally; however, in both study years, no traditional moist soil management occurred during the frog breeding season. Three moist soil units had water level manipulation after the frog breeding seasons in July and August. There were also impoundments with and without enhancements and a variety of size and age among impoundments for further comparisons.

The 3 dredged wetlands were wetlands that had problems with sedimentation at the time they were entered into the easement. These wetlands were then dredged using large farm machinery to ensure the wetlands would continue to hold water year-round as they had historically. However, after the dredging occurred, no other management was in operation on these wetlands.

There were 8 created wetlands ringed by levees. Management regimes varied from no management to intensive annual vegetation manipulation, through activities such as disking.

The reference wetlands were located on LONWR, a 7,082 ha wildlife refuge located near Marksville, Louisiana. The area was largely cleared in the 1970's for agriculture, but in 1988 LONWR was formed to create waterfowl habitat and protect and restore bottomland hardwood forests (USFWS 2000). The landscape is dominated by ridge and swale topography and oxbow lakes, reforestation (20%), mature bottomland hardwood forest (50%), and active agriculture (25%); the remaining 5% is either permanently under water or part of a moist soil management regime (USFWS 2003). The forested wetlands are dominated by baldcypress (*Taxodium distichum)*, willow (*Salix spp.)*, honey locust (*Gleditsia triancanthos*), oaks (*Quercus spp.*), sugarberry (*Celtis laevigata*), and water tupelo (*Nyssa aquatica*). Historically, the area flooded frequently from the Red River but flood control measures have drastically reduced overbank flooding. Since 1990, LONWR has severely flooded twice and moderately flooded 5 times.

I selected 4 permanently flooded lakes, 2 semi-permanent willow (*Salix spp.*) sloughs, and 2 temporarily flooded wetlands for intensive study. These sites ranged in size from 0.3 ha to 29.5 ha. Two of the 4 lakes were surrounded by reforested tracts, and

2 were surrounded by agriculture. Vegetation within the lakes ranged from very little vegetation to dense emergent and floating aquatic vegetation.

Field Methods

I determined frog richness in each wetland through the use of chorus counts (Heyer et al. 1994), egg mass searches, and dipnetting. The dominant survey consisted of a count of chorusing male frogs at night within 72 h of a rain event. Chorus counts were conducted at the edge of the wetland, beginning a half hour after sunset and ending at midnight. All species of frogs calling were recorded. Chorus counts were conducted twice a season or more if possible. Seasons were defined as winter (Dec-Feb), spring (Mar-Apr), and summer (May-June). Chorus counts were not conducted with lights or when temperatures were below 6° C.

Amphibian egg mass searches were conducted once per season, following chorus counts, along the edge of each wetland as an estimate of frog reproduction (Heyer et al. 1994). The 100-m transect was placed on the long axis of the wetland and bisected the chorus count listening station. The masses were counted and identified, if possible, but not collected. These searches were constrained to 30 min so as to equally represent each wetland.

Time-constrained dipnet surveys were conducted monthly during daylight along the 100-m transect (Heyer et al. 1994). The vegetation was scraped and netted for 30 minutes at each wetland. All larval frogs were counted and identified. All three surveys were conducted to determined overall species richness in each site; relative abundance of each species was not addressed due to time constraints and lack of intensive surveys necessary to determine abundance.

Vegetation structure was determined by surveys along the wetland edge, and transects into the wetland and surrounding upland. Vegetation surveys were conducted once per season along the same 100-m transect as the above mentioned surveys (Figure 1). Every 10 m along the transect, a 20-m transect extended perpendicular to the original transect into the wetland. Along this 20-m perpendicular transect, measurements were taken every 5 m; however, from 0 m to 5 m, measurements were taken every 1 m to ensure a concentration of sampling along the shoreline. The measurements were: canopy presence/absence, water depth, open water, woody debris, and percent cover of emergent, floating, and aquatic vegetation. The percent cover was determined by using a PVC pipe with 2 strings taped perpendicular to each other in order to create 4 quarters of view. The 20-m transects alternated in direction either extending into the wetland or upland in order to equally sample the terrestrial zone as well as the wetland. The percent cover in the surrounding upland consisted of measuring all the previously mentioned habitat categories if present, as well as several additional habitat categories, including litter, bare ground, herbaceous vegetation, and woody debris. Flooding was determined by the presence or absence of water at each vegetation survey. If water was absent at any vegetation survey, the flooding was considered temporary. If water was present at all vegetation surveys, the flooding was considered permanent. This was evaluated separately each year as climatic events greatly influenced the presence or absence of standing water in each wetland.

The landscape variables included in this study described the surrounding land use of each site which would be available to dispersing frogs (Semlitsch and Bodie 2003). The measurements were: percent agriculture, percent forest, and percent reforested within a 1-km^2. Reforested habitat was defined by trees planted less than 20 years ago. Forest habitat was defined by trees planted more than 20 years ago. The distance to closest forest was also measured. The measurements for the WRP sites were calculated from black and white aerial photographs (February, 2004), using a 2.54 cm to 201.17 m scale (1 in to 660 ft). These were obtained through the NRCS office in Marksville, Louisiana. The reference sites were measured from color satellite imagery taken in 2003 at the scale of 2.54 cm to 562.05 m (1 in to 1844 ft) (Google Earth 2005).

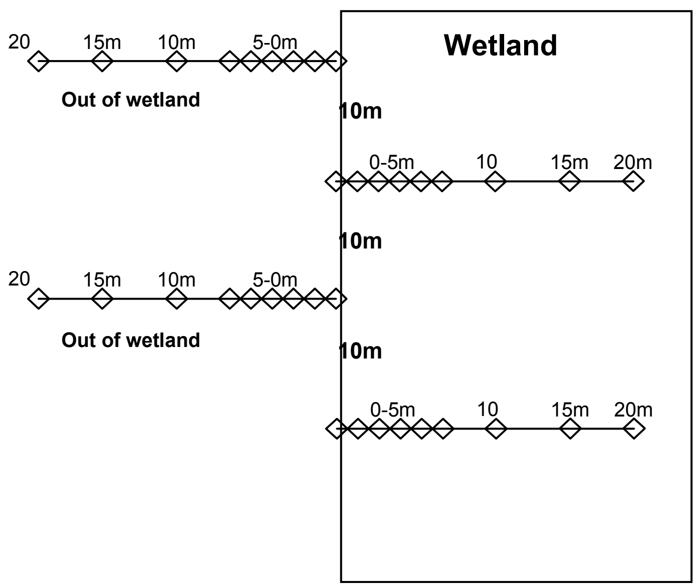

Figure 2. Illustration of vegetation survey methods repeated seasonally for both years of the study. The transect was 100 m in length along the edge of the wetland; this figure shows only 40 m of the transect.

Statistical Analysis

I evaluated the effects of local and landscape habitat characteristics on frog species richness and individual species occurrence with multiple linear and logistic regressions. I used SAS (SAS Institute Inc., Cary, North Carolina) for all statistical analysis. Stepwise selection regression was used to establish relevant and biologically interpretable models. Logistic regression was also used along with t-tests to evaluate models and compare data.

Correlation analysis was used to remove redundant habitat variables. This resulted in the removal of habitat variables sampled at 10, 15, and 20 m (Figure 1) along the 100-m transect, as they were strongly correlated (Pearson correlation coefficient > 0.60) with similar edge variables and diluted the edge variables (e.g., edge floating with non-edge floating vegetation) (Appendix I). There were two variables, edge emergent and edge aquatic vegetation, that were associated with their non-edge counterparts less than the 0.6 standard. However, when analysis included the two non-edge variables, the edge variables were never found to be influential and thus were not used in further analyses. This resulted in 13 local habitat variables used in analysis and the removal of all non-edge variables (Table 3; Appendix I). The habitat variables were collected seasonally; therefore, the corresponding maximum frog species richness total was calculated by adding all species detected at either 1 or both surveys for that particular season. All sites were visited twice per season. For example, if a bullfrog was heard at site #1 in one spring survey and not in the second spring survey, it was still counted towards the species richness total for that seasonal survey. Any species detected with dipnetting and egg mass searches were also counted towards total species richness.

This study occurred over two consecutive years during the frog breeding season. Due to dramatic climatic differences between years, year was tested for statistical difference ($P < 0.05$) using a t-test (proc TTEST) and a test for interaction with season (proc MIXED).

Table 3. Definitions of environmental and local habitat variables used in multi-model analyses with anuran richness totals and species occurrence. The u indicates the duplicate wetland variable measured in the upland.

Variable	Abbreviation	Definition
Year	1 = 2004	Year of study
	2 = 2005	
Flooding	1 = permanent	Duration of water at site
	2 = temporary	
Type	1 = restored	Type of wetland
	2 = reference	
Canopy	Edgecan	Percent of canopy closure
	uedgecan	
Woody debris	Edgedead	Percent of woody debris
	uedgedead	
Litter	uedgelitter	Depth of litter (cm)
Bare ground	Edgebare	Percent of bare ground
	uedgebare	
Herbaceous vegetation	Edgeveg	Percent of herbaceous vegetation
	uedgeveg	
Water depth	Medianw	Median water depth
Emergent vegetation	Edgeemg	Percent emergent vegetation
Aquatic vegetation	Edgeaq	Percent submerged aquatic vegetation
Floating vegetation	Edgefloat	Percent floating vegetation
Open water	Openw	Percent of open water
	Uopenw	
Maximum richness	Maxrich	Total species richness per survey

To determine the effects of local vegetation structure, landscape characteristics, and flooding on species richness, I used a stepwise regression analysis (proc REG) on full models including all local and landscape variables remaining from the variable reduction. Variables were only entered into the model if they were significant ($P < 0.05$) and all other variables in the model remained significant ($P < 0.05$). Models were examined for normality (Shapiro Wilkes $P < 0.05$) and variance inflation issues. I used summary statistics (proc UNIVARIATE) to report means and standard errors for all variables analyzed. Local habitat variables were analyzed separately from landscape variables. The landscape variables used in analysis were: distance to closest forest, percent forest, percent reforested, and percent agriculture within 1-km^2 (Table 4).

Table 4. Definitions of landscape variables used in multi-model analyses with frog species richness totals and species occurrence.

Variable	Abbreviation	Definition
Percent forest	Forest	Percent of forest within 1-km^2
Percent reforested	Ref	Percent of reforested forest within 1-km^2
Percent agriculture	Ag	Percent of agriculture within 1-km^2
Distance	Km	Distance to closest forest

The next step in analysis was to determine the influences of local and landscape habitat variables and flooding on occurrence of several frog species recorded or seen in this study. All frog species detected at 60% or more of the total visits were included in the analysis and tested with each other for correlation. This percentage was used to ensure enough overall detection of the species to render associations with the habitat variables. The frog species analyzed were: bullfrog (*Rana catesbeiana*), gray tree frog complex (*Hyla versicolor/chrysoscelis* complex), northern cricket frog (*Acris crepitans*), and bronze frog (*Rana clamitans clamitans*). The analysis involved creating a detection history for the frog species at each season, corresponding to the local and landscape habitat variables and flooding. The detection was either 0 (not detected) or 1 (detected). Logistic regression (mixed logistic regression; proc LOGISTIC) for a mixed model was then used to analyze the relationship among habitat variables, including flooding type and individual species occurrence ($P < 0.05$).

I compared species richness among restored and reference wetlands using a t-test (proc TTEST). I used either the pooled variance or the Satterthwaite results, depending upon the homogeneity of variance of the data ($P < 0.05$). I used summary statistics (proc UNIVARIATE) to report means and standard errors for species richness results for restored and reference wetlands.

To determine if habitat characteristics differed among restored and reference wetlands, I compared a subset of the habitat variables among these wetland types. The habitat variables included in the analysis were those established as influential to frog species richness and individual frog species occurrence by the previous statistical analyses. I used a t-test (proc TTEST) and also tested the homogeneity of variance and reported the appropriate results ($P < 0.05$). T-tests were used because it did not require the assumptions of normality or homogeneity of variance to be met. This data set was non-normal and did not consistently exhibit homogeneous variance.

CHAPTER II. RESULTS OF ANURAN AND HABITAT ANALYSIS AT LOCAL SCALE

RESULTS

Habitat Characteristics Analysis on Species Richness

Precipitation during the sampling periods between years varied greatly. The first study year, 2004, was an extremely wet year. Avoyelles Parish and the surrounding area recorded 57.82 cm of rain for May and June combined and 161.85 cm total (NOAA 2006). The second study year, 2005, was a drought year with Avoyelles Parish and surrounding area receiving only 88.4 cm of rain for the year (NOAA 2006). This disparity between years likely influenced the results of this study.

There are 13 species of frogs known to occur in Avoyelles Parish, Louisiana (Dundee and Rossman 1989) (Table 4). I detected 12 species of frogs; only the American toad (*Bufo americanus*) was expected to occur but was not detected (Figure 2; Figure 3). The bullfrog (*Rana catesbeiana*), gray tree frog complex (*Hyla versicolor/chrysoscelis*), northern cricket frog (*Acris crepitans*), and the bronze frog (*Rana clamitans clamitans*) all occurred in over 60% of visits to all wetlands. The Gulf coast toad (*Bufo valliceps*), Woodhouse's toad (*Bufo woodhousei*), and the upland chorus frog (*Psuedacris triseriata*) occurred in <10% of visits to all wetlands. Species that are strongly seasonal like the spring peeper (*Psuedacris crucifer*) and the southern leopard frog (*Rana sphenocephala*) were detected frequently during the winter of 2005 and early spring of both study years, but were rarely detected with chorus counts outside of their peak breeding season. Only the Upland chorus frog was detected in a created wetland and not in any reference wetlands.

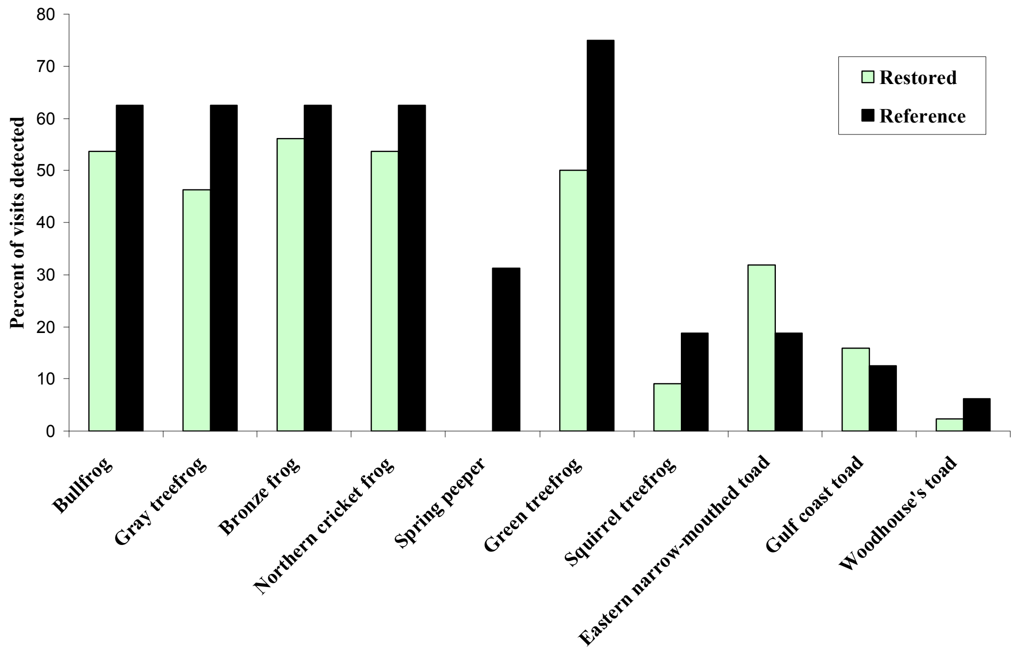

Figure 3. The percent of visits (40 total visits in reference sites; 104 total visits in restored sites) with detection for each frog species recorded during chorus counts at restored and reference wetlands in Avoyelles Parish, LA, 2004. No error bars were included because only one statistic was analyzed and thus, had no variation.

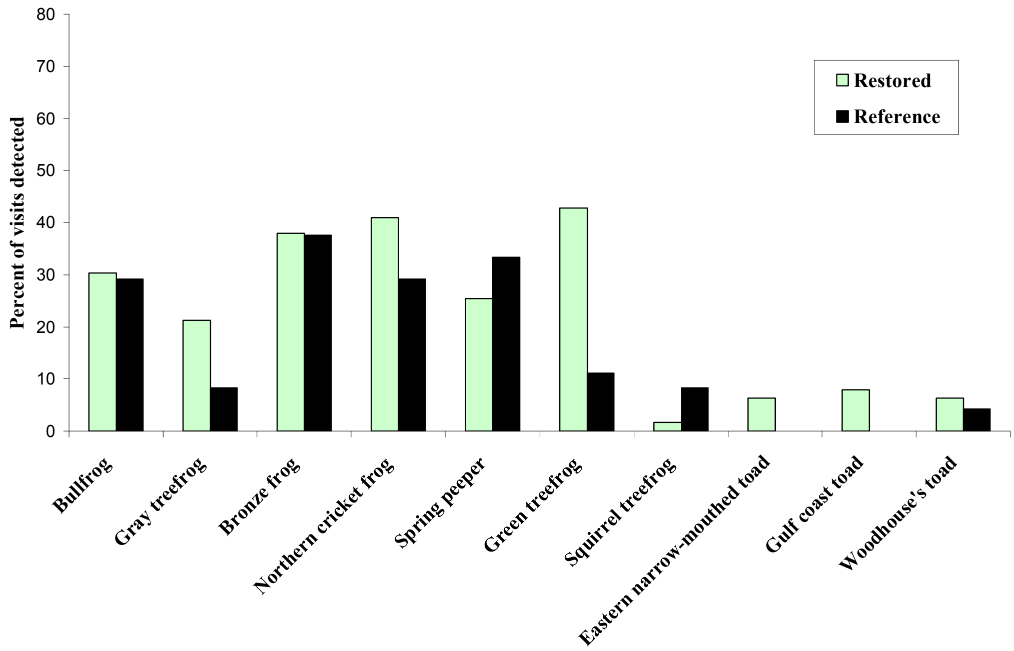

Figure 4. Percent of visits (48 total visits in reference sites; 132 total visits in restored sites) with detection for each frog species recorded during chorus counts at restored and reference wetlands in Avoyelles Parish, LA, 2005. No error bars were included because only one statistic was analyzed and thus, had no variation.

All species detected were detected with chorus counts; the other survey techniques, egg mass searching and dipnetting did not detect any species outside of those recorded during the chorus counts (Table 4). No egg masses were detected during this study. A total of 7 species were visually observed during site visits.

Table 5. Frog species expected to be encountered in Avoyelles Parish, Louisiana, and method of detection for each species during 2004 and 2005.

Scientific Name	Common Name	Chorus Counts	Dipnet Surveys	Visual Observation
Rana catesbeiana	Bullfrog	X	X	X
Rana clamitans clamitans	Bronze frog	X		X
Rana sphenocephala	Southern leopard frog	X	X	X
Psuedacris crucifer	Spring peeper	X		
Psuedacris triseriata	Upland chorus frog	X		
Hyla cinera	Green tree frog	X	X	X
Hyla versicolor/chrysoscelis	Gray tree frog complex	X		X
Hyla squirella	Squirrel tree frog	X		X
Acris crepitans	Northern cricket frog	X	X	X
Gastrophryne carolinensis	Eastern narrow-mouthed toad	X		
Bufo valliceps	Gulf coast toad	X		
Bufo woodhousei	Woodhouse's toad	X		
Bufo americanus	American toad			

Species richness per survey differed between years ($F_{1,146} = 21.01$; $P < 0.0001$); in 2004, the mean maximum species richness was higher (3.74 ±1.8) than in 2005 (2.39 ±1.7). Regression analyses indicated that species richness exhibited a year and season interaction. Only species richness in spring 2004 and summer 2005 were similar; all other year and season combinations were statistically different (Table 6). Therefore, analyses were conducted for each year individually rather than with one global model to further determine the influence year had on any and all variables. Also, season was not kept in any further analysis so that the interaction of year and season would not disguise the impact of year alone on the results of this study.

Table 6. Year by Season interaction demonstrating associations of similar species richness totals by season. Estimates sharing a letter do not differ (P < 0.05).

year	season	Estimate & Grouping	95% C.I.
2004	Summer	1.42 A	1.25 - 1.59
2004	Spring	1.21 AB	1.01 - 1.40
2005	Summer	1.11 AB	0.91 – 1.31
2005	Spring	0.94 B	0.73 – 1.16
2005	winter	0.45 C	0.17 – 0.72

The global model tested for 2004 is listed in Table 7. The species richness in 2004 was positively affected by two variables: edge canopy and median water depth ($F = 7.74$; $r^2 = 0.22$; $P = 0.0011$) (Table 8). A full variable summary for 2004 is in Appendix II.

Table 7. Global model for 2004 and 2005 regression analyses, including all local habitat variables and flooding.

Dependent variables	Independent variables
Species richness	Flooding
Individual species occurrence	Median water depth
	Edge emergent vegetation
	Edge aquatic vegetation

Table 7 cont.

Dependent variables	Independent variables
Species richness	Edge floating vegetation
Individual species occurrence	Open water
	Edge canopy
	Edge woody debris
	Edge bare ground
	Edge herbaceous vegetation
	Upland edge canopy
	Upland edge woody debris
	Upland edge bare ground
	Upland edge herbaceous vegetation
	Upland edge litter depth

Table 8. Results for 2004 stepwise regression with variable summaries by wetland type.

		Restored			Reference			
Variable	Estimate ± SE	low	high	mean	low	high	mean	Pr>F
Median water depth (m)	0.018 ± 0.005	0	150	39.1	0	63.4	21.02	0.01
Edge canopy closure (%)	0.017 ± 0.007	0	100	23.68	40	100	81.93	0.002

The global model for 2005 is listed in Table 7. The best model included 5 variables: permanent flooding, median water depth, edge floating vegetation, edge emergent vegetation, and edge canopy ($F = 9.18$; $r^2 = 0.35$; $P < 0.0001$) (Table 9). Median water depth was the only variable with a negative influence on maximum species richness. Full variable summary for 2005 is in Appendix III.

Table 9. Results for year 2 (2005) stepwise regression with variable summaries by wetland type.

Variable	Estimate (SE)	Restored low	Restored high	Restored mean	Reference low	Reference high	Reference mean	Pr>F
Permanent flooding	1.56 ± 0.41	-	-	-	-	-	-	0.0003
Median water depth (m)	-0.01 ± 0.005	0	150	40.11	0	43.4	17.89	0.01
Edge floating veg. (%)	0.03 ± 0.01	0	73.3	12.27	0	30	3.23	0.002
Edge emergent veg. (%)	0.04 ± 0.14	0	59.3	11.29	0	22.3	6.1	0.04
Edge canopy closure (%)	0.01 ± 0.004	0	100	32.43	40	100	77.1	0.082

At the landscape scale, species richness for both study years was positively influenced by the percent of forest within 1 km^2 of a wetland (P = 0.006) (Table 10). Percent reforested (P = 0.18), percent agriculture (P = 0.90), and distance to closest forest (P = 0.27) did not affect species richness. The full summary statistics of the landscape variables can be found in Appendix IV.

Table 10. Results of landscape analysis on species richness for both study years.

Variable	Estimate (SE)	Restored low	Restored high	Restored mean	Reference low	Reference high	Reference mean	Pr>F
Forest (%)	0.017 ± 0.006	0	70.0	21.36	10.0	80.0	40.0	0.006

Habitat Characteristics Analysis with Occurrence of Individual Species

Analysis of the occurrence of individual species indicated that bullfrogs were associated with sites with upland canopy closure, upland herbaceous vegetation and upland litter ($P < 0.1$), most likely because many permanently flooded sites had fully vegetated uplands. Bullfrogs were not detected as frequently at sites with high debris, mostly open water, and bare ground (Table 11). Bullfrogs were also associated with sites surrounded by forest and reforested in the surrounding landscape ($P < 0.1$). Although permanent flooding did not account for significant variation in the analysis, bullfrogs were not detected at any temporary wetlands. Similarly, bronze frogs were associated with canopy closure and upland herbaceous vegetation ($P < 0.1$). Bronze frogs were adverse to sites with bare ground and mostly open water ($P < 0.1$). Bronze frogs were also associated with sites surrounded by forest, but were negatively associated with sites surrounded by agriculture ($P < 0.1$) (Table 11).

Gray tree frogs were associated with sites with high canopy closure, but were negatively associated with emergent vegetation and upland debris ($P < 0.1$). Similar to bronze frogs, gray tree frogs were negatively associated with sites surrounded by agriculture ($P < 0.15$) (Table 11).

Northern cricket frogs were associated with sites with floating vegetation and upland litter; however, northern cricket frogs were negatively associated with a higher median water depth ($P < 0.1$). Similar to the ranids, northern cricket frogs were associated with sites surrounded by forest and reforested on the landscape ($P < 0.1$) (Table 11).

Table 11. Results logistic regression analysis of local and landscape variables on individual frog species occurrence ($P < 0.1$). Upland habitat characteristics are indicated by a U before the variable name.

Frog species	Local variables	Estimate	Pr>chi sq	Landscape variables	Estimate	Pr>chi sq
Bullfrog	Open water (m)	-0.02	0.0028	Mature (%)	0.04	<0.0001
	U canopy (%)	0.008	0.07	Reforested (%)	0.02	0.003
	Edge debris (%)	-0.04	0.013	-	-	-
	U herbaceous veg. (%)	0.02	0.064	-	-	-
	Edge bare ground (%)	-0.035	0.048	-	-	-
	U litter (cm)	0.64	0.07	-	-	-
Gray tree frog	Edge emergent veg. (%)	-0.05	0.026	Agriculture (%)	0.012	0.10
	U debris (%)	-0.033	0.0006	-	-	-
	Edge canopy closure (%)	0.013	0.013	-	-	-
Bronze frog	Edge bare ground (%)	-0.034	<0.0001	Agriculture (%)	-0.02	0.05
	Open water (%)	-0.02	0.053	Mature (%)	0.03	0.0004
	Edge canopy closure (%)	0.02	0.0004	-	-	-
	U herbaceous veg. (%)	0.02	0.015	-	-	-

Table 11 cont.

Frog species	Local variables	Estimate	Pr>chi sq	Landscape variables	Estimate	Pr>chi sq
Northern cricket frog	Median water depth (m)	-0.012	0.099	Reforested (%)	0.016	0.04
	Edge floating veg. (%)	0.04	0.008	Mature (%)	0.03	0.0008
	U litter (cm)	1.12	0.003	-	-	-

When all four frog species were tested for correlation, only the bullfrog and the bronze frog showed strong correlation (>0.6). All other species tested were below 0.6 and not considered correlated.

Restored versus Reference Wetlands

Mean maximum species richness differed between years (2004 = 3.74 ± 1.91; 2005 = 2.39 ± 1.87; $F_{1,146}$ = 21.01; $P < 0.0001$). There was no statistical difference in maximum species richness totals between the 22 restored wetlands (2.92 ±0.18) and the 8 reference wetlands (2.92 ±0.3; $F_{107,39}$ = 0.22; P = 0.98). Even when tested with the Tukey-Kramer adjustment, no difference was observed between restored and reference wetlands. The mean maximum species richness per survey for restored wetlands was 3.57 (±1.91) in 2004 and 2.45 (±1.87) in 2005; the mean maximum species richness per survey for reference wetlands was 4.19 (±1.38 SE) in 2004 and 2.23 (±1.35 SE) in 2005. However, mean maximum species richness did not differ between restored and reference wetlands for either year (Table 12). The decrease in species richness, particularly in reference wetlands, may have contributed to the lack of statistical difference in species richness between restored and reference wetlands.

Table 12. Results for species richness analysis between years and wetland types.

Year	Restored			Reference			
	low	High	mean	low	high	mean	Pr>F
2004	0.0	8.0	3.57	2.0	6.0	4.19	0.24
2005	0.0	6.0	2.45	0.0	5.0	2.23	0.32

Comparison of habitat variables between restored and reference wetlands indicated several differences. In 2004, restored wetlands had a higher median water depth ($F_{39,15}$ = 4.51; P = 0.01) while reference wetlands supported a greater canopy closure ($F_{39,15}$ = 2.59; P < 0.0001) (Table 13). In 2005, permanently flooding was the most influential variable on species richness. Restored wetlands studied included 4 temporary wetlands and 18 permanently flooded wetlands whereas natural wetlands studied included 3 temporary wetlands and 5 permanently flooded wetlands. Therefore, more wetlands with permanent flooding were found in restored sites (81.8%) (Table 13). Restored wetlands maintained a higher median water depth and supported a greater amount of wetland vegetation (P < 0.05). Reference wetlands continued to have a greater canopy closure, similar to 2004 results (P < 0.0001) (Table 13). The landscape variable of percent forest within 1-km^2 did not demonstrate a difference between restored and reference wetlands (P = 0.08) (Table 13).

Table 13. Results of local habitat variable comparisons between restored and reference wetlands with variable summaries.

Variables (2004)	Restored	Reference	Comparison
	mean ± S.E.	mean ± S.E.	Pr>F
Median water depth (m)	39.1 ± 5.33	21.02 ± 4.14	0.01
Edge canopy closure (%)	23.68 ± 5.31	81.93 ± 5.42	< 0.0001
Variables (2005)			
Median water depth (m)	40.11 ± 4.9	17.89 ± 2.48	< 0.0001
Edge emergent veg. (%)	11.29 ± 1.71	6.1 ± 1.46	0.02
Edge floating veg. (%)	12.27 ± 2.44	3.23 ± 1.31	0.002
Edge canopy closure (%)	32.43 ± 4.83	77.16 ± 5.82	< 0.0001

CHAPTER III. DISCUSSION AND MANAGEMENT IMPLICATIONS OF LOCAL AND LANDSCAPE RESULTS

DISCUSSION

The results of this study indicate that restored wetlands can develop vegetative characteristics within approximately 20 years that are conducive to recolonization by frogs in east central Louisiana. Restored wetlands supported greater densities of emergent and aquatic vegetation around wetland edges, as well as a greater abundance of upland vegetation cover than reference wetlands. Although restored wetlands supported less canopy cover, 12 of 13 species of frogs known to occur in this region were found in restored wetlands. The American toad was the only species not detected; this may have been due to the quick and short breeding behavior making American toads difficult to detect. Restored wetlands supported similar levels of maximum frog species richness as reference wetlands and only one species, the Woodhouse's toad, was unique to reference wetlands. The percent of forest surrounding a wetland, restored or reference, was the only landscape variable measured that positively affected species richness.

The results of this study were similar to findings of several other studies. Mazerolle et al. (2005) found wetland vegetation structure (herbaceous and submerged vegetation) to be an important predictor of the occurrence of green frogs (*Rana clamitans*). They also found that the percent forest within 1 km of a pond influenced pond occupancy for this species. Similarly, Hazell et al. (2004) found that constructed ponds support similar species richness as natural ponds. In addition, Hazell et al. (2004) found the amount of emergent vegetation at the wetland edge was a good predictor of species richness and the occurrence of individual species. However, they did note that chorus size was significantly larger in natural wetlands than constructed ponds (Hazell et al. 2004). Welch and MacMahon (2005) found consistent pond size, emergent vegetation, and the presence of emergent vegetation in late summer to be indicators of the presence of Columbia spotted frogs.

The results of this study are mostly based on chorusing frog surveys as dipnetting surveys did not result in the detection of many tadpoles. Though dipnetting occurred once a season, it may not have been as intensive as necessary to detect the presence of more tadpoles for some species. In addition, some sites were large wetlands and the time-constrained searches may not have covered enough of the edge to detect tadpoles. Other studies have successfully detected tadpoles using dipnetting (Kline 1998, Petranka et al. 2004). Dipnetting may have been unsuccessful in this study due to inadequate frequency, low search time, or percentage of habitat searched. However, the presence of fish was noted at 17 of the 22 restored sites (77.2%) and 6 of the 8 references sites (75%). Several studies have found the presence of fish and other predators to greatly influence the ovipositing and continued presence of frogs during the breeding season (Hazell et al. 2004, Petranka et al. 2004, Petranka and Holbrook 2006). Frogs will often actively avoid ovipositing in ponds where juveniles may encounter intense competition or high predation and will even rapidly recolonize when fish are no longer present (Petranka et al. 1994, Blaustein 1999, Petranka and Holbrook 2006). This may or may not have affected the lack of detection of egg masses and tadpoles; however, more specific research would be necessary to make such inferences. Regardless, due to the poor detection of tadpoles, the results of this study do not represent the breeding success of any sites.

The response variables of frog species richness totals and individual frog occurrences were most greatly influenced by year. In 2004, precipitation was high and caused extremely wet conditions. This climatic patttern flooded many sites and recharged dry, temporary wetlands, most likely allowing several species of frogs to move to and reproduce in otherwise unavailable temporary wetlands in the restored and reference sites. One small temporary reference wetland hosted 7 species of calling frogs in late May of 2004. In 2004, the significant habitat variables were median water depth, and canopy closure. The severe flooding caused some of the emergent vegetation to be completely submerged. Wetlands with gentle slopes and vegetation along the edge, whether herbaceous or emergent, were completely submerged under many centimeters, sometimes over a meter, of water. The shoreline then was no longer dominated by wetland vegetation and often the only wetland vegetation was herbaceous vegetation mostly underwater. Wetlands that still supported floating vegetation and had high canopy closure had greater maximum species richness as demonstrated in the stepwise regression. The trees providing canopy may have also provided vertical structure when emergent vegetation was flooded.

The second study year, 2005, marked a drought year for most of Louisiana. The drought dried up some of the temporary wetlands in March, depriving several species of spring-breeding frogs from using those sites. The drought also caused lower water levels in all sites and more stagnant water within the sites. By early May, all temporary wetlands were dry and few frogs were heard in or around these sites. The same small temporary reference wetland mentioned previously did not host any calling frogs during chorus counts in April, May, or June of 2005. In 2005 the variables significantly influencing maximum species richness were: edge aquatic and floating vegetation, median water depth, and edge canopy. With the decreased size and water depth of many sites, those wetlands that continued to hold water tended to support aquatic and floating vegetation and thus had greater maximum species richness. Also in 2005, canopy closure was likely influential because of the importance of shade during the drought as well as providing "resting" habitat during the day. Overall, the climatic events during this study, particularly rainfall, likely influenced the local habitat variables found influential as well as the response variables of species richness and individual species occurrence.

Flood duration also largely influenced frog species richness in many sites. Flood duration was different between 2004 and 2005, and overall frog species richness was associated with permanent flooding both years. This may have been due to the greater number of sites with permanent flooding, leading to an over-representation of this wetland type. Additionally, the methods did not distinguish between semi-permanent and permanent flooding, which can influence species richness (Fredrickson and Taylor 1982). Temporary flooded sites were not well-represented in this study, and have been a cause for concern in conservation and amphibian monitoring (Rowe and Dunson 1995; Dodd and Cade 1998; Babbitt and Tanner 2000; Brodman et al. 2003). However, because of the extreme climatic events during the two years of this study, the importance of the temporary wetlands may have been highlighted by flooding in 2004 and by the drought in 2005. The re-flooding of the temporary wetlands in spring of 2004 provided potential breeding habitat through the summer breeding season. However, the drought in spring of 2005 caused complete drying of all temporary wetlands, thus stunting their reproductive potential for the remainder of the spring and summer season. Overall, hydroperiod is one of the more consistently important variables in amphibian studies (Rowe and Dunson 1995, Snodgrass et al. 2000, Welch and MacMahon 2005) and variance in flooding was demonstrated in this study as well.

The local habitat variable analysis demonstrated, as many studies have done previous to this study, that frogs are dependent upon a variety of habitat variables outside of water availability. Species richness of frogs increased as emergent and aquatic vegetation, canopy cover, and herbaceous vegetation in the upland increased around the edge of wetlands. Our results were similar to other studies finding wetland vegetation (Munger et al. 1998, Monello and Wright 1999, Stolt et al. 2000, Semlitsch and Bodie 2003; Hazell et al. 2004) and canopy (Skelly et al. 2002, Porej et al. 2004, Rochelle et al. 2004) to be important to amphibian occurrence and in some cases reproduction. The local habitat variables found to be important represent the variety of habitat frogs utilize throughout their life cycle: water and wetland vegetation for reproduction, trees for shade and resting habitat, and some sort of upland habitat in close proximity to the wetland for cover and food (Dundee and Rossman 1989). The study sites, both reference and restored, must provide diverse habitat to sustain a diverse community of frogs.

The limited analysis on landscape variables demonstrated that the most influential variable on frog species richness was the percent of forest surrounding a wetland. Forest provides feeding and resting sites, as well as the hibernacula, for many species of frogs (Dundee and Rossman 1989). Mazerolle et al. (2005) also found the percent of forest within 1 km of a constructed pond to be an indicator of green frog occurrence. Even with the variation in the percent of forest surrounding reference wetlands, species richness was strongly associated with adjacent forest. This is important because as restored wetlands mature, they will potentially improve in providing suitable habitat for a suite of frog species. However, it is important to note that planting trees along the perimeter and in the surrounding landscape of restored wetlands is not always done; mature trees can provide several aspects of habitat for frogs and planting trees should not be overlooked in the restoration process (Semlitsch 2000, Skelly et al. 2002, Porej et al. 2004). Even reforested sites surrounding restored wetlands were positively associated with several frog species and most likely will continue to positively influence frog communities as wetland maturation progresses.

Agriculture surrounding some of the sites had unclear effects on frog species richness in this study. Though agriculture was not a significant influence on frog species richness, it was a negative and positive influence on individual species occurrence. Bronze frogs were negatively influenced by agriculture surrounding a site while gray tree frogs were positively influenced by agriculture around a site. Several studies have observed that agriculture surrounding a restored or created wetland had a negative influence on amphibian occurrence (Monello and Wright 1999, Pechmann et al. 2001, Guerry and Hunter Jr. 2002). Bronze frogs tend to spend their breeding season in and near water sources; however, gray tree frogs only reproduce in water and spend the rest of their life cycle on or near trees (Dundee and Rossman 1989). All four reforested sites in this study were partially surrounded by agriculture and reforested sites had an overall higher occurrence of gray tree frogs. This may have biased the results demonstrating agriculture positively influencing gray tree frog occurrence; the positive association is likely a product of the reforested sites chosen and may not represent the true influence of agriculture on gray tree frogs in restored wetlands.

Individual frogs species occurrence showed on a finer scale how frog species can have very specific habitat needs to survive at a wetland. Bullfrogs and bronze frogs tend to be more general in their habitat requirements and this may contribute to the correlation detected between these species. Also, the logistic regression indicated they occurred at sites with wetland vegetation, herbaceous vegetation in the upland, and the presence of canopy. However, the gray tree frogs occurred at sites with trees throughout the wetland to provide canopy; gray tree frogs were also adverse to more traditional wetlands with emergent vegetation. Gray tree frogs were not detected at sites without trees or saplings. This was most likely indicative of their tendency to occur in temporary wetlands and reforested sites as they spend much of their life cycle in and on trees. However, more mature restored wetlands with larger saplings and trees had gray tree frogs present and this may indicate that young restored wetlands can provide suitable gray tree frog habitat once the planted trees are established. The northern cricket frog was also slightly more specific in the habitat variables influencing its occurrence. The northern cricket frog was found at sites with floating wetland vegetation, but also at sites with a developed upland including herbaceous vegetation, canopy, and litter. Herbaceous vegetation and debris found throughout the site negatively influenced the occurrence of northern cricket frogs. Northern cricket frogs were not detected at any reforested site; however, Northern cricket frogs were associated with wetlands with herbaceous vegetation, litter in the upland, and were permanently flooded.

Analyzing habitat needs for 4 species of frogs demonstrated how diverse the influential habitat variables can be between them and how uniform wetland restoration can exclude many species of frogs. For example, creating large impoundments without any reforestation would likely provide frog habitat for bullfrogs and bronze frogs, but would probably remain mostly uninhabited by gray tree frogs. In turn, if the impoundments do not have depressional areas to continue holding water during seasonal drawdowns, then bullfrogs cannot reproduce as their tadpoles need longer flooding to fully develop. Wetland restoration could provide suitable frog habitat for many species by either incorporating many diverse wetlands into a complex or creating diverse habitat within a single wetland. Other studies have shown that even though constructed ponds supported similar species richness, the occurrence of individual species can greatly vary (Hazell et al. 2005). Hazell et al. (2005) found that two species of frogs in Australia were never detected in constructed ponds, though they were present in neighboring natural ponds. Similarly, a study in Wisconsin noted that 25% of the species found in natural wetlands did not colonize the newly constructed wetlands throughout the duration of the study (Knutson et al. 1999).

The species richness among restored and reference wetlands demonstrated that there was no significant difference, indicating that restored wetlands have the potential to provide suitable habitat for frogs. Restored wetlands seemed to provide more of the wetland vegetation (i.e. emergent and floating vegetation) necessary to frogs. Large impoundments in this study often had large expanses of cattail (*Typha latifolia*), American lotus (*Nelumbo lutea*), and water hyacinth (*Eichhorinia crassipes*), an exotic species. This wetland vegetation provides cover, food, and calling sites for frogs, but can be a management issue if it overwhelms a wetland. However, with active management wetland like disking, wetland vegetation can be controlled while still providing suitable habitat for frogs. Restored wetlands also provided more upland herbaceous vegetation, particularly in the reforested wetlands where young trees allowed enough sun to penetrate the shrub layer and the herbaceous vegetation to establish. Also, many of the impoundments and enhancements had little canopy at the edge at the time of this study, allowing for herbaceous vegetation to take over the wetland edge.

Reference wetlands had a greater percentage of canopy closure at the wetland edge. This may be indicative of wetland maturation and age as all 8 reference sites had mature trees along the wetland edge. The presence of trees at the wetland edge may have reduced light to the shrub layer, likely reducing upland herbaceous vegetation. However, forest surrounding the sites was similar in restored wetlands; this could represent thoughtful site selection for created wetlands in providing adjacent terrestrial habitat until the restored wetland has matured. Both reference and restored wetlands had a similar percentage of open water. And, as mentioned earlier in the discussion, agriculture surrounding wetlands is not necessarily a negative influence; however, it should be monitored and the wetland should be connected to some form of suitable terrestrial habitat if possible.

This study was only able to account for two years of change and I would recommend restored and created wetlands be evaluated over a longer period of time. This would allow for maturation of wetlands and time for more sites to be recolonized; thus, it would better represent the value of wetland habitat provided by restored and created wetlands. Pechmann et al. (2001) and Petranka et al. (2003) recommend more than 5 years of monitoring for newly constructed or restored wetlands to evaluate the full effects on amphibian communities; an even greater amount of time may be necessary for forested wetlands.

I would also recommend more intensive frog sampling such as seining, traps, and regular visual encounter surveys to allow for more accurate and robust species-specific results concerning detection probabilities and habitat relationships (MacKenzie et al. 2002, Royle and Nichols 2003, Bailey et al. 2004, Mazerolle et al. 2005). In addition, several recent studies have questioned the spacing and clustering of restored wetland sites and complexes (Blaustein 1999, Petranka et al. 2004, Petranka and Holbrook 2006). Blaustein (1999) raised concerns about the avoidance of frogs ovipositing in ponds with high presence of predators; in addition, Petranka et al. (2004) and Petranka and Holbrook (2006) have further noted that amphibians demonstrating ovipositing avoidance may treat clusters of wetlands as mere patches of the same habitat type and move out of the area altogether. The tight spacing of restored wetlands can negatively affect the metapopulation structure of frogs and other amphibians (Petranka et al. 2004, Smith and Green 2005, Petranka and Holbrook 2006). Additionally, too few and too isolated wetlands can produce unsatisfactory results as well (Moler and Franz 1987, Bosch et al. 2004, Cushman 2006). More research on placement of restoration sites and distances between sites could provide more suitable habitat in restored wetlands for frog communities.

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APPENDIX I. CORRELATION RESULTS FOR LOCAL EDGE HABITAT VARIABLES

	Mean water depth (R*)	Non-edge floating (R)	Non-edge open water (R)	Non-edge dead (R)	Non-edge litter depth (R)	Non-edge bare (R)	Non-edge emergent (R)	Non-edge aquatic (R)	Non-edge canopy (R)	Non-edge herbaceous veg. (R)
Median water depth	0.95	-	-	-	-	-	-	-	-	-
Edge floating	-	0.76	-	-	-	-	-	-	-	-
Edge open water	-	-	0.77	-	-	-	-	-	-	-
Edge dead	-	-	-	0.67	-	-	-	-	-	-
Edge litter depth	-	-	-	-	0.79	-	-	-	-	-
Edge bare	-	-	-	-	-	0.66	-	-	-	-
Edge emergent	-	-	-	-	-	-	0.50	-	-	-
Edge aquatic	-	-	-	-	-	-	-	0.51	-	-
Edge canopy	-	-	-	-	-	-	-	-	0.67	-
Edge herbaceous veg.	-	-	-	-	-	-	-	-	-	0.83

* Denotes removal of variable from further analysis.

APPENDIX II. SUMMARY RESULTS FOR ALL LOCAL VARIABLES IN 2004

	Restored			Reference		
Variable	Low	High	Mean	Low	High	Mean
Max. species richness	0	6	4.19	0	8	3.57
Edge emergent (%)	0	21	4.28	0	55.9	8.69
Edge floating (%)	0	12.5	1.38	0	59	11.19
Edge aquatic (%)	0	19	4.74	0	48	2.48
Open water (%)	0	87.4	31.91	0	99.8	49.5
Median water depth (cm)	0	63.4	21.02	0	150	39.1
Edge canopy (%)	40	100	81.93	0	100	23.68
Edge bare (%)	0	68.8	16.12	0	47.6	5.43
Edge debris (%)	0.30	74.9	23.41	0	48.4	11.23
Edge herbaceous veg. (%)	1.1	69.7	18.83	0	68	11.89
Upland edge litter (cm)	0.10	2.7	1.15	0.0	3.6	0.52
Upland edge debris (%)	15	78.5	43.86	0.7	55.8	19.18
Upland edge bare (%)	0.2	31.6	9.52	0	91.3	19.41
Upland edge canopy (%)	40	100	72.92	0	100	30.84
Upland edge herbaceous veg. (%)	12.3	66.2	35.71	0.5	75.4	37.68

APPENDIX III. SUMMARY RESULTS FOR ALL LOCAL VARIABLES IN 2005

	Restored			Reference		
Variable	Low	High	Mean	Low	High	Mean
Max. species richness	0	6	2.23	0	5	2.45
Edge emergent (%)	0	22.3	6.1	0	59.3	11.29
Edge floating (%)	0	30	3.23	0	73.3	12.27
Edge aquatic (%)	0	24.5	2.74	0	63.7	3.1
Open water (%)	0	83	42.61	0	100	42.69
Median water depth (cm)	0	43.4	17.89	0	150	40.11
Edge debris (%)	1.3	93.8	26.23	0	84.5	15.32
Edge bare (%)	0	57.3	12.8	0	54	9.33
Edge canopy (%)	40	100	77.2	0	100	32.42
Edge herbaceous veg. (%)	0	54.8	4.8	0	68.3	7.49
Upland edge litter (cm)	0.0	2.0	0.48	0.0	1.5	0.24
Upland edge debris (%)	12.7	90	54.43	0	87.3	31.29
Upland edge bare (%)	0	55	14.82	0	52.5	13.4
Upland edge canopy (%)	41.7	100	85.93	0	100	37.42
Upland edge herbaceous veg. (%)	0.3	78.3	23.13	0	95.7	40.44

APPENDIX IV. SUMMARY RESULTS FOR ALL LANDSCAPE VARIABLES

	Restored			Reference		
Variable	Low	High	Mean	Low	High	Mean
Forest (%)	0.0	70.0	21.18	10.0	80.0	40.0
Agriculture (%)	0.0	40.0	17.22	0.0	80.0	33.12
Reforested (%)	0.0	80.0	50.29	0.0	80.0	26.88
Distance to closest forest (km)	0.1	1.5	0.53	0.1	1.0	0.39

VITA

Sarah Barlow was born in Rhinelander, Wisconsin, where she graduated from Rhinelander High School. She began college at the University of Wisconsin-Milwaukee and transferred to the University of Wisconsin-Stevens Point where she earned a bachelor of science degree in wildlife in 2001. During her undergraduate experience, she worked with the U.S. Forest Service monitoring the Kirtland's warbler in the lower peninsula of Michigan. This position also led to additional small mammal research and valuable manuscript editing experience. Before beginning graduate school, she conducted drift fence surveys for reptiles and amphibians with the University of Florida, taught wildlife education for the Georgia Department of Natural Resources, and conducted Golden-winged warbler surveys for the Wisconsin Department of Natural Resources.

Sarah began graduate school at Louisiana State University in the fall of 2003. The research assistantship involved conducting avian point count surveys, chorusing anuran counts, drift fence surveys of reptiles and amphibians, and belt transect vegetation surveys across 4 management areas in central and eastern Louisiana. Sarah primarily worked near Marksville, Louisiana, on Pomme de Terre Wildlife Management Area and Spring Bayou Wildlife Management Area. Her thesis work was conducted on private lands in this area to coincide with the research assistantship responsibilities. Sarah resides in Grand Rapids, Minnesota, and is a substitute science teacher.