THE EFFECTS OF MINE LAND RECLAMATION
ON HERPETOFAUNAL COMMUNITIES

by

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ABSTRACT

THE EFFECTS OF MINE LAND RECLAMATION ON HERPETOFAUNAL COMMUNITIES

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Succession describes the process of community change over time after a disturbance. Understanding these processes allows ecological restoration projects to take advantage of natural community trajectories. Reclamation efforts often “jump-start” succession to restore anthropogenically disturbed land and water features to their former state. This is often accomplished by seeding for desired species or planting tree seedlings. One way of gauging whether reclamation was successful is to compare floral and faunal communities of reclaimed land with that of nearby land that was not disturbed. Most such studies have focused on plant and bird species while few have looked at the effects of reclamation on herpetofauna. Reptiles and amphibians are ectothermic, have relatively small home ranges, and are limited in their dispersal abilities. These characteristics might make them more strongly affected by environmental disturbances compared to most birds and mammals.

I studied the effects of past strip mining activity and time since reclamation on frog and turtle communities in north-central Texas. I compared the communities inhabiting ponds that had been reclaimed following strip mining to nearby reference ponds that have never been mined.

From July 2010-October 2011, I monitored turtle and frog populations, as well as environmental variables such as shoreline vegetation, aquatic vegetation, and turbidity at each pond.

While there were no significant differences in turtle and frog species richness, frog species composition was different between mined and unmined ponds. Unmined ponds had smaller turtles than mined ponds, and more species of turtles. I found few differences in the environment of these pond groups when the variables were analyzed independently, though unmined ponds had more trees along the shoreline. Mined and unmined pond environments differed when I used a principle components analysis to analyze these variables together.

There were no significant differences in turtle and frog species richness between 20-year and 30-year-old ponds, but species composition was different. Larger turtles were found at ponds that were 30 years post-reclamation. I found no environmental differences between the two pond age groups.

Several environmental variables independent of mining or time since reclamation were correlated with characteristics of turtle and frog communities. Aquatic vegetation cover did not differ consistently among pond groups, but it was positively correlated with overall frog species richness as well as hylid richness. In addition, the distance from a pond to a larger water source was unrelated to time since reclamation and mining history, but was correlated with both turtle density and bufonid richness.

Past studies in this area found riverine turtle species that were not found in my study. These species were probably unable to establish populations in a pond habitat. These studies also found several species of frogs, known to be explosive breeders, which were not found in my study. I most likely did not detect these species due to their fossorial nature and lack of rain during my field seasons.

This study suggests that ponds reclaimed after lignite coal mining are capable of supporting similar species richness' for turtles and frogs, but have different species compositions compared to unmined ponds. The similarity of turtle and frog species richness and composition

between 20 and 30 year old ponds shows that these groups are capable of repatriating an area in less than 20 years, though establishment of mature turtle communities may take longer. Finally, turtles and frogs are probably more influenced by habitat characteristics and land management, such as stocking ponds with fish or protecting stands of trees, than factors directly associated with mine reclamation or time since reclamation.

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CHAPTER 1

INTRODUCTION

Ecologists have a long history of exploring and documenting succession, the process by which communities change over time (Cowles 1899, Cooper 1923, Gleason 1926, Clements 1936). The goal of succession research is to reveal patterns of colonization in order to predict how ecosystems change after a particular disturbance. Understanding how succession affects disturbed ecosystems is important in fields such as restoration ecology, where scientists can take advantage of natural trajectories to repair anthropogenically disturbed land. This understanding allows the land to be repaired in ways that promote and sustain natural communities (Prach and Walker 2011).

1.1 Succession

Ecologists have been able to witness dramatic examples of succession and track them over time. For example, Alaska's Glacier Bay has been studied since 1794. Since then, its namesake glaciers have retreated 105 km into the mountains, leaving scoured rock in their wake (Chapin et al. 1994). The type of succession seen in Glacier Bay, where new geological substrates are exposed and are not significantly modified by organisms, is known as primary succession (Clements 1916). As with succession in other environments, colonization of the Glacier Bay habitat began with a pioneer community such as lichens and liverworts, capable of invading the harsh, disturbed environment. These pioneer plants break down rock to create soil and through their own decomposition add organic matter and nutrients to that soil (Worley 1973, Lawrence 1979). Pioneer plants also facilitate the colonization of some plants which are able to fix nitrogen, a limiting resource in most plant communities, using symbiotic bacteria. This addition allows other plants such as shrubs and trees to establish which in turn increases species richness. As more species are able to establish, the availability of resources such as sunlight, space, and soil nutrients diminishes, which initiates competition among species for these limiting resources. As competition increases, some species are unable to persist and their numbers either decrease or they disappear from the habitat entirely. This process results in a decrease in species richness as the community reaches a mature and stable equilibrium (Odum 1969). This equilibrium, however, is almost always dynamic. Tree falls, forest fires, and the abandonment of agricultural fields are smaller-scale disturbances that may initiate secondary succession. These events do not expose a new geologic substrate, therefore they may have very different stages of succession compared to primary succession.

Though primary succession in plants is well documented in many habitats, primary succession of vertebrates is less well studied. While the initial stage of plant succession is determined by substrate characteristics and nutrient availability, succession patterns of animals are typically based on plant community abundance and composition (Sousa 1984). Animal species will colonize an area once the established vegetation meets their habitat requirements. As the vegetational complexity of the habitat increases, there are more potential niches for animals to fill, and animal diversity increases (August 1983). As the succession of the plant community moves forward and the community changes, it may no longer be suitable for some animals. As with plant succession, other suitable animal species will move into the area and compete with the pioneers, resulting in the pioneer community often being either reduced or excluded from the environment (Sousa 1984, Fox et al. 2003).

For succession to occur, species must be able to disperse to the newly available habitat from a source population (Simberloff and Wilson 1969, Gaines and McClenaghan 1980). Methods of species dispersal vary widely in plant and animal communities, and confer different advantages in different situations. For example, wind-assisted seed dispersal in plants is a passive process in which plant propagules are carried potentially long distances by wind currents. These seeds typically have wings or plumes that help them catch the wind. Plants that propagate in this way can colonize areas many miles away from the parent plant (Willson et al. 1990).

Others plants produce seed bearing fruit with pulpy flesh to encourage consumption by animals that will then disperse the seeds in their feces. Another way animals facilitate plant dispersal is by carrying plant propagules on their bodies. These seeds have specialized features like hooks that attach to animals as they brush past the plant. Plants that attract flying animal species such as insects, birds, or fruit bats can have similar dispersal as wind-dispersed plant species since these animals are capable of carrying seeds far away from the parent population (Willson et al. 1990).

Animals also have a wide range of dispersal abilities. In many cases, animals must use terrestrial locomotion to disperse, as is the case for large ungulates like caribou. Flying species like birds and bats can cross most habitats, species that swim such as otters and some lizards can traverse aquatic barriers, and some smaller terrestrial species can hitch a ride with more mobile animals or mats of floating vegetation (Greenwood et al. 1979, Gaines and McClenaghan 1980). However, the ability to disperse is not the only factor involved in the ability of an animal to establish in new locations. The distance of a source population from the new habitat also determines the number of species able to reach the new habitat, as well as the abundance of those species in the new location (Simberloff and Wilson 1969).

The Krakatau Islands in Indonesia are a good system in which to study vertebrate succession. In 1883, a volcanic eruption sterilized this chain of islands; their floral and faunal communities have been monitored since 1908 (Rawlinson et al. 1992). For non-flying mammals, stretches of saltwater greater than 5 km in width are major barriers unless humans facilitate their travel (Heaney 1984). This limits the number of species able to survive swimming or rafting through saltwater to a new habitat.

The first vertebrate to colonize the islands after the eruption was recorded in 1889; *Varanus salvator*, the water monitor, is a large species of carnivorous lizard capable of swimming. Its ability to disperse to the islands through water and the availability of prey items such as sea turtle eggs, crabs, and carrion account for this species' early colonization. When *V. salvator* arrived, the vegetation of the island was characterized by grasslands in the lowlands with discontinuous young woodlands along the coast and ferns retreating to higher elevations (Whittaker et al. 1989). Another strong swimmer, *Python reticulatus*, the reticulated python, was found early in succession in 1908 (Jacobson 1909). This large python would have been able to utilize *V. salvator* as prey, as well as rats that may have been present on the islands from human expeditions. Several species of insectivorous lizard species subsequently colonized the island, most likely arriving by rafting on vegetation mats or as a result of human facilitation. Many of these early lizard species have not been found in recent years, as they were unable to establish breeding populations. *Rattus rattus*, the black rat, was first recorded on the islands in 1918 (Yukawa et al. 1984). At that time, the grasslands had begun changing into forests and the fern communities began to decline (Whittaker et al. 1989). Since then, several other species of rats have established, most of which are assumed to have arrived with humans. *Sus scrofa*, the wild boar, is an excellent swimmer native to Indonesia. This species has been documented on the islands since 1982, but so far there is no evidence that a breeding population has established (Rawlinson et al. 1992).

Flying animals are particularly adept at colonizing disturbed habitat. Between 1908 and 1919, fruit bats were able to colonize the islands as forests were developing due to their ability to disperse over water and their lack of specific roosting requirements (Tidemann et al. 1990). Because fruit bats ingest seeds when they feed on fruits and expel them in their feces, they brought more plant species to the islands that were able to establish and create even more suitable bat habitat (Whittaker et al. 1989). By far, non-migrant land birds make up the greatest number of species found on the islands after the eruption. Their diverse habitat requirements and ability to disperse over water make them excellent colonizers of island habitats. Bird species turnover was high as the succession of vegetation changed the habitat characteristics. For example, *Amaurornis phoenicurus*, the white-breasted waterhen, and *Centropus bengalensis*, the lesser coucal, were both recorded in 1928 and are no longer present on the islands. Their preferred habitat, marshes and grasslands respectively, has been extirpated by other types of vegetation as succession continues (Rawlinson et al. 1992). The study of Krakatau provides a rare example of herpetofauna being monitored during primary succession.

1.2 Mining and Reclamation

Strip mining for coal is an anthropogenic disturbance that initiates primary succession (Wagner et al. 1978, Kirmer and Mahn 2001). During this process, the surface material, known as overburden, is stripped away in order to reach the seam of coal underneath. This type of mining is only appropriate when the coal deposit is close to the surface and can be much more efficient than underground mining, recovering 90% or more of the coal from the seam as opposed to 60-75% with other techniques (World Coal Institute 2009). The mining process inherently results in the localized loss of habitat and ecosystem services provided by that habitat. Once the coal has been removed from the ground, the excavation site is either restored to its previous state, or a new habitat is created in a process called reclamation. Reclamation is an artificial process in which reclamation scientists “jump-start” the succession of plants over mined areas rather than allowing plants to colonize and establish over a longer period of time. They do this by planting native vegetation and managing the plant community and soil properties over several years, as is required by the Surface Mining Control and Reclamation Act (PL 95-87 1977). Areas can be reclaimed into several land use categories such as forestry, pastureland, cropland, fish and wildlife habitat, quail and upland bird habitat, developed water resources, or industrial commercial (Luminant 2010). Mining companies are forced to give the state they are located in bond money to safeguard the reclamation of the mine sites, portions of which are returned to the company at specific stages of reclamation. This ensures that the mining company reclaims the land in a timely manner; otherwise, the state keeps the money and uses it to reclaim the land (PL 95-87 1977).

Research on the succession of plant and animal communities can serve as a guideline for land restoration after processes such as strip mining. Information from these studies can help reclamation scientists by revealing species interactions, community transitions and trajectories, and how communities develop (Walker and Moral 2009). These studies can also help them choose an appropriate stage of community to begin restoration efforts, the timing in which to initiate actions, as well as help recreate specific ecosystem services provided by successional communities (Prach and Walker 2011). Failure to understand how communities change over time can result in inappropriate actions by reclamation scientists, deleterious impacts on the local ecosystem, as well as loss of money when actions must be taken to correct mistakes. The success of reclamation is typically measured by plant productivity: stem counts for areas with woody vegetation and percent ground cover for areas dominated by grasses. However, to fully determine if reclamation was successful, animal communities in these areas should also be evaluated.

The diversity and population characteristics of animal communities at reclaimed mine sites can give an indication of how well those sites are able to support the communities expected in undisturbed areas of similar habitat. Most studies of vertebrate succession after strip mining have focused on birds (Wray et al. 1982, Bajema and Lima 2001, DeVault et al. 2002, Scott et al. 2002, Scott and Lima 2004, Monroe and Ritchison 2005), and to a lesser extent mammals (Yeager 1942, De Capita and Bookhout 1975, Sly 1976, Ireland et al. 1994). These studies have had varying results, with some studies finding similar species richness between disturbed and undisturbed sites (Scott et al. 2002), and others greater species richness at undisturbed sites (De Capita and Bookhout 1975, Sly 1976, Ireland et al. 1994). Other studies find decreased reproductive success for birds on reclaimed land due to an increase in predation (Wray et al. 1982).

1.3 Herpetofauna

The succession of herpetofauna in reclaimed land has rarely been studied, perhaps partly because the importance of herpetofauna in ecosystems is often overlooked. These animals can be important for studies of vertebrate succession due to their particular life history characteristics. Unlike birds and mammals, reptiles and amphibians are ectotherms, requiring enough complexity in their habitat to move from areas of higher or lower temperatures as needed. This may make them more strongly affected by the habitat of reclaimed areas than birds and mammals (Huey 1991). In addition, turtles and frogs are both predators and prey, making them key organisms in both aquatic and terrestrial food webs.

Reptiles and amphibians have relatively small home ranges and limited dispersal ability when compared to other vertebrates, increasing the chance they will be extirpated after a disturbance (Sinsch 1990, Blaustein et al. 1994, Galatowitsch and vanderValk 1996, Brown et al. 1997, Driscoll 1997, Semlitsch 1998). Frogs and turtles in particular have specific dangers associated with migration. Frogs have a thin, permeable skin which makes them vulnerable to desiccation when away from a water source or appropriate refugia (Vitt and Caldwell 2009). In addition, frogs may not be able to colonize new habitat as readily as other organisms due to breeding site fidelity (Sinsch 1997), or barriers such as agricultural fields and roads (Fahrig et al. 1995, Vos and Chardon 1998, Lehtinen et al. 1999). Roads are also a dangerous barrier for turtles, particularly females migrating to lay eggs (Marchand and Litvaitis 2004).

Turtles and frogs are both indicators of habitat quality. Since turtles are typically long lived, their intermediate trophic level may allow toxins and pollutants that might be present in the environment to accumulate in their bodies (Holcomb and Parker 1979, Stone et al. 1980, Bishop et al. 1991). Aquatic turtles, in particular *Chelydra serpentina*, the common snapper, and *Trachemys scripta*, the red-eared slider, are known to travel overland up to 9 km from their starting location (Ernst and Lovich 2009, Vitt and Caldwell 2009), allowing them to leave ponds with unsuitable habitat characteristics (Roe et al. 2009).

The complex life cycle of frogs exposes them to many different microhabitats in a small geographic area. Frog life cycles typically consist of egg, larval, juvenile, and adult stages. All of these stages are accompanied by distinct morphological, physiological, and behavioral changes (Vitt and Caldwell 2009). The biphasic life history of frogs combined with permeable skin makes them particularly sensitive to water and habitat conditions, including the presence of pollutants (Welsh and Ollivier 1998, Marco et al. 1999, Barbeau and Jr 2007, Mann et al. 2009, Vitt and Caldwell 2009). Even primarily terrestrial species such as toads have offspring that remain aquatic until metamorphosis, resulting in a strong dependence on the quality of aquatic habitat. Due to this complex life cycle, frogs are also heavily influenced by microclimate (Rios-Lopez and Aide 2007). They seek varying levels of moisture and refugia to facilitate rehydration, decrease the chance of egg desiccation, and ensure the normal development of eggs (Vitt and Caldwell 2009). The presence of turtle and frog populations could be an indication of the long term success of reclaimed ponds.

Though turtles and frogs are rarely studied in primary succession, studies of herpetofaunal secondary succession are more common. Most often in these studies, succession is initiated by human disturbance such as clear cutting to make way for agriculture, or the draining of wetlands. In a project in Puerto Rico, herpetofaunal succession was accelerated by reforestation efforts (Rios-Lopez and Aide 2007). As in other studies, species richness increased with increasing vegetation heterogeneity (Heinen 1992, Rios-Lopez and Aide 2007). Microclimate and microhabitat requirements play a large role in the composition of species found during herpetofaunal succession, as does prey availability (Heinen 1992; Rios-Lopez and Aide 2007). For example, arboreal species are unable to establish if there are no trees, and carnivorous species are unable to establish if their prey items are not available.

1.4 Big Brown Mine

The Big Brown Mine near Fairfield, TX has strip mined and reclaimed over 6,000 hectares of land and has received awards for its reclamation efforts (www.luminant.com). The land reclaimed from Big Brown's mining operation is dotted with cattle ponds established from 1971 to the present. This availability of aquatic habitat and range of pond ages makes this land a good location for studying turtle and frog succession after reclamation. County records suggest a variety of herpetofauna occur in the area (Table 1). In addition, two prior studies were done on this land sampling the herpetofaunal community.

In 1985-1986, the herpetofaunal community of mined and unmined forests and fields was surveyed at the Big Brown Mine (Bradley 1987). The mined lands in the study had been reclaimed in 1981, four years before the study began. Pitfall traps were coupled with funnel traps to assess differences in species richness and density of reptiles and amphibians between disturbed and undisturbed lands. As an addition to the main project, aquatic turtle communities were sampled using hoop traps in sediment ponds to determine the effects of fish stocking on turtle community structure (Bradley 1987). This type of pond is not mined and reclaimed; they are part of an active mine site. Sediment ponds are man-made ponds created to hold runoff and sediment until the suspended solids contained within the water settle to the bottom of the pond. Strict laws manage the amount of sediment that can be contained in water discharged from mine sites, making these ponds a necessity for mining operations. This study found several species of turtles and frogs, providing information on herpetofauna historically found in this specific area (Table 1).

Another study was conducted in 1992 at the request of Luminant, known as Texas Utilities Mining Company at the time, by Espey, Huston & Associates, Inc. to evaluate the fish and wildlife resources at the Big Brown Mine (Jasper 1992). The purpose of this study was to identify the presence of any “species of concern” that might need to be protected as mining operations continued. From October 14-18 and April 6-9 1992, time-constrained techniques were used that allow herpetofauna to be quantified per unit of time spent sampling. Driving transects were also conducted to sample for amphibian breeding calls as well as herpetofauna crossing the roads. Finally, visual surveys as well as hoop traps were used in aquatic areas to sample turtles and other species. Similar to Bradley (1987), this study provides additional data about species that were present in the past (Table 1).

Both prior studies at Big Brown found the same lentic turtle species, but Bradley also found *Pseudemys concinna* and *Apalone spinifera,* species known to inhabit lotic habitats. The discrepancy between these two studies is probably a result of more extensive trapping in the Bradley study, or Bradley may have caught migrants that were unable to establish populations in the sediment ponds. Several more frog species were found by Bradley than Jasper, in particular two explosive breeders, *Gastrophryne olivacea* and *Gastrophryne carolinensis*. The absence of these two species was probably due to a lack of a large rain event during sampling in the Jasper study. Jasper found one frog species that was not found in the Bradley study, *Hyla versicolor,* the gray tree frog, a species that requires arboreal habitat upland from breeding ponds. The presence of this species in his surveys is probably due to the survey method. This study utilized road transects to listen for breeding calls, potentially placing the samplers closer to areas with remnant forests than if they were directly on the mining property.

The combination of potentially present herpetofaunal species and the previous knowledge of this area make the Big Brown Mine site an excellent location to study herpetofaunal succession. My study evaluated the ability of artificially created ponds and wetlands to support vertebrate ectotherms, as well as determine if reclamation and time since reclamation have an impact on turtle and frog species. Environmental variables were also evaluated that might affect the ability of a pond to support populations of particular species (Gibbs 1998, Knutson et al. 2004, Rios-Lopez and Aide 2007, Shulse et al. 2010, Fuller et al. 2011). I sought to answer the following questions:

Question 1: Does mining and subsequent reclamation have an effect on turtle populations and frog species richness?

Hypothesis 1: Mining has an influence on both turtles and frogs; there are fewer turtle and frog species at mined ponds, as well as denser turtle populations with larger turtles at unmined ponds. Mining changes the qualities of the soil, potentially altering soil density, turbidity, and vegetation in ways that will affect both turtles and frogs.

Question 2: Does time since reclamation affect turtle populations and frog species richness? Can a pattern of succession be seen?

Hypothesis 2: More species of turtles and frogs as well as a higher density of larger turtles are found in older ponds. Older ponds have had more time for colonizers to invade and establish, and for long-lived organisms like turtles to grow.

Question 3: Are there aspects of the environment that are more influential to turtles and frogs than those directly associated with pond age or mining activity?

Hypothesis 3: Both turtles and frogs will be affected by extreme differences in the environment, such as a lack of shoreline vegetation at one pond compared to a fully vegetated shoreline at another pond; shoreline vegetation and other environmental variables may not be closely correlated with mining activity and reclamation. Ectotherms are dependent on their environment for temperature regulation; pond water that exceeds their acceptable temperature ranges would not be suitable for colonization or egg deposition. Vegetation around the pond edge and in the water allows frogs to seek cover from predation and evaporative water loss, as well as provides food for tadpoles. Aquatic vegetation also provides food for adult turtles as well as refuges for hatchling turtles.

CHAPTER 2

METHODS

2.1 Study Area

My study area was located in the Post Oak Savannah ecoregion of Texas. This region is characterized by savannahs comprised of bunch grasses and forbs with scattered stands of trees. More extensive tree populations are typically restricted to areas along rivers and streams. The trees comprising the overstory are mostly oak, as well as hickory, elm and gum. Shade tolerant understory plants include flowering dogwood and hawthorn. The most common native bunch grasses in this ecoregion are bluestem and broomsedge bluestem (www.tpwd.state.tx.us).

Texas is the 5th largest producer of coal in the United States and uses the most coal in the nation to produce electricity (EIA 2002) with coal mining operations in Texas disturbing ~4,000 acres per year (www.tmra.com). The seam of lignite coal runs from the northeast corner of the state down to the southwest corner (Figure 2). Luminant, a subsidiary of Energy Future Holdings, mines the greatest amount of lignite coal and generates the most electricity in Texas. Their Big Brown mine, near Fairfield, TX, began mining in 1971 and has continued operations to the present.

Study ponds were located on two properties that were formerly part of the Big Brown coal mine in northeast Freestone County, Fairfield, TX and are now privately owned by Roy Casey and Jerry Robinson (Figure 3). All of the 30 year ponds plus C-90 were located on the Robinson property, and the remaining three 20 year ponds were located on the Casey property. These lands were reclaimed by Luminant Mining Company into pasture. The study ponds were dug as water features to support livestock and continued to support cattle year-round throughout the study period. Prior to being mined, this land had been exhausted by agricultural use. Because of this history, mining and subsequent reclamation improved the soil quality and texture, creating greater nutrient availability for plant establishment compared to the pre-mined soil conditions (Angel 1973).

Reference ponds, ponds that have never been mined, were located at Gus Engeling Wildlife Management Area (GEWMA), a 4435 ha property purchased in 1950 by Texas Parks and Wildlife in northwest Anderson County, TX (Figure 3). These ponds are ~25km from the mined ponds and are located in the same ecoregion, Post Oak Savannah. The reference ponds were dug within the past 50 years to support cattle; cattle are grazed on the property from April-August.

Both study ponds and reference ponds were actively managed by landowners during the course of this study. Management practices included stocking ponds with fish, controlling cattle access, mowing surrounding vegetation, and bulldozing the perimeter of the pond. The details and frequency of these activities were not disclosed by the land owners. Bulldozing occurred at one pond, Fence East, toward the end of the study, affecting vegetation surveys (Figure 4). Orthorectified aerial images from 2010 were used to gather shoreline vegetation data for this pond since I was not able to complete the vegetation survey in the field.

The second year of field work, 2011, was the most severe year of drought and high temperatures recorded in the state of Texas. By late October, 70% of the state was in a condition of exceptional drought (Figure 5). These conditions rapidly altered the size and environment of the study ponds. Three ponds (Drain Pipe, Off Road, and Skinny) had completely dried up by the end of the study (Figures 6 and 7). Aquatic vegetation measurements could not be completed at Off Road pond due to the drought.

2.2 Ponds

Ponds that were connected to a creek system were discarded from the study. This eliminated the confounding variable of wildlife traveling down the creek to populate some ponds but not others. The remaining ponds were chosen according to their age and size. The smallest ponds available were chosen to facilitate ease of sampling. Four ponds each of two different age categories were selected from the reclaimed land. These ponds were either 20-25, or 30+ years since reclamation. Records containing the exact date of pond creation were not available; these ages were determined from Luminant employees and dated maps of the mined areas. Four reference ponds of varying ages were located 25 km east of the mined and reclaimed ponds at GEWMA. The twelve study ponds range in surface area from 0.12 ha to 1.5 ha (Table 2). Sampling occurred August 2010-October 2010, and again in March 2011-October 2011.

2.3 Study Species/Capture Methods

Hoop traps were used to assess turtle communities, as they have been shown to catch a wide variety of aquatic turtle species (Rizkalla and Swihart 2006, Glorioso et al. 2010). Traps (2.54 cm treated mesh, 0.61 m hoop diameter, 96.52 cm long, double throated) were baited with beef liver and sardines and placed in the water such that several inches of the trap remained above the water line, allowing captured turtles to breathe. Trap openings were oriented both parallel and perpendicular to the shoreline with equal frequency. Each pond received equal trapping effort regardless of size. When multiple days were trapped in a row at the same location, captured turtles were removed, processed, and released; traps were re-baited, and left in the same location. Over the course of two years, ponds were trapped 12 days each with the exception of the 30 year ponds which were trapped 13 days. A trap day consisted of 3 hoop traps per pond, left out for 24 hours.

I sexed turtles by secondary sexual characteristics; if these characteristics were intermediate the turtle was classified as a juvenile (Marchand and Litvaitis 2004, Glorioso et al. 2010). Turtles were weighed with a hanging fisherman's scale and marked by a dremel tool with a unique number on the plastron. Straight-line carapace length and plastron length were measured at the midline with tree calipers to the nearest millimeter (Marchand and Litvaitis 2004, Glorioso et al. 2010). Carapace length was used as the general measure of size and age, as mass and plastron annuli are less reliable (Gibbons and Lovich 1990). Turtles were released where they were caught immediately after being processed.

Amphibian surveys were conducted at night between 8:00pm-11:00pm, March-June 2011. Frog calls were recorded for 10 minutes at each pond and analyzed for presence/absence of species (Crouch and Paton 2002, Gooch et al. 2006); reclaimed ponds were sampled 7 times; reference ponds were each sampled 8 times. Frog calls are species specific (Vitt and Caldwell 2009), allowing a positive identification to be made without seeing the individual, however some frog species are not reliable callers (Crouch and Paton 2002, Vitt and Caldwell 2009). To account for these species, visual surveys were conducted by walking the pond edge with spotlights while recording for calls. Frogs that were seen or heard during day-time turtle trapping were also recorded. A grey tree frog was found through visual surveys; it is unknown whether this was *Hyla chrysoscelis* or *Hyla versicolor* since these two species are indistinguishable by sight. This specimen is referred to as *Hyla chrysoscelis/versicolor* throughout the text. Minnow traps were placed in the ponds to capture tadpoles with the same frequency and distribution as hoop traps.

2.4 Environmental Variables

Several environmental variables were measured for each pond in order to classify ponds by environment as well as relate turtle and frog populations to specific environmental characteristics (Table 3). Water temperature was measured at the depth of the hoop traps at each pond after traps were set and checked. This variable was excluded from analyses due to the temporal variation in measurements among ponds. Turbidity was measured using a secchi disk. All turbidity measurements are reported as “secchi depth,” therefore a higher number equates to less turbid water. Soil series information was obtained from the National Resources Conservation Service Web Soil Survey (www.websoilsurvey.nrcs.usda.gov). Basking site availability was divided into three categories. Ponds labeled “low” had no basking sites other than the shore; ponds labeled “med” had 1-5 basking sites; ponds labeled “high” had >5 basking sites. Examples of these sites include partially submerged objects like limbs, roots, large tires, or rocks that would typically be utilized by turtles and frogs for basking and perching.

Vegetation at each pond was characterized in two ways: shoreline composition and percent aquatic vegetation cover. The perimeter of each pond was mapped using satellite photographs and ArcGIS version 10. The circumference of each pond was walked within 2m of the waterline, and a measuring tape used to estimate the proportion of each vegetation category present. The vegetative composition of the shoreline was broken down into five categories (clear, trees, reeds, herbaceous shrubs, and woody shrubs). The total in meters for each vegetation type was determined and divided by the total perimeter of the pond, yielding the proportion of that vegetation type around the entire pond (Table 4). Aquatic vegetation was assessed in 1m² quadrats at 20m intervals around the shoreline of the ponds. Quadrats were placed on the waterline extending 1m into the water. Percent cover of all vegetation types within the quadrat was visually estimated. All quadrats for a pond were averaged together to obtain the average percent cover for each pond.

2.5 Statistical Analyses

All data collected for the three groups of ponds were analyzed in two different ways. First, data from the 20-year and 30-year-old mined ponds were combined into one category (mined) and compared with the reference ponds (unmined). Second, data from the 20-year and 30-year-old ponds were separated and compared to each other to examine how time since reclamation affected characteristics of the ponds, turtles and frogs. The reference ponds were not all created at the same time, nor were they all older than the mined ponds; they were therefore unsuitable for testing the effects of time since reclamation.

When necessary, I transformed variables to achieve normality. When data could not be transformed to meet parametric model assumptions, non-parametric equivalents were used. Proportional data were arc-sin square-root transformed before all analyses (percent shoreline vegetation, percent aquatic vegetation cover, percent adult turtles, and percent male turtles)(Sokal and Rohlf 1995). Contingency tables were performed by Vassar Stats online calculators. All other analyses were performed with IBMs SPSS version 19. Significant effects were accepted at α=0.05.

Chelydra serpentina were not included in analyses due to low sample sizes (2% of unique turtles), the absence of this species from most study ponds, and drastically different morphometrics from *Trachemys scripta.* For analyses involving the size and age of *T. scripta* I separated juvenile from adult turtles by carapace length. Males and females of this species exhibit sexual size dimorphism, therefore females with a carapace less than 16 cm were considered juveniles; males with a carapace length less than 10 cm were considered juveniles (Marchand and Litvaitis 2004, Ernst and Lovich 2009). For all analyses, turtle density refers to the total number of unique turtles captured.

Chi-squared contingency tables were used to examine turtle categorical data (age and sex) among ponds. Yate's adjusted chi-squared values were reported (Sokal and Rohlf 1995). Differences in non-categorical turtle population characteristics were evaluated among ponds with Kruskal-Wallis tests (turtle species richness and mean carapace length). ANOVAs and Kruskal-Wallis tests were used to find differences in total frog species richness and frog family richness among ponds. Differences in environmental variables among ponds were evaluated with ANOVAs.

To reduce the colinearity of variables that describe the pond habitats, I ran a principle components analysis (PCA) with the environmental variables (distance to the nearest paved road, distance to a larger water source, secchi depth, percent shoreline vegetation, percent aquatic vegetation cover, and pond surface area). This procedure characterized the environmental variation in the ponds as well as generated orthogonal axes that were suitable to use in a multiple regression. The components with eigenvalues >1 were run in a forward and backward stepwise regression model selection with turtle density and again with frog species richness; both tests were run with a p-value to enter and p-value to exit of 0.15.

I evaluated the relationship between turtle population characteristics (turtle density, mean carapace length, percent adults, and percent males) and frog species richness with the environmental variables of the ponds (percent aquatic vegetation cover, percent shoreline vegetation, basking site availability, pond surface area, secchi depth, distance to the nearest paved road, and distance to a larger water source) using Pearson’s product-moment correlations and Spearman’s rho correlations. Total frog species richness as well as frog species richness within each family [hylids (tree frogs), ranids (true frogs), and bufonids (true toads)] were tested.

CHAPTER 3

RESULTS

I captured turtles 887 times, representing 416 unique turtles (overall recapture rate of 47%). Recapture rates at individual ponds ranged from 0-70% (Table 5). I encountered two turtle species, *Trachemys scripta* and *Chelydra serpentina.* Several species of turtles were not encountered that would have been expected in this area from previous studies and county records; *Kinosternon subrubrum, Apalone spinifera,* and *Pseudemys concinna* (Bradley 1987, Jasper 1992, Dixon 2000, Table 1). Eight turtles moved between study ponds (Table 6, Figures 8, 9). All of these turtles were large adults (>20 cm carapace length) and all but one were female, supporting previous studies' claims that females are more mobile than males (Bodie and Semlitsch 2000). The proportion of adult turtles at a pond was positively correlated with mean carapace length ($r=0.933$, $p<0.001$). Carapace length frequency distribution for *T. scripta* was bimodal with a primary mode of approximately 10.5 cm and a secondary mode of approximately 23.5 cm, reflecting the juvenile and adult portions of the populations (Glorioso et al. 2010).

A total of 8 frog species were encountered at the study ponds including three species of tree frogs, two species of toads, and three species of true frogs (Table 7). Several additional species that were expected to be found in these areas from previous studies and county records were not, most notably species that require heavy rain events to emerge from burrows to breed (Bradley 1987, Jasper 1992, Dixon 2000, Table 1). I caught tadpoles at three ponds; *Lithobates catesbeianus* at Egret, and *Lithobates sphenocephalus* at both C-90 and Skinny. All tadpoles represented species that were also detected with call surveys. Sample sizes of tadpoles were too low to perform statistical analyses.

3.1 Effects of Mining

Mined ponds tended to have larger turtles and fewer turtle species compared to unmined ponds. The less common of the two turtle species, *Chelydra serpentina*, was found almost exclusively at unmined ponds. Mined ponds have turtles with significantly longer carapaces than unmined ponds ($\chi^2=7.752$, $p=0.005$), with a mean of 16.7±0.4 cm compared to 15.5±0.5 cm (Table 5). There was no difference in the ratio of juveniles to adults or males to females between mined and unmined ponds.

Though frog species richness did not differ between mined and unmined ponds, six species at mined ponds compared to seven species at unmined ponds, *Anaxyrus woodhousii* and *Hyla chrysoscelis/versicolor* were only found at unmined ponds and *Incilius nebulifer* was only found at mined ponds (Table 7).

Unmined ponds had more trees along their shorelines than mined ponds ($F_{1,\ 10}=5.248$, $p=0.04$). There was no difference between mined and unmined ponds for most of the environmental variables, probably due to small sample sizes and a wide variability of environmental characteristics within pond groups. Water temperature, a variable that might have an effect on ectotherms, was similar for all ponds in the spring, indicating that it might be similar throughout the rest of the year. However, pond depth, a variable not measured in this study, would affect water temperature since deeper bodies of water are more resistant to air temperature changes. Although not an effect of mining, unmined ponds were closer to a large source of water compared to mined ponds ($F_{1,\ 10}= 11.332$, $p=0.007$; Table 2).

3.2 Effects of Pond Age

Turtle population characteristics differed between 20 year and 30 year ponds; older ponds tended to have older, larger turtles. Twenty year ponds had more juveniles than 30 year ponds ($\chi^2_{0.05,\ 315}= 7.35$, $p=0.007$, Table 5). Turtles in the 30 year ponds had significantly longer carapaces than turtles in 20 year ponds ($\chi^2=6.753$, $p=0.009$), with a mean of 17.9±0.5 cm compared to 16.0±0.5 cm (Table 5). There was no significant difference in turtle species richness or sex distribution between pond age classes.

Frog species richness did not differ significantly between pond age classes, with 20 year ponds and 30 year ponds both supporting five species of frogs (Table 7), but there were differences in species composition and frequency of observation (Table 8). *Hyla cinerea* was found at 30 year ponds but not 20 year ponds, and *Lithobates clamitans* was found at 20 year ponds but not 30 year ponds.

There were no significant environmental differences between the ponds due to age since reclamation. The variability in the environmental characteristics of the 20 year and 30 year ponds and small sample size may account for the lack of differences found. Aquatic vegetation cover and secchi depth were particularly variable within age groups. Although not an effect of pond age, ponds in the 30 year age category were farther from the nearest paved road than 20 year ponds ($F_{1,6}=9.056$, $p=0.02$; Table 2).

3.3 Effects of the Environment

To determine if ponds had different environments independent of their history of mining and reclamation, the variation in environmental variables among all ponds was evaluated with a PCA. The first four components of the PCA explained 76% of the variation in the environmental variables (Table 9). The variables that explained the most variance in pond environment across principal component one were percent of reeds at the shoreline, distance to the nearest paved road, distance to a larger water source, and pond surface area (Table 9). The variables that explained the most variance across principle component two were a clear shoreline, aquatic vegetation cover, and percent herbaceous shrubs at the shoreline. The data from the PCA results suggest that unmined ponds are associated with more herbaceous shrubs and aquatic vegetation cover than mined ponds. Mined ponds tended to have a clearer shoreline than unmined ponds (Figure 10). Two of the mined ponds did not fit the trend of the other mined ponds. Of the 30 year ponds, Skinny pond is an outlier along principal component two due to an uncharacteristically high proportion of herbaceous shrubs along the shoreline combined with a much greater proportion of aquatic vegetation cover compared to other mined ponds (Table 4). Of the 20 year ponds, C-90 pond is an outlier along both principle components. It had the most reeds of any pond, as well as almost 100% aquatic vegetation cover, both of which were abnormal for mined ponds.

Hardly any of the environmental variables were correlated with each other. Reeds were negatively correlated with distance to the nearest paved road (r_s=-0.661, p=0.02), and secchi depth was positively correlated with percent aquatic vegetation cover (r_s =0.747, p=0.005, Table 10). These results were likely affected by the variation in some of the data, as well as the fact that pond management may have differed among ponds.

Many of the turtle population characteristics correlated with environmental variables. Turtle density was unexpectedly positively correlated with distance to a larger water source (r_s =0.725, p=0.008). Mean carapace length was positively correlated with secchi depth (r_s =0.797, p=0.002), suggesting that turtles were able to grow larger in less turbid water. The proportion of adults was positively correlated with pond surface area (r=0.606, p=0.04, Table 10), not surprisingly showing that larger ponds were able to support larger turtles. Turtle populations did not correlate with any of the vegetation variables, though the positive correlation with secchi depth may be indicative of a positive trend with aquatic vegetation cover. None of the environmental PCA axes were significant predictors of turtle density.

Total frog species richness and frog family richness were also related to environmental variables somewhat independent of mining history. The relationships of environmental variables to species richness changed depending on the family of frog. Total frog species richness was positively correlated with aquatic vegetation cover (r_s =0.618, p=0.03) and negatively correlated with percent clear shoreline (r=-0.658, p=0.02). Hylid species richness followed this pattern by being negatively correlated with percent clear shoreline (r=-0.598, p=0.04) and positively correlated with percent aquatic vegetation cover (r_s=0.783, p=0.003). Hylid species richness was additionally positively correlated with secchi depth (r_s=0.589, p=0.04) and percent reeds at the shoreline (r_s=0.583, p=0.047). Ranid species richness was positively correlated with the availability of basking sites (r_s=0.649, p=0.02). Bufonid species richness was negatively correlated with distance to a larger water source (r_s -0.579, p=0.048, Table 10).

The second component of the PCA was a significant predictor of frog species richness (r^2=0.335, p=0.049). The positive eigenvectors of component two that determined this relationship were aquatic vegetation cover and percent herbaceous shrubs at the shoreline; the highest negative eigenvector was percent clear shoreline (Table 9). These environmental variables likely reveal the most favorable type of habitat for many of the frog species sampled.

3.4 Non-Focal Species Occurrence

Water snakes were found at all ponds except Drain Pipe. *Nerodia rhombifer* was most commonly encountered, followed by *Nerodia eurythrogaster* and *Nerodia fasciata*. *Nerodia rhombifer* was witnessed preying on *Hyla cinerea* at Skinny pond. Fish were found at all ponds except Drain Pipe and Skinny. Large carp were found in Fence East which began dying off in the summer of 2011 as the water level in the pond decreased. Off Road pond was stocked with young catfish once during the study period; stocking rates for all other ponds are unknown. Crayfish were found at several of the ponds, most abundantly at Open pond. Some turtles sampled from Egret, Left Tree, Skinny, and Goose Neck had leeches attached to their carapaces or plastrons.

CHAPTER 4

DISCUSSION

4.1 Mined Ponds Compared to Unmined Ponds – Reclamation

Mined and unmined ponds differed in environmental variables that might affect turtle and frog populations. Unmined ponds had, on average, more trees along their shorelines. Given that many species of herpetofauna rely on wooded habitat surrounding water sources, it was surprising that there were no correlations between shoreline trees and turtle and frog populations. For example, tree frogs utilize this habitat for calling sites and refugia after breeding, and many species of turtle take advantage of the sheltered land under the canopy to lay their eggs. However, this relationship may not be as simple as the amount of tree cover along the shoreline of the pond. Trees nearby, but not at the shoreline, were not recorded in this study, but might have a strong impact on these communities. A lack of tree cover in a habitat might result in a decrease in herpetofauna species richness, or at least a different composition of species (Rios-Lopez and Aide 2007). Mined ponds were reclaimed as pastureland, the plans for which do not typically include large stands of trees. Conversely, the unmined pond habitat has been managed to protect the post oak stands since 1950 and before that time was not extensively cleared (www.tpwd.state.tx.gov). This has allowed trees to surround most of the ponds in the unmined area. If this land had been assigned a different land use for reclamation, such as forestry or wildlife habitat, it would have been planted with more trees. As such, the difference between ponds in tree cover is not due to mining itself, but due to the land-use type the area was assigned after being mined.

Unmined ponds were located closer to a larger water source than mined ponds, which is also not an effect of mining. Catfish Creek runs to the east of the unmined ponds, and the Trinity River runs to the west. This proximity to two source populations makes it more likely that unmined ponds would be colonized by more species of turtles and frogs. This proximity may have facilitated the establishment of *Chelydra serpentina* at unmined ponds. If lotic turtle species such as *Pseudemys concinna* and *Apalone spinifera* were able to colonize these ponds, they would likely have come from either of those two sources. Previous studies show that these species are commonly caught with the trapping method used in this study, indicating that their absence is probably not due to sampling bias (Bodie et al. 2000, Rizkalla and Swihart 2006, Aresco 2009, Glorioso et al. 2010). Since the unmined ponds have not been colonized, the lack of typically lotic species in the mined ponds is probably not due to mining.

4.1.1 Turtles

The effects of mining and subsequent reclamation on turtle populations are not clear-cut. *Chelydra serpentina* was found almost exclusively at unmined ponds. This result supports my hypothesis that more species would be found at unmined ponds, but with only two species in the analyses it was difficult to test this hypothesis. In general, unmined ponds had more aquatic vegetation and less turbid water than mined ponds, both of which are more common in *C. serpentina* habitat. *C. serpentina* prefers to hide underwater beneath objects such as roots, stumps, or tires, which might be the most important factor defining the range of this species in this study (Froese 1978). The three unmined ponds where *C. serpentina* was found (Fence East, Fence West, and Goose Neck) had more complexity than other ponds in the water along the shoreline, including submerged roots, tires, and bank burrows. Homogenization of pond banks during reclamation results in less shoreline complexity at mined ponds, and tree debris such as stumps and downed limbs are not present. This homogenous shoreline is a direct effect of the ponds being man-made during reclamation combined with the disturbance of cattle moving in and out of the water.

Mined ponds are able to support larger, and therefore older*,* *Trachemys scripta* individuals. These results may be exaggerated by Open pond, the smallest unmined pond; only two turtles were caught at this pond, both under 7 cm in carapace length. However, the mean carapace lengths of turtles in the other unmined ponds were also smaller than those from mined ponds (Table 5). Unmined ponds were much closer to wooded areas compared to mined ponds, potentially providing more cover for predators such as raccoons, opossums, skunks, and foxes. *Alligator mississippiensis,* the American alligator, is known to inhabit the unmined ponds at Gus Engeling WMA, and is well documented as being a predator of aquatic turtles (Janes and Gutzke 2002). The increased predation pressure may keep the population of turtles at unmined ponds younger and smaller than at mined ponds.

4.1.2 Frogs

Frog species richness did not differ between mined and unmined ponds, refuting my hypothesis. However, frog species composition was different. *Incilius nebulifer,* a species found only in mined areas of this study, is known to outcompete and displace *Anaxyrus fowleri* in anthropogenically disturbed habitat (Vogel and Pechmann 2010). *Anaxyrus woodhousii* and *A. fowleri* are closely related species (Fontenot et al. 2011), and anecdotal evidence suggests this same competitive displacement might happen between *I. nebulifer* and *A. woodhousii* (Fontenot, pers. comm.). In this case, the disturbance of mining could have had a direct influence on the species richness in this area.

Another difference in species composition was between tree frog species. *Hyla chrysoscelis/versicolor* has similar habitat requirements as *Hyla cinerea,* but was found at only one unmined pond, Fence West. The limited encounters with this species are likely due to a lack of suitable upland habitat composed of trees and to its cryptic nature (Gibbs 1998). This species was only found through visual surveys and never heard calling. The mottled coloration of *H. chrysoscelis/versicolor* makes it much more difficult to see compared to the bright green *H. cinerea*. Additionally, *H. chrysoscelis/versicolor* is known to have low reproductive success at agricultural ponds, probably due to the elevated phosphorus levels and higher turbidity associated with ponds used for watering cattle (Knutson et al. 2004). Agricultural ponds also support the invasive bullfrog, *Lithobates catesbeianus,* a known predator of *H.*

chrysoscelis/versicolor (Schwartz et al. 2000). *L. catesbeianus* was encountered at almost every pond except for the unmined Fence West, which might have made this pond a safe haven for *H. chrysoscelis/versicolor*.

In general, the typically steep slope of ponds built for agricultural purposes can be unsuitable for many amphibian species who use the shallows to breed, which might account for some of the species differences between mined and unmined ponds (Shulse et al. 2010). Many frog species choose to breed in ephemeral ponds because of the lack of predatory fish (Knutson et al. 2004). At least one of the mined ponds was stocked with fish, and several others had fish present which might explain the lack of certain species in these permanent ponds.

4.2 20 Year Old Ponds Compared to 30 Year Old Ponds – Succession

There were no significant environmental differences due to pond age between the ponds that were 20 and 30 years since reclamation. Since they were reclaimed as pastureland, the vegetation at these ponds probably takes less than 20 years to establish (quick growing grasses with little or no trees). If this land had been assigned a different land use, such as forestry or wildlife habitat, the results may have been different. Independent of time since reclamation, ponds that were 30 years since reclamation were farther from paved roads than those 20 years since reclamation. Thirty year old ponds being farther from paved roads may contribute to larger turtles being found at those ponds, as road mortality during turtle migration is well documented (Marchand and Litvaitis 2004).

4.2.1 Turtles

There was no significant difference in turtle species richness between pond ages. These results do not support my hypothesis that more species would be found at older ponds. The two species found at the mined ponds were *Trachemys scripta,* found at all 20 and 30 year ponds, and *Chelydra serpentina,* found at one 20 year pond. Previous studies suggest that 20 years would be enough time for most herpetofauna to repatriate an area, particularly if the vegetation had recovered (Heinen 1992, Rios-Lopez and Aide 2007, Carrozzino 2009). The difference in turtle species composition probably has more to do with prey availability and pond habitat than with pond age. As stated previously, *C. serpentina* prefers a complex shoreline with refugia such as tree roots and stumps which were much more common at unmined ponds (Froese 1978). This species is primarily a carnivore that prefers plenty of aquatic vegetation and clear water (Bodie et al. 2000). *C. serpentina* was only found at C-90; this pond was large (0.98 ha surface area), had lots of aquatic vegetation (81% cover), low turbidity (secchi depth 74.33 ± 10.84 cm), and was fairly unique compared to the other mined ponds. Evaluation of the PCA plot shows this pond as an outlier among the mined ponds, specifically because of the aforementioned characteristics (Figure 10). This pond also contained a multitude of prey items such as fish, crayfish, snails, and insects. These characteristics might account for the presence of *C. serpentina* in C-90 and its absence in other mined ponds.

Pond age had the expected effect on turtle populations, allowing larger, older turtles to persist at ponds that have been established for longer. These results support my hypothesis that larger turtles would be found at older ponds. Since these ponds had no environmental differences due to age since reclamation, these results are likely attributable almost solely to the age of the pond. Turtles are long-lived and continue to grow throughout their lifetime, with species such as *Trachemys scripta* living beyond 30 years and reaching over 30 cm in carapace length (Snider and Bowler 1992, Tucker et al. 2006). This indicates that these reclaimed ponds are suitable habitat for this species of turtle, and suggests they will continue to thrive long past reclamation. These results might also suggest a lack of recruitment from other ponds, or limited breeding success at the 30 year old ponds. *T. scripta* can have dramatic responses to a changing environment including changes in reproduction and immigration. During drought, these turtles have been shown to have fewer successfully producing females, and different residential adult populations compared to non-drought years (Scribner et al. 1995). The data gathered in this study may be representative of the effects of drought conditions rather than the effects of pond age.

4.2.2 Frogs

Though frog species richness did not differ between pond age groups, species composition did. These results do not support my hypothesis that more frog species would be found at older ponds.

Other studies have also found that age since restoration was not a good predictor of amphibian species richness (e.g., Lehtinen and Galatowitsch 2001). *Lithobates clamitans*, the bronze frog, was not found at ponds that were 30 year since reclamation, and was only found at Egret pond which was 20 years since reclamation. This may be due to the fact that this species typically inhabits permanent wetlands and three of the four 30 year ponds experienced drying events during the study. Egret is the largest pond in the study, and is therefore more resistant to water loss during drought. The shoreline of Egret pond was almost 10% herbaceous shrubs, coupled with 33% aquatic vegetation cover comprised in large part of lilies, providing the vegetation this species requires for depositing eggs and sheltering tadpoles. This reinforces the idea that habitat requirements are probably more important predictors of frog species richness than time since reclamation (Walker 1946).

Hyla cinerea, the green tree frog, was found in 30 year old ponds but not 20 year old ponds. This species prefers floating vegetation such as duckweed to deposit their eggs in, rather than the larger vegetation *Lithobates clamitans* prefers (Walker 1946, Garton and Brandon 1975). Not only is the pond itself important, but the habitat surrounding the pond is also important to amphibian species (Simon et al. 2009). *H. cinerea* retreats upland to forested habitat after breeding, which may explain the ponds at which it was found (Drain Pipe, Skinny, and Left Tree of the 30 year ponds). The individual found at Drain Pipe was caught by hand during the day and was very small. Drain Pipe is an inhospitable environment for this species due to its lack of aquatic and shoreline vegetation and extreme distance from trees, indicating that this frog was probably migrating to a better location after metamorphosing. Skinny and Left Tree have high percentages of aquatic vegetation, and were both very close to a small creek surrounded by trees along its banks. These ponds could provide good breeding and egg deposition habitat, while also being close to suitable upland habitat (Shulse et al. 2010). It is unlikely that land reclaimed as pastureland will be able to support all of the available frog species richness in a habitat due to the lack of trees and diverse water sources such as ephemeral ponds.

4.3 Environmental Difference of Ponds Independent of Past Mining Activity

Several environmental variables were correlated with turtle and frog populations independent of the habitat's mining history or time since reclamation. Most environmental variables were not correlated with each other; however secchi depth was positively correlated with aquatic vegetation cover. This relationship was expected; with increasing aquatic vegetation, less sunlight can penetrate into the water, resulting in less algae and clearer water (Jackson 2003).

4.3.1 Turtles

Some population characteristics of *Trachemys scripta* were influenced by the environment independent of mining history. These results partially support my hypothesis that turtle species would be influenced by habitat characteristics. The proportion of adult turtles at a pond was positively correlated with the pond's surface area. These results are probably related to the productivity and availability of resources at the pond. Larger ponds probably have more resources which can support larger, and therefore older, turtles.

The proportion of adult turtles at a pond was also positively correlated with secchi depth. Secchi depth may have an influence on predator-prey interactions in ponds. Turtle eggs and hatchlings are preyed upon by many animals such as fire ants, raccoons, opossums, foxes, birds, snakes, and alligators (Goodpaster and Hoffmeister 1952, Rose and Manning 1996, Janzen et al. 2000, Janes and Gutzke 2002, Ferrell 2006). Less turbid water, as indicated by an higher secchi depth reading, might allow hatchling turtles to see approaching predators more easily than turbid water (Turesson and Brönmark 2007). With predation risks lessened, more hatchlings would grow to adulthood.

The density of turtles at a pond was positively related to the distance of that pond to a larger water source. The nature of this relationship is unclear, and probably related to variables that were not measured in this study. Greater turtle densities would be expected closer to source populations (Simberloff and Wilson 1969), in this case Fairfield Lake or Catfish Creek, not farther away as is seen in these results.

Turtle population characteristics were not related to any other environmental variables, which emphasizes the generalist nature of *Trachemys scripta.* This species is widespread throughout the eastern half of the United States, and can be found in lentic and lotic habitats (Ernst and Lovich 2009).

4.3.2 Frogs

Overall frog species richness and frog species richness within families were influenced by the environment to a greater degree than turtles. Frogs are more sensitive to the environment when compared to turtles due to their permeable skin and biphasic life cycle. These results support my hypothesis that frogs would be affected by environmental variables that were not strictly related to land use history, specifically the vegetation in and around the pond. Frog species richness was positively related to aquatic vegetative cover and negatively related to a clear shoreline. All of the frog species encountered in this study lay their eggs in water and have a larval aquatic tadpole stage before metamorphosis. These life stages require aquatic vegetation to protect the eggs from predation and desiccation as well as to provide tadpoles with food and refugia. Adults require aquatic vegetation for the same reasons, cover from predation while calling and breeding, and habitat structure to decrease evaporative water loss (Vitt and Caldwell 2009).

Frog species richness was also correlated with the environmental variables that made up component two of the PCA. This component had positive loadings for aquatic vegetation cover and herbaceous shrubs along the shoreline, and negative loadings for a clear shoreline. This shows that frogs are more common at ponds that have plenty of vegetation in the pond as well as along the shoreline.

Hylids, the tree frogs, were negatively associated with a clear shoreline and positively associated with aquatic vegetation and reeds. Tree frogs in particular require complex vegetative structure like trees and reeds for calling perches and refugia after breeding, as well as floating or emergent aquatic vegetation to protect them while searching for a mate and to protect their eggs after they have been laid (Gunzburger and Travis 2004).

Ranid (true frog) presence was positively associated with the availability of basking sites. In this study, basking sites were considered structures emerging from the ponds such as rocks, tree limbs, and tires that were likely areas for turtles to utilize for basking. Some ranids, such as bullfrogs, *Lithobates catesbeianus,* are known to bask in order to thermoregulate, but they do not necessarily utilize the same structures that turtles would use. Typically these frogs will bask while floating in water or while sitting on the bare shore of the pond (Brattstrom 1963). For this relationship, basking site availability as measured in my study may be a proxy for the complexity of the shoreline. The ranids in this study are large frogs that may see shoreline basking sites as desirable areas to advertise for a mate, or the increased complexity of the habitat may allow more frogs to call in the same area while providing refugia from predation and desiccation. The basis for this relationship is unclear and might be due to variables that were not examined in this study.

Bufonids (true toads) were negatively associated with the distance of ponds to the nearest larger water source. In general, toads are less prone to desiccation than hylids or ranids, so it would be expected that they could disperse farther from their source population (Hillman 1980). However, the relationship between toads and distance to a larger water source is not clear and might be an effect of variables not sampled in this study; further study of the metapopulation dynamics of this system is needed in order to fully understand this relationship.

4.4 Previous Studies

I found a lower species richness of turtles and a different composition of frog species compared to previous studies conducted in the area. Previous sampling found *Kinosternon subrubrum, Apalone spinifera,* and *Pseudemys concinna* in addition to *Trachemys scripta* and *Chelydra serpentina* (Bradley 1987, Jasper 1992; Table 1). *A. spinifera*, the spiny softshell turtle, and *P. concinna*, the river cooter, accounted for less than 10% of the total number of turtles captured by Bradley (1987). These two species are typically found in lotic habitats like streams and rivers, which accounts for the small sample sizes since he trapped in ponds, and are not likely to recolonize a lentic habitat like cattle ponds. Neither of these species is known for long distance travel overland (Ernst and Lovich 2009), further decreasing the chances of recolonization. This may indicate that the individuals in the previous studies were migrating through the ponds, or were not able to establish populations in the pond habitat. Additionally, *A. spinifera* requires sandy habitat in which to lay eggs, whereas most of the soil surrounding the mined ponds was clay. *K. subrubrum,* the eastern mud turtle, is often found in farm ponds and is known to spend more time on land than many other aquatic turtles (Bennett et al. 1970). For this species, terrestrial habitat is just as important as aquatic habitat (Rizkalla and Swihart 2006). Studies have shown that *K. subrubrum* prefers terrestrial habitat with no grass, a moderate amount of canopy cover, and do not tend to choose habitat that has been heavily disturbed (Harden et al. 2009). Unfortunately for this species, the mined ponds have little to no canopy cover and are surrounded by a matrix of grass (Harden et al. 2009). *K. subrubrum* has also been known to be absent in habitat that seems otherwise suitable (Glorioso et al. 2010).

Frog species richness was similar compared to previous studies, but species composition was different. The most notable difference was the absence of *Hyla cinerea* from previous studies. This species was found at three of the 30 year ponds and none of the 20 year ponds during this study, suggesting that this species is a late colonizer in succession, most likely due to its need for trees in the surrounding habitat. *Scaphiopus holbrookii, Gastrophryne olivacea,* and

Gastrophryne carolinensis had differential detection among the studies which is probably explained by weather conditions discussed below.

4.5 Other Considerations

4.5.1 Drought

The drought experienced throughout Texas in 2011 could have had a major impact on the detectability of frog species. Previous studies in the area found three species of frogs that were not encountered in this study: *Gastrophryne olivacea,* the Great Plains narrowmouth toad, *Gastrophryne carolinensis,* the eastern narrowmouth toad, and *Scaphiopus holbrookii,* the eastern spadefoot toad (Bradley 1987, Jasper 1992). All three of these species are explosive breeders, only emerging from their burrows during heavy rain events (Greenberg and Tanner 2004, Saenz et al. 2006); there were no heavy rain events during the field seasons of this study. Extreme drought has also been associated with a marked decrease in amphibian egg clutch size, as well as a decline in adult migration (Palis et al. 2006, Piha et al. 2007).

The individual turtle movement between ponds documented in this study might be typical migrations, or might be a response to drying ponds (Gibbons et al. 1983, Bodie et al. 2000, Joyal et al. 2001, Gibbs and Shriver 2002). The lack of hatchlings found at most ponds might also indicate fewer females laying eggs due to the drought (Gibbons et al. 1983). Additional sampling of both turtles and frogs during non-drought years is needed to properly assess the impact of drought on this study.

4.5.2 Cattle

The presence of cattle at the ponds may be more influential at present than mining and reclamation in the past. Though ungulate grazing is an important process in many ecosystems, livestock grazing typically overtaxes ecosystems. Cattle are known for disturbing aquatic habitat by defecating in the water, uprooting aquatic and emergent vegetation, preventing trees from establishing along pond perimeters, increasing turbidity, and decreasing oxygen content of the water (Kauffman and Krueger 1984, Trimble 1994, Knutson et al. 2004). In addition to these general habitat impacts, the use of ponds by cattle might increase the prevalence of Frog Virus 3 in ranid tadpoles (Gray et al. 2007). In general, amphibian species richness and diversity has been shown to be greater in wetlands without cattle (Schmutzer et al. 2008), but cattle effects tend to vary by species. *Lithobates clamitans* captures have been shown to decrease with the presence of cattle, but the invasive and agriculturally tolerant bullfrog, *Lithobates catesbeianus,* has been shown to increase in numbers with the presence of cattle (Burton et al. 2009).

Turtle species also show differential responses to cattle presence. Morphology and egg size of *Chrysemys picta*, the painted turtle which is closely related to *Trachemys scripta*, was not negatively correlated with cattle impact, however *Kinosternon subrubrum* egg size was (Lindsay and Dorcas 2001). These differences in response may have to do with the diets of these two species. *K. subrubrum* eats mostly invertebrates which would be impacted by the low oxygen levels and high sedimentation in ponds associated with cattle, whereas *C. picta*, an herbivore, would not be affected in this way (Lindsay and Dorcas 2001).

4.6 Future Directions

The results of this study raise many questions that could be answered by future projects. I have assumed for this study that larger bodies of water are the source populations for many turtle and frog species. Sampling at Fairfield Lake, the Trinity River, and Catfish Creek would show what species actually inhabit these larger water bodies. Limited sampling of Fairfield Lake during this study showed that large *Trachemys scripta* are present. This study would also benefit from more seasons of data to include non-drought years. The absence of a particular frog species at a pond during one breeding season does not show that the species never inhabits that pond, and may have been a result of drought conditions. Additional sampling methods such as cover boards and pitfall traps would broaden the range of sampling for both turtles and frogs. A broader range of pond ages should be sampled as well. Land containing 10 year old ponds was not accessible for this study, but future projects might be able to sample those resources. This study used a time series that substituted distance for time, but future studies could focus on a longitudinal study with ponds that had just been reclaimed, and sample them for several seasons to assess the initial colonization of herpetofauna.

4.7 Conclusions

This study showed that turtle populations and frog species richness are probably more influenced by habitat characteristics independent of mining and time since reclamation than factors directly associated with them. The similarity of turtle and frog species richness and composition between 20 and 30 year old ponds shows that these animals are capable of repatriating an area in less than 20 years, though mature turtle communities may take longer to establish. This study also suggests that ponds reclaimed after coal mining are capable of supporting similar species richness' for turtles and frogs, but different species compositions compared to unmined ponds. In general, the species that are lacking in reclaimed areas are species that would need more trees, such as *Kinosternon subrubrum, Hyla cinerea,* and *Hyla chrysoscelis/versicolor* (Table 1). To encourage a higher diversity of herpetofauna on reclaimed pastureland, a buffer of trees around and perhaps connecting water sources should be added to reclamation plans. These trees would provide upland habitat needed by some species, as well as sheltered corridors for movement between ponds.

APPENDIX A

FIGURES

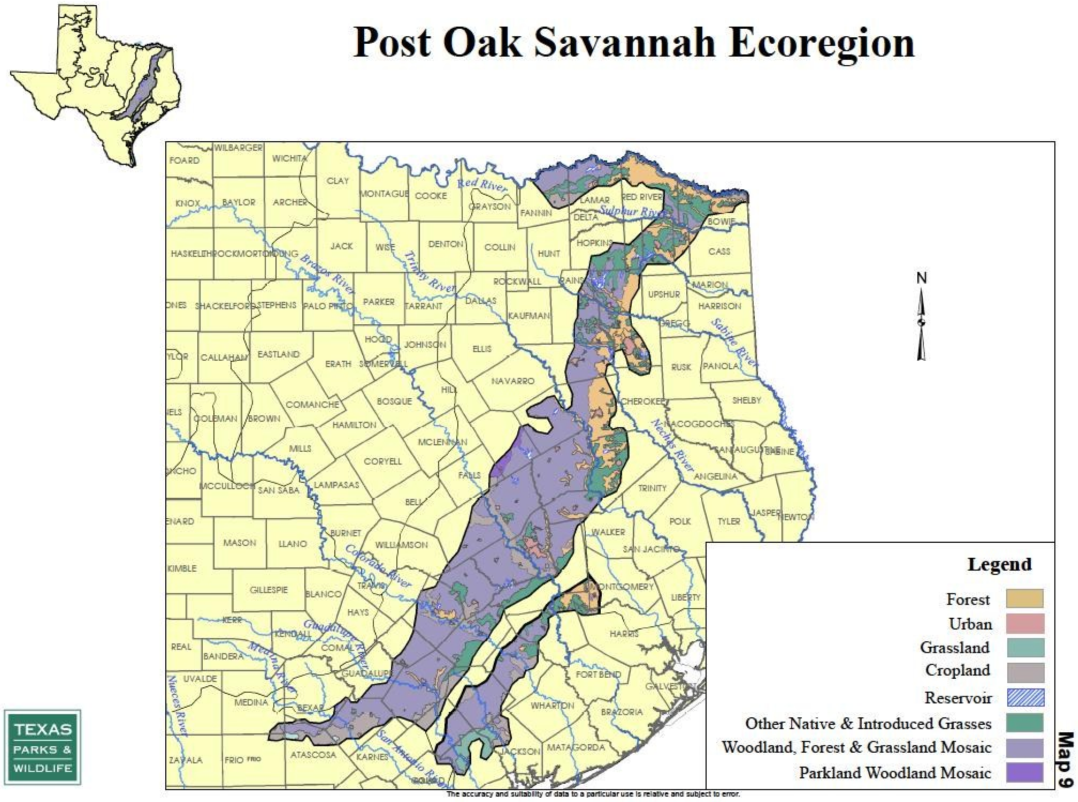
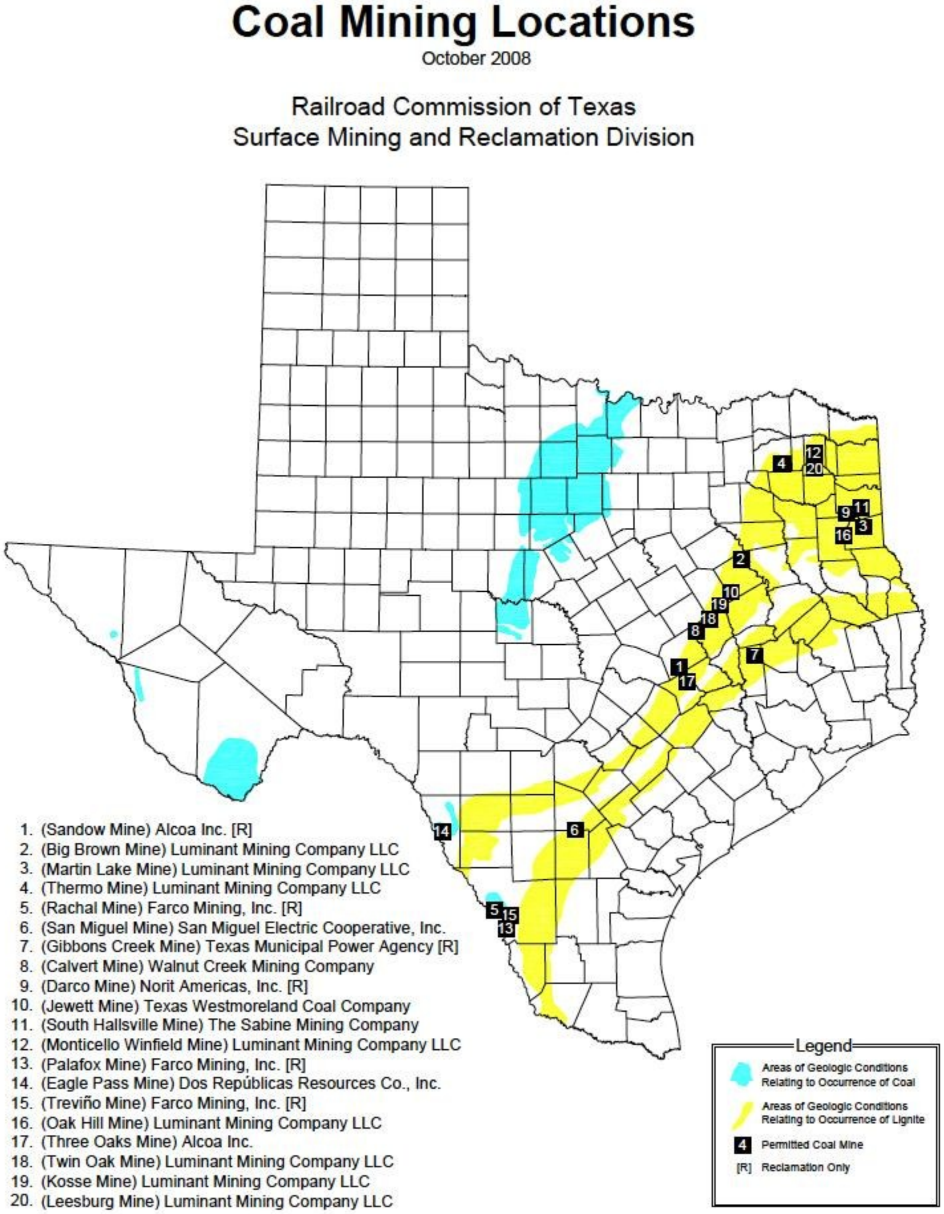


Figure 1. Post Oak Savannah ecoregion of Texas.



Coal Mining Locations

October 2008

Railroad Commission of Texas

Surface Mining and Reclamation Division

1. (Sandow Mine) Alcoa Inc. [R]
2. (Big Brown Mine) Luminant Mining Company LLC
3. (Martin Lake Mine) Luminant Mining Company LLC
4. (Thermo Mine) Luminant Mining Company LLC
5. (Rachal Mine) Farco Mining, Inc. [R]
6. (San Miguel Mine) San Miguel Electric Cooperative, Inc.
7. (Gibbons Creek Mine) Texas Municipal Power Agency [R]
8. (Calvert Mine) Walnut Creek Mining Company
9. (Darco Mine) Norit Americas, Inc. [R]
10. (Jewett Mine) Texas Westmoreland Coal Company
11. (South Hallsville Mine) The Sabine Mining Company
12. (Monticello Winfield Mine) Luminant Mining Company LLC
13. (Palafox Mine) Farco Mining, Inc. [R]
14. (Eagle Pass Mine) Dos Repúblicas Resources Co., Inc.
15. (Treviño Mine) Farco Mining, Inc. [R]
16. (Oak Hill Mine) Luminant Mining Company LLC
17. (Three Oaks Mine) Alcoa Inc.
18. (Twin Oak Mine) Luminant Mining Company LLC
19. (Kosse Mine) Luminant Mining Company LLC
20. (Leesburg Mine) Luminant Mining Company LLC

Figure 2. Seam of lignite coal in the state of Texas.

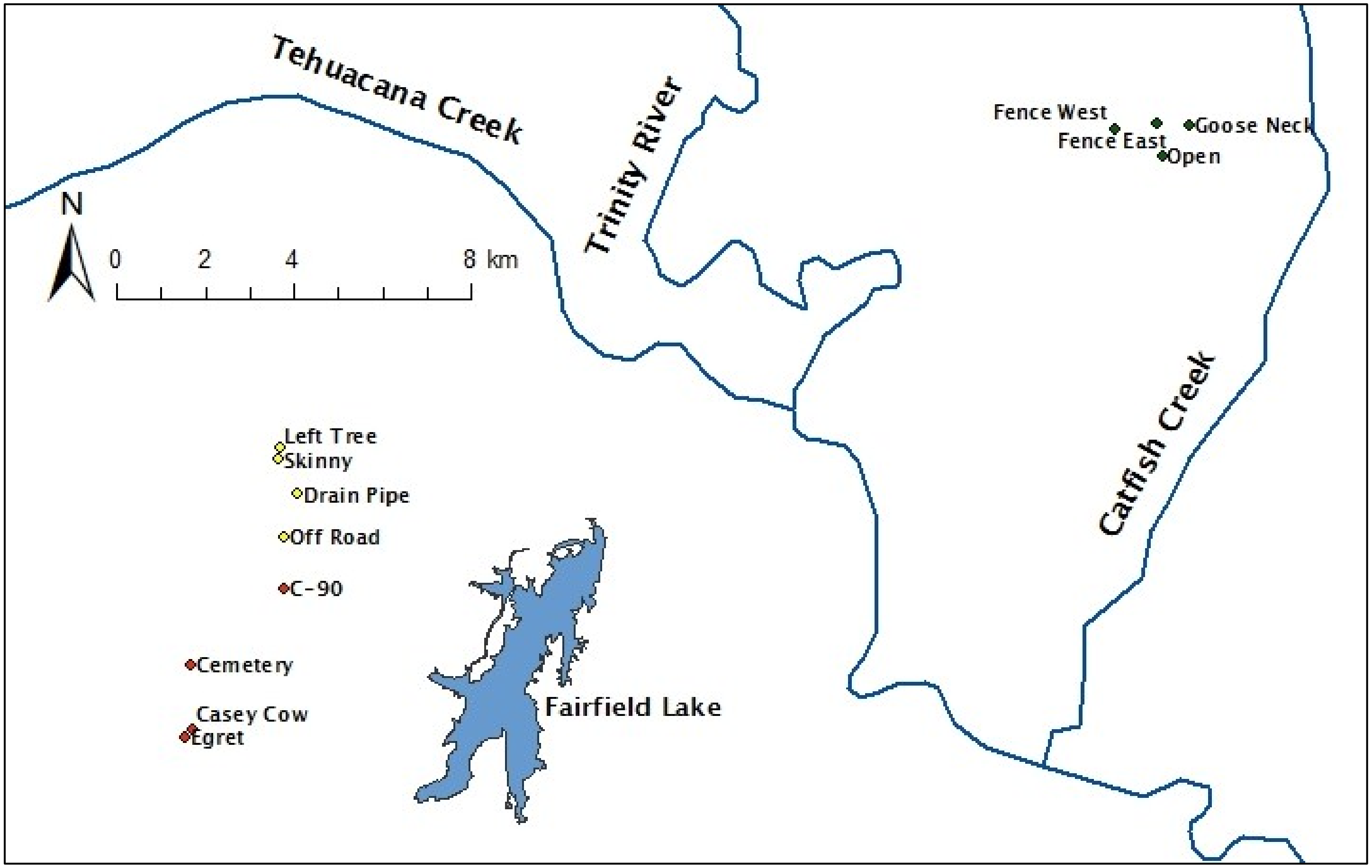
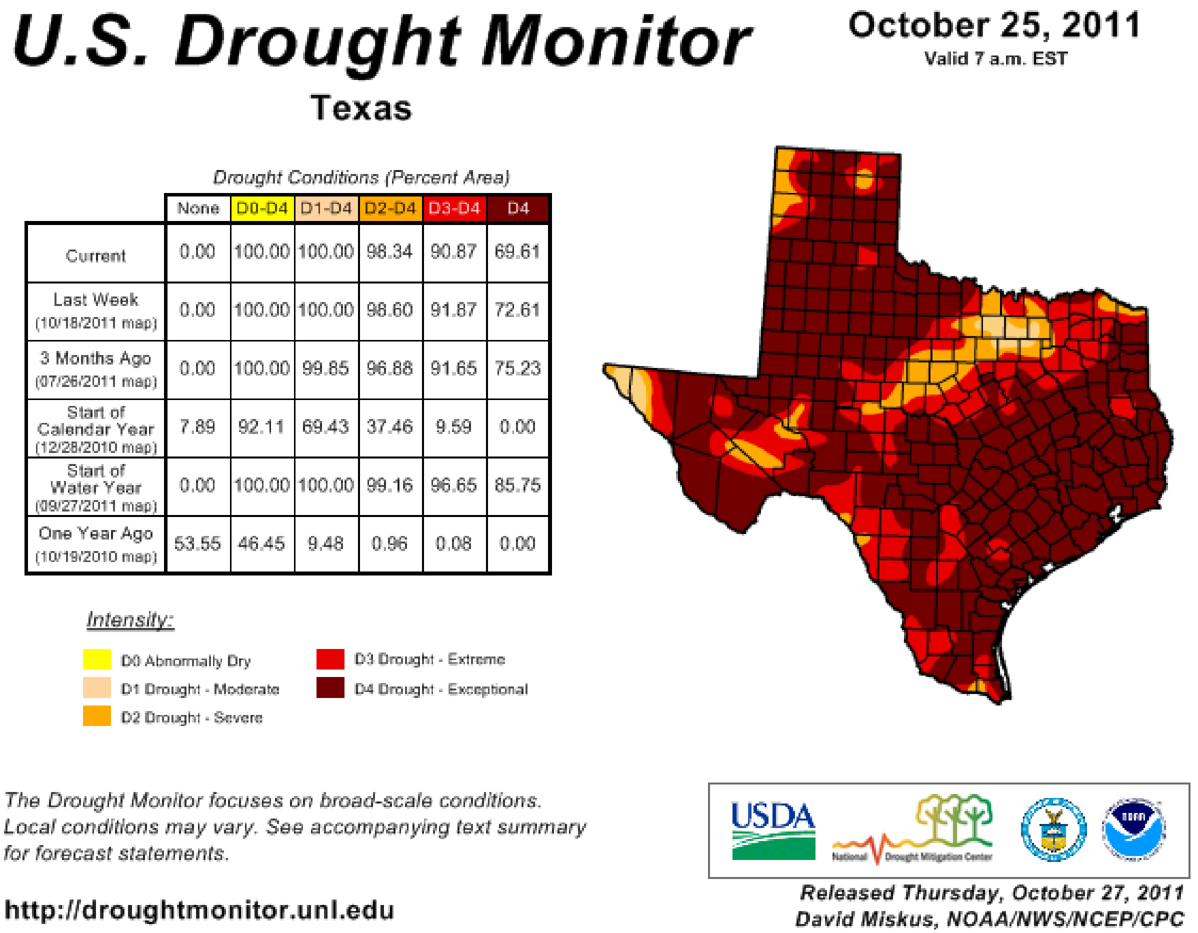


Figure 3. Location of study ponds. Red points indicate 30 year old mined ponds, yellow are 20 year old mined ponds, and green are unmined ponds.

Figure 4. Off Road pond (a) 7/21/10 before bulldozing and (b) 9/17/11 after bulldozing.



U.S. Drought Monitor

Texas

October 25, 2011
Valid 7 a.m. EST

Drought Conditions (Percent Area)

	None	D0-D4	D1-D4	D2-D4	D3-D4	D4
Current	0.00	100.00	100.00	98.34	90.87	69.61
Last Week (10/18/2011 map)	0.00	100.00	100.00	98.60	91.87	72.61
3 Months Ago (07/26/2011 map)	0.00	100.00	99.85	96.88	91.65	75.23
Start of Calendar Year (12/28/2010 map)	7.89	92.11	69.43	37.46	9.59	0.00
Start of Water Year (09/27/2011 map)	0.00	100.00	100.00	99.16	96.65	85.75
One Year Ago (10/19/2010 map)	53.55	46.45	9.48	0.96	0.08	0.00

Intensity:

- 🟨 D0 Abnormally Dry
- 🟧 D1 Drought - Moderate
- 🟧 D2 Drought - Severe
- 🟥 D3 Drought - Extreme
- 🟫 D4 Drought - Exceptional

The Drought Monitor focuses on broad-scale conditions. Local conditions may vary. See accompanying text summary for forecast statements.

http://droughtmonitor.unl.edu

Released Thursday, October 27, 2011
David Miskus, NOAA/NWS/NCEP/CPC

Figure 5. Drought conditions in Texas on October 25, 2011.

Figure 6. Drain Pipe pond (a) 7/21/10 before and (b) 9/17/11 after drought conditions.

Figure 7. Skinny pond (a) 7/21/10 before and (b) 9/17/11 after drought conditions.

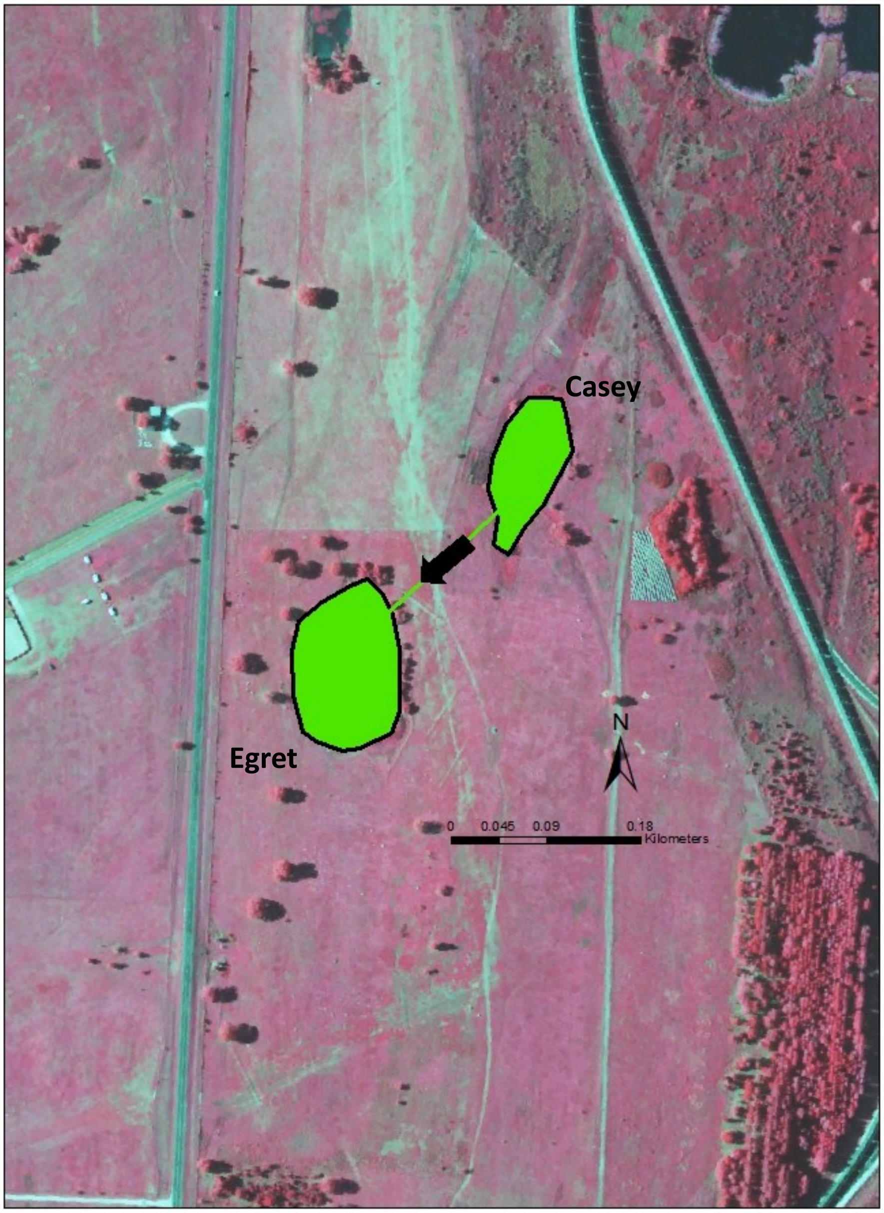


Figure 8. Turtle movement between Casey pond and Egret pond.

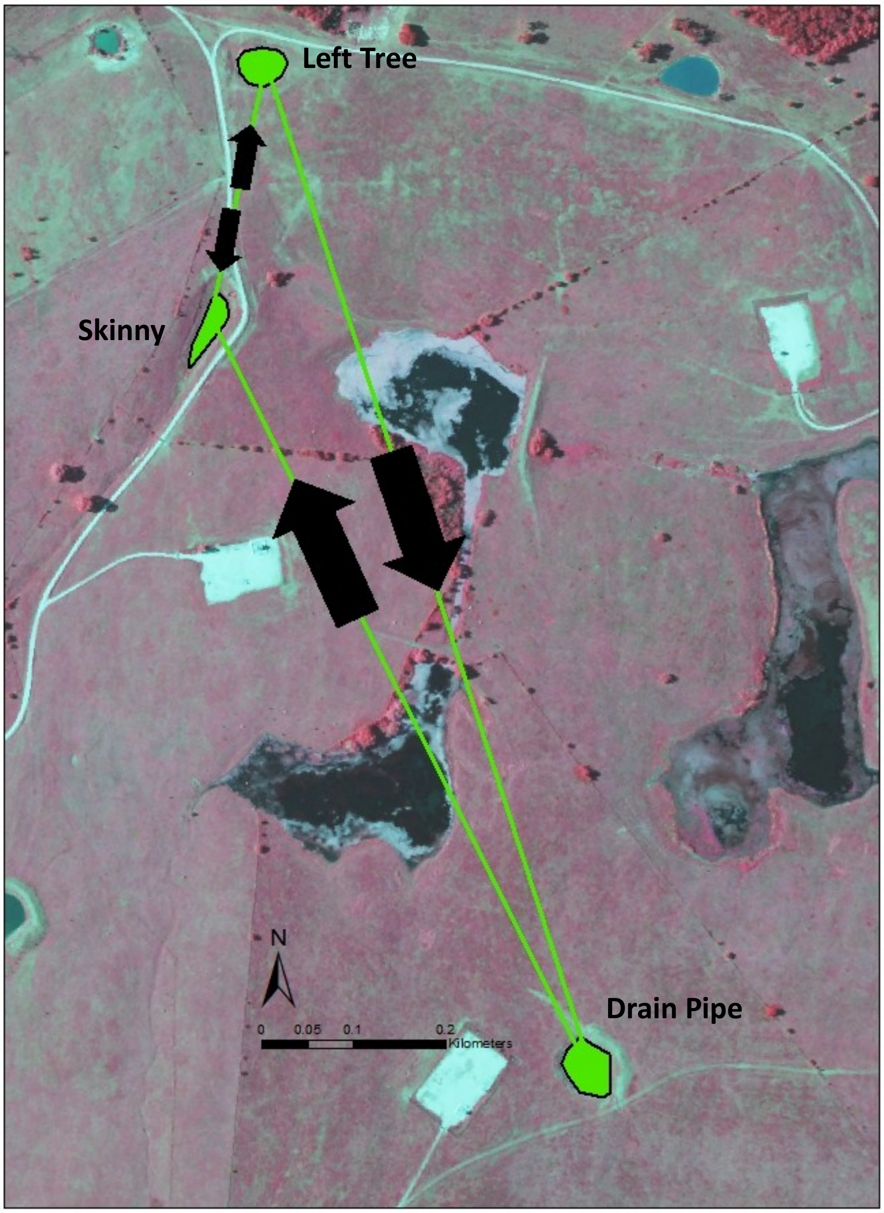


Figure 9. Turtle movement among Left Tree, Drain Pipe, and Skinny ponds.

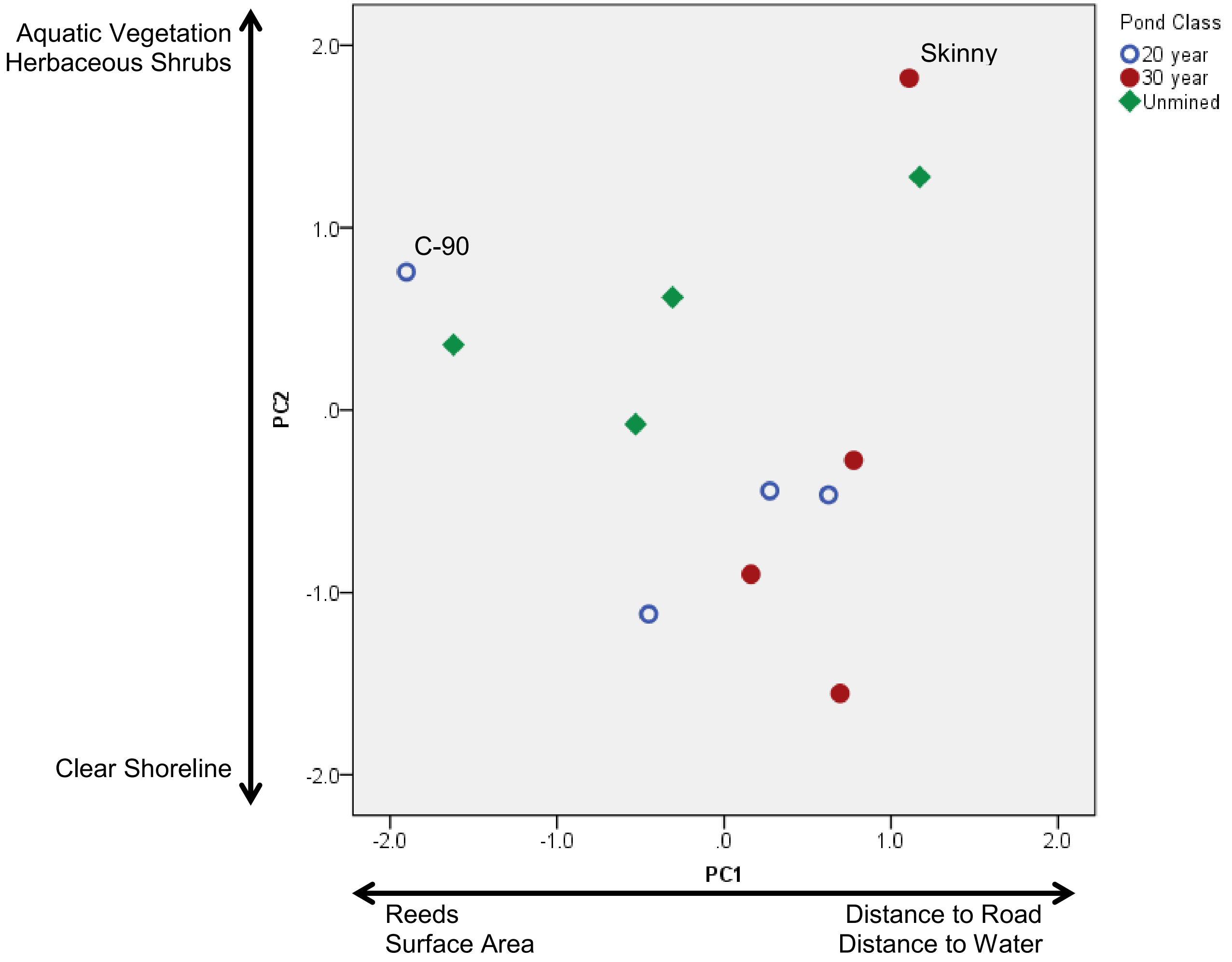


Figure 10. Principal components 1 and 2. Axis labels indicate the heaviest loading variables of each component. For example, increasing values on the x-axis are related to an increasing distance from the nearest paved road. See Table 8 for loadings on each component and explanation of variance.

APPENDIX B

TABLES

Table 1. Turtle and anuran species found in Anderson and Freestone counties. This list was compiled based on Dixon's county records of each species to date. Bradley and Jasper show species found in previous studies on the same land or nearby land as the current study, Walton.

	Dixon 2000		**Bradley 1987**	**Jasper 1992**	**Walton 2012**
	Common Name	**Species**			
Turtles	common snapping turtle	*Chelydra serpentina*	x		x
	alligator snapping turtle	*Macroclemys temminckii*			
	stinkpot	*Sternotherus odoratus*			
	razorback musk turtle	*Sternotherus carinatus*			
	eastern mud turtle	*Kinosternon subrubrum*	x*		
	yellow mud turtle	*Kinosternon flavescens*			
	Mississippi map turtle	*Graptemys pseudogeographica*			
	red eared slider	*Trachemys scripta*	x	x	x
	river cooter	*Pseudemys concinna*	x		
	Texas river cooter	*Pseudemys texana*			
	chicken turtle	*Deirochelys reticularia*			
	spiny softshell turtle	*Apalone spinifera*	x		
Frogs	Couch's spadefoot	*Scaphiopus couchii*			
	Hurter's spadefoot	*Scaphiopus hurteri*			
	Eastern spadefoot	*Scaphiopus holbrookii*	x*	x	
	Texas toad	*Bufo speciosus*			
	gulf coast toad	*Bufo nebulifer*	x		x
	redspotted toad	*Bufo punctatus*			
	Woodhouse's toad	*Bufo woodhousii*	x	x	x
	cricket frog	*Acris crepitans*	x*	x	x
	green tree frog	*Hyla cinerea*			x
	gray tree frog	*Hyla chrysoscelis/ versicolor*		x	x
	upland chorus frog	*Pseudacris triseriata*			
	spotted chorus frog	*Pseudacris clarkii*			
	Strecker's chorus frog	*Pseudacris streckeri*			
	spring peeper	*Pseudacris crucifer*			
	great plains narrowmouth toad	*Gastrophryne olivacea*	x		
	eastern narrowmouth toad	*Gastrophryne carolinensis*	x		
	bullfrog	*Lithobates catesbeianus*	x	x	x
	bronze frog	*Lithobates clamitans*			x
	southern leopard frog	*Lithobates sphenocephalus*	x	x	x
	pickerel frog	*Lithobates palustris*			
	southern crawfish frog	*Rana areolata*			

**Found on nearby land, not mine land*

Table 2. Physical characteristics of study ponds.

Status	**Age Class**	**Pond Name**	**County**	**Latitude**	**Longitude**	**Surface Area (ha)**	**Perimeter (m)**	**Distance to Paved Road (km)**	**Distance to Water Source (km)**
Mined	20 year	C-90	Freestone	31.813083	-96.117194	0.98	423	0.07	3.76
		Casey Cow	Freestone	31.784917	-96.139972	0.86	389	0.31	5.22
		Cemetery	Freestone	31.783278	-96.14175	0.12	128	0.14	5.33
		Egret	Freestone	31.798111	-96.14	1.50	456	0.28	5.33
	Average					0.87 ± 0.28	349 ± 75	0.20 ± 0.06	4.91 ± 0.38
	30 year	Drain Pipe	Freestone	31.832083	-96.113556	0.27	197	1.27	4.33
		Left Tree	Freestone	31.841778	-96.117639	0.18	164	1.26	5.24
		Off Road	Freestone	31.823417	-96.117167	0.27	212	0.25	4.29
		Skinny	Freestone	31.839167	-96.118167	0.15	201	1.27	5.11
	Average					0.22 ± 0.03	194 ± 10	1.01 ± 0.25	4.74 ± 0.25
Average						0.54 ± 0.18	271 ± 45	0.60 ± 0.20	4.83 ± 0.21
Unmined		Fence East	Anderson	31.903139	-95.90725	0.45	328	0.59	3.50
		Fence West	Anderson	31.902333	-95.917194	0.20	231	0.19	4.44
		Goose Neck	Anderson	31.902556	-95.899389	0.29	322	0.15	2.76
		Open	Anderson	31.896556	-95.905944	0.17	163	0.80	3.42
Average						0.28 ± 0.06	261 ± 39	0.44 ± 0.16	3.53 ± 0.35

Table 3. Environmental characteristics of the study ponds.

Status	**Age Class**	**Pond**	**Mean Secchi Depth (cm)**	**Basking Sites[1]**	**Soil Type[2]**	**Mean Aquatic Vegetation Cover**	**Dominant Shoreline Vegetation**
Mined	20 year	C-90	74.3 ± 10.8	low	BoC	0.81	Reeds
		Casey Cow	37.0 ± 3.0	med	BoC	0.00	Clear
		Cemetery	7.5 ± 1.0	med	BoC	0.00	Clear
		Egret	40.3 ± 4.0	high	BoC	0.33	Clear
	Average		37.5 ± 6.4			0.28 ± 0.19	
	30 year	Drain Pipe	8.5 ± 1.3	low	BoC	0.00	Clear
		Left Tree	57.3 ± 8.4	low	BoC	0.87	Clear
		Off Road	11.3 ± 4.4	low	BoC	[3]	Clear
		Skinny	51.8 ± 14.0	med	BoC	1.00	Herbaceous Shrubs
	Average		35.4 ± 7.5			0.62 ± 0.31	
Average			36.5 ± 4.8			0.43 ± 0.16	

Table 3 Continued

Status	Pond	Mean Secchi Depth (cm)	Basking Sites[1]	Soil Type[2]	Mean Aquatic Vegetation Cover	Dominant Shoreline Vegetation
Unmined	Fence East	20.3 ± 5.3	med	Fs	0.24	Trees[4]
	Fence West	26.3 ± 6.6	low	LuA	0.58	Trees
	Goose Neck	27.8 ± 11.7	high	LuA	0.90	Trees
	Open	19.0 ± 2.7	low	TpC	0.00	Woody Shrubs
Average		22.5 ± 3.5			0.43 ± 0.20	

[1] Basking site key:
low= no basking sites
med= 1-5 basking sites
high= >5 basking sites

[2] Soil type key:
BoC = Big brown silty clay loam
LuA = Lufkin fine sandy loam
TpC = Trep loamy fine sand
Fs = Freestone-Lufkin complex

[3] Aquatic vegetation cover for Off Road could not be measured.
The pond dried up before this measurement could be taken.

[4] Shoreline vegetation cover for Fence East was obtained through satellite orthoimagery.

Table 4. Percent of the shoreline composed of different vegetation types.

Status	20 year	Pond	Clear	Trees	Reeds	Herbaceous Shrubs	Woody Shrubs
Mined		C-90	0.12	0.00	0.88	0.00	0.00
		Casey Cow	0.41	0.32	0.00	0.27	0.00
		Cemetery	0.67	0.12	0.00	0.21	0.00
		Egret	0.61	0.31	0.00	0.08	0.00
	Average		0.45 ± 0.12	0.19 ± 0.08	0.22 ± 0.22	0.14 ± 0.06	0.00
	30 year	Drain Pipe	1.00	0.00	0.00	0.00	0.00
		Left Tree	0.81	0.19	0.00	0.00	0.00
		Off Road	0.79	0.21	0.00	0.00	0.00
		Skinny	0.00	0.00	0.00	1.00	0.00
	Average		0.65 ± 0.22	0.10 ± 0.06	0.00	0.25 ± 0.25	0.00
Average			0.55 ± 0.12	0.14 ± 0.05	0.11 ± 0.11	0.20 ± 0.12	0.00
Unmined		Fence East	0.10	0.90	0.00	0.00	0.00
		Fence West	0.03	0.82	0.00	0.15	0.00
		Goose Neck	0.12	0.73	0.15	0.00	0.00
		Open	0.07	0.00	0.00	0.09	0.84
Average			0.08 ± 0.02	0.61 ± 0.21	0.04 ± 0.04	0.06 ± 0.04	0.21 ± 0.21

Table 5. Turtle capture and population data for each pond.

Status	**Age Class**	**Pond**	***Trachemys scripta***	***Chelydra serpentina***	**Total Captures**	**Total Unique Turtles**	**Proportion Recaptures**
Mined	20 year	C-90	x	x	9	9	0.00
		Casey	x		36	31	0.14
		Cemetery	x		136	95	0.30
		Egret	x		76	68	0.11
	Average				64.25 ± 27.59	50.75 ± 19.12	0.21
	30 year	Drain Pipe	x		39	28	0.28
		Left Tree	x		81	55	0.32
		Off Road	x		15	14	0.07
		Skinny	x		37	18	0.51
	Average				43 ± 13.78	28.75 ± 9.23	0.33
Average					53.63 ± 14.83	39.5 ± 10.8	0.47
Unmined		Fence East	x	x	75	44	0.40
		Fence West	x	x	99	28	0.70
		Goose Neck	x	x	33	24	0.27
		Open	x		3	2	0.33
Average					52.5 ± 21.41	24.5 ± 8.66	0.53

Table 5 Continued

Status	Age Class	Pond	Mean Mass*	Mean Carapace Length*	Mean Plastron Length*	Proportion Male*	Proportion Adult*
Mined	20 year	C-90	1.84 ± 0.27	22.0 ± 0.9	20.3 ± 0.8	0.14	1.00
		Casey	1.34 ± 0.08	20.5 ± 0.4	18.9 ± 0.0	0.45	1.00
		Cemetery	0.46 ± 0.07	11.6 ± 0.6	10.6 ± 0.6	0.40	0.37
		Egret	1.31 ± 0.08	19.4 ± 0.6	17.8 ± 0.5	0.57	0.93
	Average		0.91 ± 0.06	16.0 ± 0.5	14.6 ± 0.4	0.46	0.68
	30 year	Drain Pipe	1.09 ± 0.18	17.1 ± 1.3	15.5 ± 1.2	0.46	0.64
		Left Tree	1.11 ± 0.09	19.1 ± 0.5	17.5 ± 0.5	0.42	0.93
		Off Road	0.44 ± 0.14	11.3 ± 1.5	10.4 ± 1.4	0.71	0.64
		Skinny	1.47 ± 0.18	20.6 ± 1.1	19.0 ± 1.1	0.28	0.89
	Average		1.07 ± 0.08	17.9 ± 0.5	16.4 ± 0.5	0.44	0.82
Average			0.97 ± 0.04	16.7 ± 0.4	15.3 ± 0.3	0.45	0.73
Unmined		Fence East	0.74 ± 0.08	16.0 ± 0.7	14.6 ± 0.7	0.53	0.81
		Fence West	0.53 ± 0.07	14.2 ± 0.8	15.6 ± 0.8	0.44	0.74
		Goose Neck	0.80 ± 0.13	17.0 ± 1.0	15.6 ± 0.9	0.55	0.75
		Open	0.05 ± 0.01	6.0 ± 0.6	5.4 ± 0.5	0.50	0.00
Average			0.67 ± 0.05	15.5 ± 0.5	14.1 ± 0.5	0.51	0.76

*Mass, carapace length, plastron length, proportion adult, and proportion male do not include *Chelydra serpentina*

Table 6. Turtle movement between ponds. All dates are from 2011.

Turtle	Pond	Date	Distance (m)	Sex	Carapace Length
59	Casey	3/19	129	Female	20.3
	Egret	6/22			
152	Casey	5/16	129	Female	23.2
	Casey	5/18			
	Egret	6/21			
258*	Left Tree	6/7	1091	Female	25.3
	Drain Pipe	6/29			
113	Left Tree	4/8	219	Male	20.7
	Left Tree	6/29			
	Skinny	5/23			
	Skinny	6/27			
	Skinny	6/29			
216	Skinny	5/25	219	Female	23.5
	Skinny	6/7			
	Skinny	6/27			
	Left Tree	6/6			
263	Left Tree	6/8	219	Female	22.3
	Skinny	6/29			
369	Skinny	6/28	219	Female	24
	Left Tree	6/29			
261*	Drain Pipe	6/7	846	Female	22.3
	Skinny	6/29			

*May be the result of unclear markings and not an actual migration event.

Table 7. Frog species occurrence by pond.

			Bufonids		**Ranids**		
Status	**Age Class**	**Pond**	***Anaxyrus woodhousii***	***Incilius nebulifer***	***Lithobates sphenocephalus***	***Lithobates catesbeianus***	***Lithobates clamitans***
			Woodhouse's toad	Gulf Coast Toad	Leopard frog	Bullfrog	Bronze frog
Mined	20 year	C-90		x	x	x	
		Casey Cow				x	
		Cemetery		x	x	x	
		Egret			x	x	x
	Total						
	30 year	Drain Pipe				x	
		Left Tree				x	
		Off Road		x		x	
		Skinny			x	x	
	Total						
Total							
Unmined		Fence East	x			x	
		Fence West					
		Goose Neck	x*		x		x
		Open	x		x	x	
Total							

*This specimen may have been a *Gastrophryne carolinensis*.

Table 7 Continued

				Hylids		
Status	**Age Class**	**Pond**	***Acris crepitans***	***Hyla cinerea***	***Hyla chrysoscelis/ versicolor*****	**Total species**
			Cricket frog	Green tree frog	Grey tree frog	
Mined	20 year	C-90	x			4
		Casey Cow	x			2
		Cemetery				3
		Egret	x			4
	Total					5
	30 year	Drain Pipe		x		2
		Left Tree	x	x		3
		Off Road				2
		Skinny	x	x		4
	Total					5
Total						6
Unmined		Fence East	x	x		4
		Fence West	x	x	x	3
		Goose Neck	x	x		5
		Open		x		4
Total						7

**This encounter was visual and these species are indistinguishable except for their call.

Table 8. The frequency of frog species observations at each pond.

Species	**20 year**	**30 year**	**Reference**
Acris crepitans	0.82	0.43	0.59
Anaxyrus woodhousii	0	0	0.06
Hyla chrysoscelis/versicolor	0	0	0.03
Hyla cinerea	0	0.21	0.50
Incilius nebulifer	0.11	0.04	0
Lithobates catesbeianus	0.50	0.46	0.16
Lithobates clamitans	0.04	0	0.09
Lithobates sphenocephalus	0.25	0.07	0.16

Table 9. Principle component analysis component matrix. Eigenvectors <0.1 are suppressed.

	Component			
	1	2	3	4
Reeds	-.757	.281	-.160	.499
Distance to Nearest Road	.748			-.272
Pond Surface Area	-.579	-.314		.329
Clear Shoreline	.262	-.889		.179
Woody Shrubs	.369	.403	-.742	
Aquatic Vegetation Cover	-.349	.556	.566	
Herbaceous Shrubs	.461	.543	.564	.162
Distance to Water Source	.519	-.291	.559	.442
Secchi Depth	.376	.379	-.382	.307
Trees	-.423	-.134	.179	-.729
Eigenvalues	2.596	1.980	1.726	1.314
% of Variance Explained	25.962	19.796	17.260	13.144
Cumulative % Variance Explained	25.962	45.758	63.018	76.162

Table 10. Correlations of turtle and frogs variables with environmental variables. For all tests N=12. cc=Correlation Coefficient. Values in italics were run with a Spearman's rho test. Values in plain text were run with a Pearson's r test. Bold=significant

		Aquatic Vegetation	Clear Shoreline	Trees	Reeds	Herbaceous Shrubs	Woody Shrubs	Proportion Adult Turtles	Proportion Male Turtles	Carapace Length
Clear Shoreline	cc	*-.399*								
Trees	cc	*.070*	.104							
Reeds	cc	*.495*	*-.330*	*.145*						
Herbaceous Shrubs	cc	*-.062*	*-.463*	*-.099*	*-.260*					
Woody Shrubs	cc	*-.317*	*-.306*	*-.356*	*-.172*	*.140*				
Proportion Adult Turtles	cc	*.495*	.007	.190	*.226*	*-.024*	*-.482*			
Proportion Male Turtles	cc	*-.326*	.347	.458	*-.229*	*-.377*	*.131*	-.345		
Carapace Length	cc p-value	*.529*	*-.056*	*-.189*	*.211*	*.090*	*-.480*	***0.874*** ***<0.001***	*-.469*	
Turtle Density	cc	*-.084*	.388	.236	*-.372*	*.125*	*-.481*	.074	.104	*.067*
Hylid Richness	cc p-value	***0.783*** ***.003***	**-0.598** **.040**	.380	***0.583*** ***.047***	*-.095*	*-.185*	.350	-.402	*.355*
Ranid Richness	cc	*.107*	-.008	-.258	*-.235*	*.264*	*.285*	-.216	.118	*.019*

Table 10 Continued

		Turtle Density	Hylid Richness	Ranid Richness	Bufonid Richness	Frog Richness	Basking Sites	Surface Area	Secchi Depth	Distance to Paved Road
Hylid Richness	cc p-value	-.288								
Ranid Richness	cc	.315	-.393							
Bufonid Richness	cc	*-.339*	*-.230*	*.158*						
Frog Richness	cc p-value	-.078	.444	.504	*.356*					
Basking Sites	cc p-value	*.451*	*-.049*	***0.649*** ***.023***	*.000*	*.474*				
Surface Area	cc p-value	-.020	.021	.150	*-.048*	.085	*.275*			
Secchi Depth	cc p-value	-.195	***.589*** ***.044***	-.436	*-.338*	-.306	*.107*	-.266		
Distance to Paved Road	cc p-value	-.092	.049	.032	*-.531*	-.176	*-.168*	-.376	.061	
Distance to Water Source	cc p-value	***.725*** ***.008***	-.267	.084	***-0.579*** ***.048***	-.524	*.168*	.013	.178	.199

Table 10 Continued

		Aquatic Vegetation	Clear Shoreline	Trees	Reeds	Herbaceous Shrubs	Woody Shrubs	Proportion Adult Turtles	Proportion Male Turtles	Carapace Length
Bufonid Richness	cc	*-.200*	*-.097*	*.000*	*.253*	*-.361*	*.302*	*-.388*	*.193*	*-.483*
Frog Richness	cc p-value	***0.618*** ***.032***	**-0.658** **.020**	.172	*.416*	*-.083*	*.230*	-.075	-.217	*.147*
Basking Sites	cc p-value	*.237*	*-.130*	*.408*	*-.060*	*.265*	*-.286*	*.218*	*.260*	*.237*
Surface Area	cc p-value	*.022*	.036	.133	*.303*	*-.392*	*-.306*	**0.606** **.037**	-.039	*.469*
Secchi Depth	cc p-value	***0.747*** ***.005***	-.206	-.210	*.358*	*.000*	*-.218*	***.874*** ***<0.001***	-.266	***0.797*** ***.002***
Distance to Paved Road	cc p-value	*-.044*	.193	-.251	***-0.661*** ***.019***	*.063*	*.218*	-.220	.149	*.091*
Distance to Water Source	cc p-value	*-.080*	.367	-.241	*-.441*	*.489*	*-.393*	.307	-.128	*.259*

REFERENCES

95-87, P. 1977. Public Law 95-87 - Surface Mining Control and Reclamation Act.*in* D. o. t. Interior, editor.

Angel, P. N. 1973. A soil analysis of the strip mine spoil bank at Fairfield, Texas. Stephen F. Austin State University, Nacogdoches.

Aresco, M. J. 2009. Environmental Correlates of the Abundances of Three Species of Freshwater Turtles in Lakes of Northern Florida. Copeia **2009**:545-555.

August, P. V. 1983. The Role of Habitat Complexity and Heterogeneity in Structuring Tropical Mammal Communities. Ecology **64**:1495-1507.

Bajema, R. A. and S. L. Lima. 2001. Landscape-Level Analyses of Henslow's Sparrow (Ammodramus Henslowii) Abundance in Reclaimed Coal Mine Grasslands. American Midland Naturalist **145**:288-298.

Barbeau, T. R. and L. J. G. Jr. 2007. Altered Ovarian Steroids in Xenopus laevis Exposed to Environmentally Relevant Concentrations of Nitrate. Journal of Herpetology **41**:590-596.

Bennett, D. H., J. W. Gibbons, and J. C. Franson. 1970. Terrestrial Activity in Aquatic Turtles. Ecology **51**:738-740.

Bishop, C. A., R. J. Brooks, J. H. Carey, P. Ng, R. J. Norstrom, and D. R. S. Lean. 1991. The Case for a Cause-Effect Linkage between Environmental Contamination and Development in Eggs of the Common Snapping Turtle (Chelydra-S Serpentina) from Ontario, Canada. Journal of Toxicology and Environmental Health **33**:521-547.

Blaustein, A. R., D. B. Wake, and W. P. Sousa. 1994. Amphibian Declines - Judging Stability, Persistence, and Susceptibility of Populations to Local and Global Extinctions. Conservation Biology **8**:60-71.

Bodie, J. R. and R. D. Semlitsch. 2000. Spatial and temporal use of floodplain habitats by lentic and lotic species of aquatic turtles. Oecologia **122**:138-146.

Bodie, J. R., R. D. Semlitsch, and R. B. Renken. 2000. Diversity and structure of turtle assemblages: associations with wetland characters across a floodplain landscape. Ecography **23**:444-456.

Bradley, P. J. 1987. The Effect of Strip Mining and Reclamation on Herpetofaunal Communities. University of Texas at Arlington, Arlington.

Brattstrom, B. H. 1963. A Preliminary Review of the Thermal Requirements of Amphibians. Ecology **44**:238-255.

Brown, S. C., K. Smith, and D. Batzer. 1997. Macroinvertebrate responses to wetland restoration in northern New York. Environmental Entomology **26**:1016-1024.

Burton, E. C., M. J. Gray, A. C. Schmutzer, and D. L. Miller. 2009. Differential Responses of Postmetamorphic Amphibians to Cattle Grazing in Wetlands. The Journal of Wildlife Management **73**:269-277.

Carrozzino, A. L. 2009. Evaluating wildlife response to vegetation restoration on reclaimed mine lands in southwestern Virginia. Virginia Polytechnic Institute and State University, Blacksburg.

Chapin, F. S., L. R. Walker, C. L. Fastie, and L. C. Sharman. 1994. Mechanisms of Primary Succession Following Deglaciation at Glacier Bay, Alaska. Ecological Monographs **64**:149-175.

Clements, F. E. 1916. Plant succession: an analysis of the development of vegetation. Carnegie Institute of Washington Publications, Washington, D.C.

Clements, F. E. 1936. Nature and Structure of the Climax. The Journal of Ecology **24**:252-284.

Cooper, W. S. 1923. The recent ecological history of Glacier Bay, Alaska. I. The interglacial forests of Glacier Bay. Ecology **4**:93-128.

Cowles, H. C. 1899. The ecological relations of the vegetation on the sand dunes of Lake Michigan. The Botanical Gazette **27**:95-117,167-202,281-308,361-191.

Crouch, W. B. and P. W. C. Paton. 2002. Assessing the Use of Call Surveys to Monitor Breeding Anurans in Rhode Island. Journal of Herpetology **36**:185-192.

De Capita, M. E. and T. A. Bookhout. 1975. Small mammal populations, vegetational cover, and hunting use of an Ohio strip-mined area. The Ohio Journal of Science **75**:305-313.

DeVault, T. L., P. E. Scott, R. A. Bajema, and S. L. Lima. 2002. Breeding Bird Communities of Reclaimed Coal-Mine Grasslands in the American Midwest (Comunidades de aves anidantes de los pastizales creados en minas de carbón reclamadas en el medio oeste americano). Journal of Field Ornithology **73**:268-275.

Dixon, J. R. 2000. Amphibians and Reptiles of Texas. 2nd edition. Texas A&M University Press, College Station.

Driscoll, D. A. 1997. Mobility and metapopulation structure of Geocrinia alba and Geocrinia vitellina, two endangered frog species from southwestern Australia. Australian Journal of Ecology **22**:185-195.

EIA. 2002. Annual Coal Report.*in* U. S. E. I. Administration, editor.

Ernst, C. H. and J. E. Lovich. 2009. Turtles of the United States and Canada. 2nd edition. The Johns Hopkins University Press, Baltimore.

Fahrig, L., J. H. Pedlar, S. E. Pope, P. D. Taylor, and J. F. Wegner. 1995. Effect of Road Traffic on Amphibian Density. Biological Conservation **73**:177-182.

Ferrell, S. C. 2006. Trachemys scripta scripta (yellow-bellied slider): fire ants and nest failure. Herpetological Review **37**:343.

Fontenot, B. E., R. Makowsky, and P. T. Chippindale. 2011. Nuclear-mitochondrial discordance and gene flow in a recent radiation of toads. Molecular Phylogenetics and Evolution **59**:66-80.

Fox, B. J., J. E. Taylor, and P. T. Thompson. 2003. Experimental Manipulation of Habitat Structure: A Retrogression of the Small Mammal Succession. Journal of Animal Ecology **72**:927-940.

Froese, A. D. 1978. Habitat Preferences of the Common Snapping Turtle, Chelydra s. serpentina (Reptilia, Testudines, Chelydridae). Journal of Herpetology **12**:53-58.

Fuller, T. E., K. L. Pope, D. T. Ashton, and H. H. Welsh. 2011. Linking the Distribution of an Invasive Amphibian (Rana catesbeiana) to Habitat Conditions in a Managed River System in Northern California. Restoration Ecology **19**:204-213.

Gaines, M. S. and L. R. McClenaghan, Jr. 1980. Dispersal in Small Mammals. Annual Review of Ecology and Systematics **11**:163-196.

Galatowitsch, S. M. and A. G. vanderValk. 1996. Characteristics of recently restored wetlands in the prairie pothole region. wetlands **16**:75-83.

Garton, J. S. and R. A. Brandon. 1975. Reproductive Ecology of the Green Treefrog, Hyla cinerea, in Southern Illinois (Anura: Hylidae). Herpetologica **31**:150-161.

Gibbons, J. W., J. L. Greene, and J. D. Congdon. 1983. Drought-Related Responses of Aquatic Turtle Populations. Journal of Herpetology **17**:242-246.

Gibbons, J. W. and J. E. Lovich. 1990. Sexual Dimorphism in Turtles with Emphasis on the Slider Turtle (Trachemys scripta). Herpetological Monographs **4**:1-29.

Gibbs, J. P. 1998. Distribution of woodland amphibians along a forest fragmentation gradient. Landscape Ecology **13**:263-268.

Gibbs, J. P. and W. G. Shriver. 2002. Estimating the effects of road mortality on turtle populations. Conservation Biology **16**:1647-1652.

Gleason, H. A. 1926. The individualistic concept of the plant association. Bulletin of the Torrey Botanical Club **53**:7-26.

Glorioso, B. M., A. J. Vaughn, and J. H. Waddle. 2010. The Aquatic Turtle Assemblage Inhabiting a Highly Altered Landscape in Southeast Missouri. Journal of Fish and Wildlife Management **1**:161-168.

Gooch, M. M., A. M. Heupel, S. J. Price, and M. E. Dorcas. 2006. The effects of survey protocol on detection probabilities and site occupancy estimates of summer breeding anurans. Applied Herpetology **3**:129-142.

Goodpaster, W. W. and D. F. Hoffmeister. 1952. Notes on the mammals of western Tennessee. Journal of Mammalogy **33**:362-371.

Gray, M. J., D. L. Miller, A. C. Schmutzer, and C. A. Baldwin. 2007. Frog virus 3 prevalence in tadpole populations inhabiting cattle-access and non-access wetlands in Tennessee, USA. Diseases of Aquatic Organisms **77**:97-103.

Greenberg, C. H. and G. W. Tanner. 2004. Breeding Pond Selection and Movement Patterns by Eastern Spadefoot Toads (Scaphiopus holbrookii) in Relation to Weather and Edaphic Conditions. Journal of Herpetology **38**:569-577.

Greenwood, P. J., P. H. Harvey, and C. M. Perrins. 1979. The Role of Dispersal in the Great Tit (Parus major): The Causes, Consequences and Heritability of Natal Dispersal. Journal of Animal Ecology **48**:123-142.

Gunzburger, M. S. and J. Travis. 2004. Evaluating predation pressure on green treefrog larvae across a habitat gradient. Oecologia **140**:422-429.

Harden, L., S. Price, and M. Dorcas. 2009. Terrestrial Activity and Habitat Selection of Eastern Mud Turtles (Kinosternon subrubrum) in a Fragmented Landscape: Implications for Habitat Management of Golf Courses and Other Suburban Environments. Copeia **2009**:78-84.

Heaney, L. R. 1984. Mammalian species richness on islands on the Sunda Shelf, Southeast Asia. Oecologia **61**:11-17.

Heinen, J. T. 1992. Comparisons of the Leaf Litter Herpetofauna in Abandoned Cacao Plantations and Primary Rain-Forest in Costa-Rica - Some Implications for Faunal Restoration. Biotropica **24**:431-439.

Hillman, S. S. 1980. Physiological Correlates of Differential Dehydration Tolerance in Anuran Amphibians. Copeia **1980**:125-129.

Holcomb, C. M. and W. S. Parker. 1979. Mirex Residues in Eggs and Livers of 2 Long-Lived Reptiles (Chrysemys-Scripta and Terrapene-Carolina) in Mississippi, 1970-1977. Bulletin of Environmental Contamination and Toxicology **23**:369-371.

Huey, R. B. 1991. Physiological Consequences of Habitat Selection. American Naturalist **137**:S91-S115.

Institute, W. C. 2009. The Coal Resource: A Comprehensive Overview of Coal.*in* W. C. Institute, editor.

Ireland, T. T., G. L. Wolters, and S. D. Schemnitz. 1994. Recolonization of Wildlife on a Coal Strip-Mine in Northwestern New Mexico. The Southwestern Naturalist **39**:53-57.

Jackson, L. J. 2003. Macrophyte-Dominated and Turbid States of Shallow Lakes: Evidence from Alberta Lakes. Ecosystems **6**:213-223.

Jacobson, E. R. 1909. De nieuwe fauna van Krakatau. Jversl. topogr. Dienst Ned.-Indie:192-197.

Janes, D. and W. H. N. Gutzke. 2002. Factors affecting retention time of turtle scutes in stomachs of American alligators, Alligator mississippiensis. American Midland Naturalist **148**:115-119.

Janzen, F. J., J. F. Tucker, and G. L. Paukstis. 2000. Experimental analysis of an early life-history stage: avian predation selects for larger body size of hatchling turtles. Journal of Evolutionary Biology **13**:947-954.

Jasper, C. 1992. Fish and Wildlife Resources Information, Big Brown CVI Revision (F.M. 2570 South), Freestone County, Texas. Espey, Huston & Associates, Inc.

Joyal, L. A., M. McCollough, and M. L. Hunter. 2001. Landscape ecology approaches to wetland species conservation: a case study of two turtle species in southern Maine. Conservation Biology **15**:1755-1762.

Kauffman, J. B. and W. C. Krueger. 1984. Livestock Impacts on Riparian Ecosystems and Streamside Management Implications... A Review. Journal of Range Management **37**:430-438.

Kirmer, A. and E.-G. Mahn. 2001. Spontaneous and initiated succession on unvegetated slopes in the abandoned lignite-mining area of Goitsche, Germany. Applied Vegetation Science **4**:19-27.

Knutson, M. G., W. B. Richardson, D. M. Reineke, B. R. Gray, J. R. Parmelee, and S. E. Weick. 2004. Agricultural ponds support amphibian populations. Ecological Applications **14**:669-684.

Lawrence, D. B. 1979. Primary versus secondary succession at Glacier Bay National Monument, S.E. Alaska. Pages 213-224 *in* Proceedings of the first conference on scientific research in the national parks, Nation park Serive Transactions and Proceedings, Series 5, Washington D.C., USA.

Lehtinen, R. M. and S. M. Galatowitsch. 2001. Colonization of Restored Wetlands by Amphibians in Minnesota. The American Midland Naturalist **145**:388-396.

Lehtinen, R. M., S. M. Galatowitsch, and J. R. Tester. 1999. Consequences of habitat loss and fragmentation for wetland amphibian assemblages. wetlands **19**:1-12.

Lindsay, S. D. and M. E. Dorcas. 2001. Effects of cattle on reproduction and morphology of pond-dwelling turtles in North Carolina. The Journal of the Elisha Mitchell Scientific Society **117**:249-257.

Luminant. 2010. Environmental Responsibilities Training.

Mann, R. M., R. V. Hyne, C. B. Choung, and S. P. Wilson. 2009. Amphibians and agricultural chemicals: Review of the risks in a complex environment. Environmental Pollution **157**:2903-2927.

Marchand, M. N. and J. A. Litvaitis. 2004. Effects of Habitat Features and Landscape Composition on the Population Structure of a Common Aquatic Turtle in a Region Undergoing Rapid Development. Conservation Biology **18**:758-767.

Marco, A., C. Quilchano, and A. R. Blaustein. 1999. Sensitivity to nitrate and nitrite in pond-breeding amphibians from the Pacific northwest, USA. Environmental Toxicology and Chemistry **18**:2836-2839.

Monroe, M. S. and G. Ritchison. 2005. Breeding Biology of Henslow's Sparrows on Reclaimed Coal Mine Grasslands in Kentucky / Biología reproductiva de Ammodramus henslowii en un hierbazal establecido en donde hubo una mina de carbón en Kentucky. Journal of Field Ornithology **76**:143-149.

Odum, E. P. 1969. The Strategy of Ecosystem Development. Science **164**:262-270.

Palis, J. G., M. J. Aresco, and S. Kilpatrick. 2006. Breeding Biology of a Florida Population of Ambystoma cingulatum (Flatwoods Salamander) during a Drought. Southeastern Naturalist **5**:1-8.

Piha, H., M. Luoto, M. Piha, and J. Meril. 2007. Anuran abundance and persistence in agricultural landscapes during a climatic extreme. Global Change Biology **13**:300-311.

Prach, K. and L. R. Walker. 2011. Four opportunities for studies of ecological succession. Trends in Ecology and Evolution **26**:119-123.

Rawlinson, P. A., R. A. Zann, S. Balen, and I. W. B. Thornton. 1992. Colonization of the Krakatau islands by vertebrates. GeoJournal **28**:225-231.

Rios-Lopez, N. and T. M. Aide. 2007. Herpetofaunal dynamics during secondary succession. Herpetologica **63**:35-50.

Rizkalla, C. and R. Swihart. 2006. Community structure and differential responses of aquatic turtles to agriculturally induced habitat fragmentation. Landscape Ecology **21**:1361-1375.

Roe, J. H., A. C. Brinton, and A. Georges. 2009. Temporal and spatial variation in landscape connectivity for a freshwater turtle in a temporally dynamic wetland system. Ecological Applications **19**:1288-1299.

Rose, F. L. and R. W. Manning. 1996. Notes on the biology of the slider, Trachemys scripta elegans (Reptilia: Emydidae), inhabiting man-made cattle ponds in West Texas. Texas Journal of Science **48**:191-206.

Saenz, D., L. A. Fitzgerald, K. A. Baum, R. N. Conner, and D. Adams. 2006. ABIOTIC CORRELATES OF ANURAN CALLING PHENOLOGY: THE IMPORTANCE OF RAIN, TEMPERATURE, AND SEASON. Herpetological Monographs **20**:64-82.

Schmutzer, A. C., M. J. Gray, E. C. Burton, and D. L. Miller. 2008. Impacts of cattle on amphibian larvae and the aquatic environment. Freshwater Biology **53**:2613-2625.

Schwartz, J. J., M. A. Bee, and S. D. Tanner. 2000. A Behavioral and Neurobiological Study of the Responses of Gray Treefrogs, Hyla versicolor to the Calls of a Predator, Rana catesbeiana. Herpetologica **56**:27-37.

Scott, P. E., T. L. DeVault, R. A. Bajema, and S. L. Lima. 2002. Grassland Vegetation and Bird Abundances on Reclaimed Midwestern Coal Mines. Wildlife Society Bulletin **30**:1006-1014.

Scott, P. E. and S. L. Lima. 2004. Exotic Grasslands on Reclaimed Midwestern Coal Mines: An Ornithological Perspective. Weed Technology **18**:1518-1521.

Scribner, K. T., S. J. Morreale, M. H. Smith, and J. W. Gibbons. 1995. Factors Contributing to Temporal and Age-Specific Genetic Variation in the Freshwater Turtle Trachemys scripta. Copeia **1995**:970-977.

Semlitsch, R. D. 1998. Biological delineation of terrestrial buffer zones for pond-breeding salamanders. Conservation Biology **12**:1113-1119.

Shulse, C., R. Semlitsch, K. Trauth, and A. Williams. 2010. Influences of Design and Landscape Placement Parameters on Amphibian Abundance in Constructed Wetlands. wetlands **30**:915-928.

Simberloff, D. S. and E. O. Wilson. 1969. Experimental Zoogeography of Islands: The Colonization of Empty Islands. Ecology **50**:278-296.

Simon, J. A., J. W. Snodgrass, R. E. Casey, and D. W. Sparling. 2009. Spatial correlates of amphibian use of constructed wetlands in an urban landscape. Landscape Ecology **24**:361-373.

Sinsch, U. 1990. Migration and Orientation in Anuran Amphibians. Ethology Ecology & Evolution **2**:65-79.

Sinsch, U. 1997. Postmetamorphic dispersal and recruitment of first breeders in a Bufo calamita metapopulation. Oecologia **112**:42-47.

Sly, G. R. 1976. Small Mammal Succession on Strip-mined Land in Vigo County, Indiana. American Midland Naturalist **95**:257-267.

Snider, A. T. and J. K. Bowler. 1992. Longevity of reptiles and amphibians in North American collections. Soc. Stud. Amphib. Rept. Herpetol. Circ. **21**:1-40.

Sokal, R. R. and F. J. Rohlf. 1995. Biometry the Principles and Practice of Statistics in Biological Research. 3rd edition. W.H. Freeman and Company, New York.

Sousa, W. P. 1984. The Role of Disturbance in Natural Communities. Annual Review of Ecology and Systematics **15**:353-391.

Stone, W. B., E. Kiviat, and S. A. Butkas. 1980. Toxicants in Snapping Turtles. New York Fish and Game Journal **27**:39-50.

Tidemann, C. R., D. J. Kitchener, R. A. Zann, and I. W. B. Thornton. 1990. Recolonization of the Krakatau Islands and Adjacent Areas of West Java, Indonesia, by Bats (Chiroptera) 1883-1986. Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences **328**:123-130.

Trimble, S. W. 1994. Erosional effects of cattle on streambanks in Tennessee, U.S.A. Earth Surface Processes and Landforms **19**:451-464.

Tucker, J. K., J. T. Lamer, C. R. Dolan, and E. A. Dustman. 2006. Chelonian species: Record carapace lengths for Illinois. Herpetological Review **37**:453-455.

Turesson, H. and C. Brönmark. 2007. Predator-Prey Encounter Rates in Freshwater Piscivores: Effects of Prey Density and Water Transparency. Oecologia **153**:281-290.

Vitt, L. J. and J. P. Caldwell. 2009. Herpetology. 3rd edition. Academic Press, Burlington.

Vogel, L. S. and J. H. K. Pechmann. 2010. Response of Fowler's Toad (Anaxyrus fowleri) to Competition and Hydroperiod in the Presence of the Invasive Coastal Plain Toad (Incilius nebulifer). Journal of Herpetology **44**:382-389.

Vos, C. C. and J. P. Chardon. 1998. Effects of habitat fragmentation and road density on the distribution pattern of the moor frog Rana arvalis. Journal of Applied Ecology **35**:44-56.

Wagner, W. L., W. C. Martin, and E. F. Aldon. 1978. Natural succession on strip-mined lands in northwestern New Mexico. Journal Name: Reclam. Rev.; (United States); Journal Volume: 1:2:Medium: X; Size: Pages: 67-73.

Walker, C. F. 1946. The amphibians of Ohio. Part 1. Frogs and toads. . Ohio State Museum of Science Bulletin **1**.

Walker, L. R. and R. d. Moral. 2009. Lessons from Primary Succession for Restoration of Severely Damaged Habitats. Applied Vegetation Science **12**:55-67.

Welsh, H. H. and L. M. Ollivier. 1998. Stream amphibians as indicators of ecosystem stress: A case study from California's redwoods. Ecological Applications **8**:1118-1132.

Whittaker, R. J., M. B. Bush, and K. Richards. 1989. Plant Recolonization and Vegetation Succession on the Krakatau Islands, Indonesia. Ecological Monographs **59**:59-123.

Willson, M. F., B. L. Rice, and M. Westoby. 1990. Seed Dispersal Spectra: A Comparison of Temperate Plant Communities. Journal of Vegetation Science **1**:547-562.

Worley, I. A. 1973. The "black crust" phenomenon in upper Glacier Bay, Alaska. Northwest Science **47**:20-29.

Wray, T., II, K. A. Strait, and R. C. Whitmore. 1982. Reproductive Success of Grassland Sparrows on a Reclaimed Surface Mine in West Virginia. The Auk **99**:157-164.

Yeager, L. E. 1942. Coal-Stripped Land as a Mammal Habitat, With Special Reference to Fur Animals. American Midland Naturalist **27**:613-635.

Yukawa, J., T. Abe, T. Iwamoto, and Y. Seiki. 1984. The fauna of the Krakatau, Peucang and Panaitan Islands. . Researches on the ecological succession and the formation process of volcanic ash on the Krakatau Islands:91-114.

BIOGRAPHICAL INFORMATION

Jayme Walton was born in Anchorage, Alaska on March 16, 1983, and became a Texan at the age of 5. She received her Bachelor of Science degree in Zoology from the University of Oklahoma in 2005 and her Master of Science degree in Biology from the University of Texas at Arlington in 2012. Between those years she worked at the Fort Worth Zoo in the Nutrition and Education departments. In 2010, Jayme was the recipient of an Environmental Research Program Fellowship from Luminant which funded her Master's research. She also received the Annual Celebration of Excellence by Students Graduate Sustainability Award for her Master's research in 2012. Jayme intends to pursue a career that will in some way help preserve and protect the natural areas of our country.