HOW AQUATIC FAUNA RESPONDED TO LARGE SCALE MANAGEMENT ON LAKE TOHOPEKALIGA, FLORIDA

By

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To my parents, whose love, encouragement and support got me this far; and to my husband, who was supportive and kept me sane throughout the process.
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HOW AQUATIC FAUNA RESPONDED TO LARGE SCALE MANAGEMENT ON LAKE TOHOPEKALIGA, FLORIDA

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Many Florida lakes have undergone dramatic changes in hydrology that have restricted
water level fluctuation, reduced floodplain area and triggered accumulation of dense
monospecific stands of vegetation and muck (tussocks) in narrowed littoral zones. Dense plant
communities and the conditions they impart on the environment are considered unsuitable habitat
for aquatic fauna, causing lake managers to employ draw downs and sediment removal as a
means of offsetting these conditions. Multiple studies document the benefit conveyed to
economically important sportfish, primarily largemouth bass (Micropterus salmoides).
However, most fish studies fail to meet two major criteria, including ability to actually sample
fishes within dense tussock formations and the failure to consider detection probabilities.
Surprisingly scant data exists documenting the impacts of lake management on other aquatic
inhabitants. Our study sampled a variety of littoral fauna within dense pickerelweed (Pontederia
cordata) tussocks, including sport and forage fishes, reptiles, amphibians and apple snails prior
to, and after a major lake management project on Lake Tohopekaliga, FL. Unfortunately an
active hurricane season followed on the heels of management actions, complicating the study
somewhat. However, we still gained valuable insight into faunal response to extreme habitat
management.
Results for fish were mixed. Of the thirteen species captured enough to obtain statistical estimates, five including bluegill and largemouth bass increased occupancy after the management. One sportfish, the redear sunfish barely responded. The other seven species including two sportfish, the warmouth and spotted sunfish, as well as some forage fish species declined in occupancy. All herpetofaunal species declined dramatically after the management. The large-scale extent of this project probably triggered migrations, survival strategies and caused direct mortality for many caught in the process of the dry down and sediment removal. Some species commonly associated with vegetated and murky environments began to recover by 2006-2007, indicating that pre-management conditions were returning. As vegetation continues to recover across the lake, animals will continue to respond. This study documents the immediate responses to management, which ought to be supplemented with further monitoring to understand long term impacts of lake management.
CHAPTER 1
INTRODUCTION

Brief History of Florida’s Wetland Landscape

The state of Florida is literally awash in water with over 7700 lakes dotting the landscape, the enormous swath of Everglades swamp, meandering rivers, floodplains, springheads, and numerous urban retention ponds. Unlike many lakes around the world, most of Florida lakes are very shallow, with large surface area to volume ratios. Even the largest lake, Lake Okeechobee only has a mean depth of less than 3m (Havens et. al 1996). Most naturally formed lakes are the result of either solution processes or relict sea-bottom depressions that filled with freshwater as the oceans receded (Edmiston and Myers 1983).

Before human settlement and manipulation of this vast wetland system, much of Florida was essentially a large floodplain, receiving and storing rainfall by flooding the extensive littoral zones of lakes and swelling and spilling over river banks. The Kissimmee Chain of Lakes drained through the Kissimmee River which then emptied into Lake Okeechobee. During high water events, the giant lake’s southern rim would spill over into the vast Everglades, a wetland ecosystem comprised of sawgrass ridges, sloughs and tree islands. This giant freshwater wetland expanded to the east and west coasts, transitioning to brackish marsh and mangrove communities that greeted the ocean. This vast expanse of the greater Everglades watershed beginning in central Florida (Figure 1-1), would ebb and flow depending on weather patterns, shaping the Floridian landscape and its native flora and fauna.

These seasonally inundated and fluctuating wetlands provide habitat for many animals. Local flora and fauna are adapted to this dynamic system, and comprise a diverse food web. Aquatic plants support epiphytic algae which feeds a macroinvertebrate community that in turn nourishes small vertebrates, consumed by larger animals such as bass, egrets, otters and
alligators to name a few. The structural complexity provided by aquatic vegetation offers protection from predators for many animals including young fish species and herpetofauna (Savino and Stein 1982, Rozas and Odum 1988). Wet seasons tend to trigger the onset of breeding for many animals, and the degree of water fluctuation influences nesting success of many aquatic and semi-aquatic vertebrates from turtles to snail kites (*Rostrhamus sociabilis*). The dry season will send some animals into aestivation or force migration, until suitable conditions return.

In addition to providing habitat to many animals, wetlands provide valuable ecosystem services and functions including storage of excess runoff, flood and erosion control, groundwater recharge and discharge and improved water quality through filtration processes (SFWMD 2003). As water spills into floodplains moving across the landscape it carries with it an assortment of sediments that disperse and settle out. This is accomplished through many mechanisms including stabilizing and trapping sediments thereby reducing turbidity (Schriver et al. 1995; Vermaat et al. 2000), uptake of nutrients through plant tissues and their oxidation of sediments that reduce phosphorous fluxes in the water column (Wigand et al. 1997).

The natural functions of wetlands have been degraded over the years, due mostly to human settlement. Flooded shorelines impacted agriculture and other human uses, often causing great catastrophe and loss of lives, particularly after large tropical storm events. In response, many Florida lakes were dyked, dredged, and lowered, canals were dug and existing outflow channels were straightened (Schiffer 1998). The alteration of this landscape for the purpose of flood control and use of adjacent floodplain began in the 1800’s and continues today. The result of less flooding and stabilized water conditions for humans has resulted in a slew of unintended negative consequences. Increased eutrophy and lake succession, decreased water quality,
decreased plant diversity, dense vegetation growth, loss of wildlife habitat, reduced recreational use and poor aesthetic quality are common symptoms of highly regulated lakes.

Some milestone projects are underway to reverse years of wetland habitat degradation such as the Kissimmee River Restoration Project and the Comprehensive Everglades Restoration Plan. However, many Florida lakes face continued pressure as the human population continues to increase and demands for land and water increase. Lake management has evolved in response to the problems afflicting lakes, but also to ensure continued protection against catastrophic flooding and destruction of human property and lives. Because of this later concern, management actions typically cannot fix the source problem, but can be effective at delaying continued degradation.

**Florida Fisheries and Lake Management**

Restoring historical flow to Florida’s floodplains is rarely a viable management option. Although natural water variation has been considerably stifled (Figure 1-2), lake managers still allow for some variation throughout the year. A standard management response to address the accelerated lake succession and degradation is routine draw downs sometimes accompanied with plant and mud removal. These “lake management” projects are regularly used on many Florida lakes including Lake Tohopekaliga (hereafter referred to as Lake Toho) to offset negative (or “degraded”) conditions. Draw-downs work by consolidating and oxidizing organic sediments, stimulating germination of native plants and increasing macroinvertebrate and forage fish populations (Moyer et al. 1995). Coupled with sediment removal, the effects of draw-downs are more powerful and long-lasting.

Dense stands of vegetation including off-shore floating mats and accumulated organic sediments are undesirable consequences of restricted flow and cultural eutrophication. These conditions have been documented to alter the abiotic properties of water, resulting in poor habitat
quality for aquatic organisms (Crowder and Cooper 1982, Savino and Stein 1982, Sculthorpe 1985, Frodge et al. 1990, Moyer et al. 1995, Miranda et al. 2000, Miranda and Hodges 2000, Allen and Tugend 2002). Such conditions are thought to be detrimental to the thriving freshwater fishing industry in Florida, which brings in an estimated two billion dollars to the state each year. Thus improving sportfish populations and maintaining boater access is a major objective of many management practices in addition to offsetting plant and muck accumulation (Moyer et al. 1995, Olson et al. 1998, Tugend 2001, Allen and Tugend 2002, Allen et al. 2003).

For years, fisheries science and management have subscribed to the notion that dense vegetation is unsupportive of healthy and abundant sport fish populations. They attempt to offset conditions that have resulted from years of impoundment and cultural eutrophication. While most managers can probably agree that dense vegetation is undesirable, they also recognize the importance of vegetation in supporting fish populations as well as other aquatic fauna.

Vegetation provides structural complexity, substrate for macro and microinvertebrates, egg attachment for various species, protection from predation and weather and an increased prey base. Littoral food webs are complex, containing more trophic guilds, species and links than pelagic zones (Havens et al. 1996). Although pelagic food webs can also be complicated, they are usually simpler and support lower diversity. However, when plant densities reach exceedingly high levels in littoral zones, can the animals that rely on vegetated habitats survive the ensuing conditions? Determining how much plant biomass is detrimental versus beneficial is no easy task.

This study presented a unique opportunity to examine the habitability of dense pickerelweed (*Pontederia cordata*) to reptiles, amphibians, fish and apple snails inhabiting a eutrophic, degraded lake. An unprecedented management project was planned for Lake Toho,
involving dry down and removal of an estimated 6.9 million m$^3$ or 1,351 ha of the organic plant and muck accumulated in most of the littoral zone. What follows is a description of the lake, a review of past research and the objectives of this study.

**Lake Tohopekaliga**

Lake Toho is a large shallow lake, about 9800 hectares with an average depth of 2.1 meters. It is a naturally formed lake located in Kissimmee, Florida (Osceola County), a highly developed and populated area. Like many Florida wetlands, it has suffered the consequences of restricted water flow, cultural eutrophication and intense human use.

Between 1942 and 1964, the lake fluctuated with a range of about 3.2 meters (USGS, unpublished data). After a lock and spillway structure was completed, the time between 1964 and 1970 experienced a range of only 1.44 meters (Wegener and Williams 1974). Further regulation reduced the range even further to 0.91 meters, with a one in three year drop of an extra 0.15 meters (Hoyer et al. 2008). Thus the historical recorded fluctuation dropped from about 3.2 meters to 0.91 meters (Figure 1-2).

Beginning in the 1950’s, the first municipal wastewater discharge entered the lake (Williams 2001), deteriorating water quality and aquatic habitat. Phosphorous and nitrogen loading was enormous, causing managers to step in to address the problem. By 1987 the wastewater discharge was almost completely eliminated and there has been substantial improvement since. However the damage done during this time period was exacerbated by water level restrictions that led to algal production and accelerated lake succession (Hoyer et al. 2008).

Draw downs and occasional muck removal became standard practice, with the first draw down on Lake Toho occurring in 1971, and another in 1979. In 1987 after the wastewater discharge was nearly eliminated, managers drew the lake down again, this time accompanying it
with mechanical removal of an estimated 172,000 m³ vegetation and muck. Within two years however, there was almost complete recovery of *Pontederia cordata*, the vegetation targeted for removal, though several grass species increased in frequency as well (Moyer et al. 1989). An unprecedented initiative in 2004 to draw down and remove 6.9 million m³ of sediment had three main objectives: offset lake succession, improve lake access and aesthetics, and restore fish and wildlife habitat (Hoyer et al. 2008).

The lake was lowered from 16.8 m to 14.9 m NGVD exposing and drying the entire emergent littoral zone, and permitting access and use of heavy equipment to remove materials. Twenty nine in-lake spoil islands were created, in addition to material placement upland (FFWCC 2004). Upon completion, water stages returned to normal levels during late summer/early fall of 2004.

Need for New Sampling and Statistical Techniques

Degraded habitats are continually targeted for removal, yet there has been inadequate evidence for its unsuitability to any guild of aquatic vertebrate other than sportfish. Further, the majority of fish studies were unsuccessful at sampling in dense vegetation, rendering dubious results. Simply put, traditional sampling techniques perform poorly in dense vegetation. The same might be said for herpetofaunal sampling methods. While some of these methods might function well under different circumstances, none are particularly useful or practical for sampling in dense aquatic vegetation.

Fish sampling methods typically include electrofishing, rotenone/blocknet and Wegener rings (Moyer et al. 1995, Hoyer and Canfield 1996, Tugend 2001, Allen and Tugend 2002, Allen et al. 2003, Bonvechio and Bonvechio 2006). These methods only perform well in sparsely vegetated and open water habitats due to limitations of equipment and boat access. Many studies attest to have employed these methods in degraded habitats when they are probably sampling at
the edges of these inaccessible areas. Moyer et al. (1995) admitted this, and also that far fewer control sites were sampled because of dense vegetation. Allen and Tugend (2002) found it “nearly impossible” to sample with blocknets in the dense vegetation and thus used only rotenone, whilst using blocknets in enhanced sites.

Bonvechio and Bonvechio (2006) found discrepancies between angler catch rates and electrofishing, suggesting that either one or both methods were poor indicators of largemouth bass abundance. They noted that environmental factors such as hydrilla (Hydrilla verticillata) coverage, trophic state and water quality can limit the ability of electrofishing methods to estimate abundance. Furthermore, large adult fish are more readily captured during electrofishing (Rey 1996, Bayley and Austen 2002) which neglects smaller fish representative littoral zone inhabitants.

Creel surveys are another method used in fisheries research, used to attain fishing pressure estimates from angler counts. Various methods are available, and the estimates can be very inconsistent, affecting “virtually every aspect of creel survey results” (Soupir et al. 2006). For example, aerial estimation methods have been referred to as “fair-weather fishing estimates” because of their reliability on good flying weather (Soupir et al. 2006). Additionally, shoreline anglers can be difficult to detect and the high costs that typically limit the number of surveys, reduces the accuracy and precision of the study (Soupir et al. 2006). Importantly, angler success might be the result of factors other than actual fish abundance. Changes in fish behavior, fishing technique, effort, access, environment, and reliability of angler information can influence these surveys (Bonvechio and Bonvechio 2006). While creel surveys provide useful information about harvest rates, fishing pressure and angler effort, they should not be used to infer habitat or
population changes. These types of surveys should be supplemental to research that investigates
the effectiveness of management on habitat quality and its effects on species of interest.

Trapping has been used intermittently to sample littoral fishes (Bendell and McNichol
1987, Conrow et al. 1990, Chick and McIvor 1994, Jackson and Harvey 1997, Whittier and
Hughes 1998, MacRae and Jackson 2006, Bunch et al. 2008). Various trapping methods target
an array of fish sizes and species and have been successful in vegetated habitats. However,
Conrow et al. (1990) struggled with vegetation entanglement, compromising their method
somewhat. Bunch et al. (2008) used active trapping methods which were successful in various
types of dense vegetation. Other than this, there has been little use of traps probably because of
the time investment required to use them. Studies of lake management effects in Florida
continue to rely primarily on the traditional fishing methods referred to above.

If a densely vegetated habitat is not accessible prior to management but then becomes
available afterwards, any perceived changes in habitat quality or abundance of fish could be
misleading. What is likely provided is “a description of the surveyors’ ability to find the species
on the landscape, not where the species is on the landscape” (MacKenzie 2005). Although
improved boater access is an important and commonly stated goal for management projects,
whether or not it is improving habitat for target species is questionable based on these methods.

Studies have shown that aquatic macrophytes are commonly used by larval and juvenile
fishes as well as small forage fishes (Shireman et al. 1981, Killgore et al. 1989, Chick and
2008). Following the removal of this habitat, some studies document higher juvenile catch rates
and abundances (Moyer et al. 1995, Allen and Tugend 2002, Allen et al. 2003). This might be a
result of the previously inaccessible fishes now being more exposed and readily captured in open water where the traditional fisheries techniques excel. Allen et al. (2003) explain their overestimation of age-1 largemouth bass abundance as a likely result of improved catchability after the management.

Allen et al. (2003) also noted no difference between pre and post management catch of harvestable-sized fish (≥356mm TL) sampled via electrofishing and creel surveys. Since adult bass do not occupy littoral vegetation to the extent that juveniles do, their undetected change in abundance is not surprising. Hoyer and Canfield (1996) confirmed that in some small Florida lakes, adult largemouth bass have been known to exist without any macrophyte coverage. They also recognized positive correlations between juvenile bass abundance and aquatic macrophyte abundance, but no such relationship with adult bass. They acknowledge that large lakes such as Toho, Kissimmee and Okeechobee have significantly less shoreline to surface area ratios and might require macrophytes as refuge for young bass, and call for more research on larger lake systems (>300ha).

All of the studies focusing on fish response to management have failed to incorporate detection probability into their analyses. This weakness is also common in many herp studies (Mazerolle et al. 2007). Such approaches assume perfect detection (100% chance of detecting the animal in each sample), which is seldom realistic. Time of day, season, temperature, water level, vegetation cover, observer technique, trap type and boat noise are the sorts of things that can influence detection probabilities and are affecting how scientists and managers interpret faunal responses to management. For example, a simple observed change in count might be the result of random variations or changes in detection (MacKenzie et al. 2002), rather than an actual change in abundance, density or occupancy. Additionally, non-detection of a species does not
necessarily imply absence, but also the possibility that it simply went unnoticed. Statisticians have devised sophisticated approaches to this problem (Mazerolle et al. 2007), which are explored in this study. Although counts uncorrected for detection (ad hoc) perform poorly, they continue to be used extensively (MacKenzie 2005, Mazerolle et al. 2007). In fact, a supposedly famous fisheries saying is, “studying fish is like studying trees, except that they move around and are invisible” (Shepherd in Hillborn 2002).

**Study Objectives**

Because of the paucity of research on faunal responses other than targeted sport fishes, and lack of effective management analysis (OPPAGA 2001, 2003), the FWC proposed a study to document how herpetofauna and fishes would respond to the “2003 Lake Tohopekaliga Management Project”. This unprecedented large-scale project provided a unique opportunity to study lake-wide impacts of the management project with both before and after lake-wide comparisons as well as smaller scale resolution control and treatment habitats. Armed with an appropriate sampling and statistical analysis approach, we were able to study changes in occupancy of littoral species after the management project. This study was intended to answer the following research questions and hypotheses.

1) What faunal species occupied the dense *P. cordata* littoral habitat of Lake Tohopekaliga prior to management?

2) Does the estimated proportion of area occupied for each species change after management?  
   **Hypothesis 1**: species that have high occupancy estimates pre-management will decrease after management operations.  
   **Hypothesis 2**: Mud-burrowing species such as amphiumas and sirens will decrease in occupancy post-management.  
   **Hypothesis 3**: Largemouth bass, bluegill and Seminole killifish will increase occupancy post-management.

3) Does the estimated proportion of area occupied for each species differ between control and treatment habitats?  
   **Hypothesis 4**: species with high occupancy pre-management will have the highest occupancy in control habitats.
**Hypothesis 5:** largemouth bass, bluegill and Seminole killifish will have the highest occupancy in treated habitats.

4) What environmental factors influence detection probabilities and other parameters, particularly occupancy for each species?  
**Hypothesis 6:** lake stage and temperature are strong determinants of detection probability; habitat is a strong determinant of occupancy.

**Overview of Terms Used**

**Habitat**

In this study we identify three habitat types after the management project: control, treatment and all-lake. The control and treatments that comprised the small scale resolution focus studies, allowed for post-management treatment and control comparisons. Each study area consisted of 1600m of shoreline, with two control and two treatment blocks each 400m. One study area was located in Goblet’s Cove (Area 3), another along South Steer Beach (Area 2) and the third south of Brown’s Point (Area 1) refer to Figure 2-2. The third habitat type identified is all-lake, which can simply be thought of as an alternative treatment type. These areas were located where extensive shoreline scraping had occurred, primarily along the southeastern shoreline and entire western shoreline up to Brown’s Point. This treatment type might differ from the study area treatments because of lack of proximity to vegetation, and large tracts of exposed open water habitat which might influence its habitability. Each of the three habitats control, treatment and all-lake were intentionally surveyed in order to detect any differences among them.

**Location**

Five distinct locations around the perimeter of the lake were identified (Figure 2-2). These are study areas 1, 2, 3, all-lake east, and all-lake west. Lake Toho is large enough that differences might exist based on location due to various factors such as fetch, natural sedimentation and human disturbances that might influence our statistical parameter estimations.
Statistical Parameters and Covariates

Parameters for this study are occupancy, detection probability and colonization. Environmental covariates that may influence these parameters are habitat, location, lake stage, temperature, and season. The software program PRESENCE (Hines J.E. 2006) was used to calculate these estimates.

Patch occupancy analysis can be thought of as proportion of area occupied (PAO). This estimate of PAO obtained from repeated sampling, can be used to infer the extent to which an animal occupies the system in general. For example, if a particular species obtains an occupancy estimate of 0.40, this means that the animal occupies approximately 40% of the area- the PAO is 40%. A colonization value of 0.30 means that an unoccupied site has a 30% chance of becoming colonized. A detection probability of 0.50 means that if an animal is present in the habitat, there is a 50% chance of detecting it.

In occupancy analyses, the term “season” does not imply an actual season (breeding season, winter, summer, dry, wet etc.). Instead, it is defined as some period of time in which the observer considers an area closed to localized extinctions and colonizations of a species. In this study two consecutive sample occasions that share similar lake stage and temperature were considered closed seasons. The reason for selecting these discrete time periods is explained later in the methods section under “Assumption Violations”.

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Figure 1-1: Map of Lake Tohopekaliga within greater Everglades watershed (source: Brush 2006)
Figure 1-2: Daily mean lake stage in feet (NGVD) from January 1942 until June 2008. The dotted red lines indicate managed drawdowns. Impoundment occurred in 1964 and lake stage was regulated between 52-55 ft NGVD (Source: Brush et al. 2008)
CHAPTER 2
METHODS AND STATISTICAL ANALYSES

Trapping

Traditional methods of sampling fish and herpetofauna would have been ineffective and restrictive in the observed species. Densely vegetated littoral zones require a different technique capable of sampling a variety of littoral fauna. Some researchers have explored funnel traps (Darby et al. 2001, Casaza 2000, Sorensen 2003), leading to a pilot study by Muench (2004) who tested a few trap designs. Muench discovered that 1.3cm mesh size modified crayfish traps and floating minnow traps worked well at capturing herpetofauna, and a variety of fish species in dense *P. cordata*.

Traps retained mostly small adult species and juveniles of larger fish. For example, small forage fish species included blue-spotted sunfish (*Enneacanthus gloriosus*), Seminole killifish (*Fundulus seminolis*) and sailfin mollies (*Poecilia latipinna*). Juvenile fish included many sportfishes such as bluegill (*Lepomis macrochirus*), largemouth bass (*Micropterus salmoides*), redear sunfish (*Lepomis microlophus*), spotted sunfish (*Lepomis punctatus*), and warmouth (*Lepomis gulosus*). Table B-1 lists all species captured.

Herpetofaunal species were mostly adults, as young were able to escape through the mesh. Stinkpots (*Sternotherus odoratus*) and striped mud turtles (*Kinosternon baurii*) are very small adults that were successfully captured. Large adults such as softshell turtles (*Apalone ferox*), snapping turtles (*Chelydra serpentina osceola*) and peninsula cooters (*Pseudemys floridana peninsularis*) were excluded from the small trap size and only a few young were ever encountered. Despite the ability to retain very small or very large individuals, overall the traps captured a variety of fauna representative of the littoral zone. For more details on the trap selection process see Muench (2004).
Trap Descriptions

Both trap types were made from green vinyl-coated hardware cloth. Crayfish traps were about 80cm tall including an extension ‘chimney’ which allowed additional space above water for breathing of trapped animals (Figure 2-1). Three entry funnels at the base were each approximately 6cm in diameter. The tops of the traps were fitted with lids made from similar hardware cloth and small bungee cords. Rectangular minnow traps measured 60 x 25 x 18cm with an entry funnel at opposite ends approximately 9cm wide and 6cm tall. Minnow traps were fitted with floating devices to keep the funnels about even with the water surface (Figure 2-1). Prior to management styrofoam pool toys (“wacky noodles”) were used, but these did not hold up in the years following management and were switched to industrial marine floats from Sterling Net and Twine. Two float sizes per trap were used (#SB10–4 x 31/2in and SB11-4 x 6in). They were affixed through their central opening with zip-ties to the wire mesh. These floats resisted sun, desiccation, water-logging and attack by vultures (which regularly destroyed the pool toys when they were more visible after the management). Traps were not baited, although trapped animals may have encouraged others to enter.

Sampling Methods

The following methodology was developed and implemented by Muench (2004) and later adapted and modified for the entire study. Sampling was conducted around the periphery of the southern two-thirds of the lake, in order to understand occupancy at the lake-wide scale (Figure 2-2). The northern third was not scraped, thus it was not sampled. Eighteen sites were randomly selected and at each, a transect was set-up comprising 3 trap points about 10m apart, perpendicular to the shoreline. One crayfish and one minnow trap were attached to a pole. The crayfish trap rested on the bottom, with the floating minnow trap beside it (Figure 2-1), both partially above water to permit surfacing of trapped animals. Where dense tussocks existed,
traps were punched through and rested among the organic sediment and vegetation. In 2002, the traps were placed in the most lakeward portion of the pickerelweed zone when possible. This was approximately 0.6-0.9m deep, with the lake stage holding at 13.8m NGVD. Trapping proceeded throughout the entire year, pending suitable water levels (greater than 16m NGVD).

Traps remained in place and were checked once per week throughout the year. Efforts were made to space sampling seven days apart but this was not always logistically possible. Traps were approached by airboat and observers would walk to the three trap points, retrieving the animals. All animals were identified and weighed with only herpetofauna receiving length measurements. Herpetofauna were weighed individually, while fish species were grouped and weighed together. All animals were then released on site.

Trapping began on January 24, 2002. From May 2-June 12 traps were removed to due low water levels that stranded traps on dry ground. Trapping resumed entirely by July 24 (traps were replaced slowly as water and time permitted). By December 3, traps were again removed to due low water levels associated with the attempted 2002 drawdown.

**Post-Management Field Methodology Adjustments**

After the management project was completed, trapping was continued in a similar manner. Due to differences in habitat structure, some modifications had to be made in order for trapping to function properly. First, exact 2002 locations were not possible to re-sample due to deeper water from scraping and associated loss of organic sediment/vegetated bottom. Transects were moved shoreward from original GPS locations to where they were comfortably above water. The chimneys of the crayfish traps were extended even further to approximately 125cm to allow for more fluctuation in lake stages without swamping the trap.

Wind and waves became a serious issue, causing loss and damage of traps. The flimsy 1 inch PVC poles originally used were replaced with sturdier 2 inch poles. Furthermore, these
poles had to be pounded into the lake substrate, as there was no longer vegetation to support either the traps or poles. Coated electrical wire was used to affix traps to the poles, as the original zip tie method only resulted in snapped ties and lost traps. Every effort was made to make sure transects were properly functioning, but environmental conditions often resulted in missed samples due to trap disrepair. These logistical issues as well as a change in technicians, prevented trapping from fully resuming until May 3, 2005. It took a few months of revisions to find an adequate method and familiarize staff. Unfortunately, Hurricane Wilma struck the Kissimmee area in late October, which destroyed and displaced most traps. Thus from October 25, 2005 through to the end of November, trapping was sparse and intermittent as time and resources slowly allowed replacement of all traps.

Post-management trapping also included 12 additional transects coinciding with the newly created study areas. One transect per control and treatment block was randomly selected and transects were established in the same manner as 2002. There were two controls and two treatments per study area, thus 4 blocks in 3 study areas provided 12 additional transects. This brought the total number of transects to 30 (90 trap points and 180 traps). After 7 months of checking these 30 transects, trapping effort was reduced by eliminating the original 18 all-lake transects, in order to focus on control and treatment habitats. Beginning December 1, 2005 only the 12 study areas were sampled. This method continued until October 17, 2006.

Up to this point, all transects were in open water, even control plots presumably due to 2004-2005 hurricane damage. Vegetation was recovering shoreward, but the traps were not sampling within this community. Thus an adjustment to the protocol was implemented, with longer transects spaced out according to water depth. The first trap point began close to shore at 15cm deep, then 25cm, 35cm, 45cm, 55cm (at 16.20m NGVD). This allowed sampling in the
recovering littoral zone and out into the deeper reaches as well. With this protocol we were better reflecting the recovering littoral zone faunal communities. The 12 study area transects were used, in addition to 6 all-lake transects randomly selected from the original 18 (excluding locations that fell within proximity of a study area, in order to adequately sample expansive scraped shoreline habitat; Figure 2-2). This provided even sampling of control, treatment and all-lake areas, with 6 transects per habitat type. Trapping resumed in the same manner as in previous years with no other revisions until October 26, 2007.

**Hurricane Complications**

The 2005 hurricane season was very active in the state of Florida. Shortly after management had been completed and water was steadily returning to the lake, three major hurricanes swept through. Hurricanes Charley, Frances and Jeanne all struck within 40 miles of Lake Toho. Not surprisingly, the copious rainfall resulted in a rapid and dramatic rise in lake level. Water levels were at their highest since the 1960’s, approximately 2 feet above maximum regulated levels (Brush et al. 2008). Basically, the water rose over 8 feet in a 4 month period. The timing couldn’t have been worse, following so closely behind management activities.

Based on hurricane trajectories, Lake Toho’s geography and direct observation of the aftermath, it was surmised that South Steer Beach and Goblet’s Cove were somewhat sheltered from battering wind and waves, while the southern and south western parts of the lake were highly impacted (Brush et al. 2008). This is reflected by later vegetation samples showing slower recovery than might be expected, particularly in the south part of the lake (Brush et al. 2008). Unfortunately the hurricanes rendered control and treatment sites to look very similar, especially in the southern areas.

It is difficult to tease out the effects of the hurricanes versus the treatment and to compare control and treatment especially in the immediate time period after both management and
hurricanes. Possible confounding hurricane effects are noted throughout this thesis where appropriate.

Data Analyses

The statistical modeling program PRESENCE (Hines, J.E. 2006) was used to obtain patch occupancy estimates. If an animal was detected, a 1 was recorded, a 0 if it was not, and a “–“if the sample was missed. Missed occasions do not affect the data, they simply contribute nothing. Over time, a detection history is obtained comprising of a series of 1s, 0s and dashes for each transect. For example, a particular transect might have a warmouth detection history of 10 00 01 which indicates two consecutive sample occasions that comprise a season for a total of three seasons. Each species at each transect was recorded in this way, and the detection histories were entered into Microsoft Excel and copied directly into PRESENCE software.

Covariates are variables thought to potentially influence detection probabilities, occupancy, colonization and/or extinction rates. Categorical site covariates were location on lake (Figure 2-2) and habitat (post-management only: treatment, control, and all-lake). Continuous sample covariates (environmental variables that change for each sample occasion) were average air temperature, lake stage and season. Lake stage for each sample occasion was obtained from the South Florida Water management District’s DBHYDRO browser website (http://my.sfwmd.gov/dbhydrosql/show_dbkey_info.main_menu). The mean daily average lake level at headwater station S61_H was used. This is the water control structure in the south part of Lake Toho leading to Lake Cypress via the South Port Canal). Temperature was obtained from http://www7.ncdc.noaa.gov/CDO/cdodateoutmod.cmd. Average daily temperatures from the day of the previous sample occasion until the day before the current sample occasion were averaged.
These covariates were standardized so that each value was between 0-1, which performs more efficiently in PRESENCE, compared to entering the actual number. For example, 26 degrees Celsius was standardized to 0.26 instead. The categorical covariates were coded with ones and zeroes so that each had a unique code.

An example of a model name might be psi (location), gam (.), p (lake stage). What this means is occupancy (psi) differs by location on the lake, colonization (gam) is constant across all sites at all times, and detection probability (p) changes with lake stage. All model outputs are reported in Appendix A and B.

The second model parameterization type in PRESENCE labeled “seasonal occupancy and colonization, detection” was selected as the standard model to run for all species. This way any seasonal occupancy differences could be estimated.

**Assumption Violations**

Occupancy analyses entail a set of assumptions that should be addressed by study design. One major assumption is that occupancy status at each site does not change during the season, in other words the sites are “closed” to changes in occupancy (MacKenzie et al. 2006). This does not refer to individuals, but to a species. Unfortunately the original design does not accommodate this assumption, as animals are trapped year round, week after week. Surely in this time period with fluctuating water levels, temperature and vegetation changes, species are moving into and out of the sample areas.

In order to satisfy this assumption, two subsequent sample occasions that remained relatively equal in lake stage and temperature were grouped as a single closed season. Thus, only one week elapsed per season. The next two occasions were omitted, the next two used etc. In some instances where a dramatic change occurred (ie: lake stage or temperature dropped or rose sharply), that sample would be discarded if it could not be grouped with the subsequent
sample occasion of similar lake stage and temperature. This was the only sensible way to ensure that closure assumptions for the statistical model were satisfied.

This technique was applied to the 2002 data, which had been analyzed differently by Muench (2004). Four years later, the program PRESENCE is more powerful, and with the addition of the post-management data we were able to run several multi-season models. Muench (2004) was unable to successfully enter covariates into the program, but this feature was now possible. This method is preferable to using NMS correlation data that does not account for detection probabilities and is not generally advisable for animal studies. In light of these changes, any future sampling should be adjusted to avoid data discarding, see Appendix D.

Model Selection Criteria

Multi-season models for each year were run for every species. First, detection probabilities were modeled with a mixture of covariates until a reasonable model was obtained. This “best” model was added to by incorporating covariates into the occupancy (psi) and colonization (gamma) parameters. In many cases, these models were unable to explain the data adequately, and thus a constant psi model was selected. Seasonal occupancy estimations would not run, returning spurious results resulting in constraining psi as a constant throughout the year. Thus, one occupancy estimate was obtained for each sample year.

Model selection was accomplished using the AIC (Akaike Information Criterion) method. It is a method based on likelihood, which applies penalties for incorporating too many parameters into the model (MacKenzie et al. 2006). The simpler, more parsimonious the model, the better. The actual AIC value reported is not of importance, but rather the differences in AIC between competing models that allows one to select “the best” model. Burnham and Anderson (2002) provide a rule of thumb that models with AIC differences (ΔAIC) of 2 or less have a
substantial level of support. As it turns out, there are often several contending models, with no “best” model.

Before deciding on the model(s) to draw inferences from, output results were examined. Several issues may appear in the results output, that were not evident in the model list. If an animal was captured very infrequently resulting in low detection probabilities (<0.3), this tended to drive occupancy estimates towards 1.0. Thus if a psi value close to or equal to 1.0 was reported, the p value and detection history was scanned to see if psi was a reasonable estimate. Many species with infrequent captures returned spurious results that were ranked among the top models, so these models had to be discarded. If the species was detected so infrequently that results appeared suspicious, the species was determined to be unanalyzed.

Some model outputs contained the warning “Numerical convergence was not reached. Parameter estimates converged to approximately x significant digits”. However, greater than 3 significant digits reported was okay, and the warning could be safely ignored. Anything less and the model was suspicious.

The standard errors and confidence intervals for each parameter estimate were examined. If they were very large relative to the estimate itself, particularly for more than one parameter (ie: psi and p were both outrageous) then the model was deleted. This technique was slightly problematic, because the relatively small number of transects as well as sparse detection histories of some species would necessarily result in higher errors, although the general pattern was still evident. Thus discretion on a case by case basis was used to decide if a model was appropriate or not.

To summarize, detection histories for individual species were inputted along with site and sample covariates into the program PRESENCE. Models were first run to find which covariates
best explained detection probability, $p$. Once settled, that model was tested with covariates for psi and gamma to explore which models fitted the data best. Using the AIC method, the highest ranked models (anything within 2 $\Delta$AIC) were carefully examined for any dubious results including inflated psi values, very low $p$ values, no numerical convergence and high standard errors. Once satisfied with the model selection outputs, the best model(s) were used for inference about the system.

Figure 2-1: Trap set up showing both crayfish and floating minnow trap fitted with new float design
Figure 2-2: Lake Toho transect locations. Inset A shows all transect locations for each year. Note than in 2005-2006, all the yellow 2002 dots were also sampled for the first part of the sampling, but were scaled back to only the study area (blue dots). See text for details. Insert B shows transect set-up post-management in open water. C shows control and treatment plots just prior to re-flooding of newly scraped study areas. The bottom right corner shows a spoil island.
CHAPTER 3
RESPONSE OF HERPETOFAUNAL COMMUNITIES

Introduction

Wetland ecosystems can support a diversity of herpetofaunal species (hereafter referred to as herps) including snakes, frogs, salamanders and turtles. The shallow vegetated littoral zone offers critical habitat used for refuge, feeding and reproduction. Some are considered semi-aquatic, intermittently using terrestrial and littoral habitats. For example, some turtles spend much of their life in the water, but emerge during breeding season to nest in upland habitat. Others are entirely aquatic, carrying out all of their life functions in the water.

Most Florida wetlands are dynamic ecosystems, with fluctuating water levels throughout the year that consequently alters the physical and chemical environment. Native fauna have adapted to these changing conditions, even developing means by which to survive extreme conditions. Amphibians tend to be less mobile, occurring in metapopulations and exhibiting high site fidelity (Harrison and Taylor 1997, Hanski 1999, Sinsch 1990, Berry 2001). Thus, a typical response to drought conditions is to burrow into mud, where some species (ie: sirens and amphiumas) can survive without food and water for prolonged periods of time. More mobile species can either follow the receding water or migrate in search for suitable habitat. The extremes to which these characteristics limit adaptation and survival in a changing environment will vary greatly depending on the species and surrounding landscape (Smith and Green 2005).

The littoral zones of many Florida lakes have dramatically changed since the onset of flood control measures and other human disturbances. On Lake Toho, years of restricted water levels coupled with both natural and anthropogenic eutrophication has resulted in dense monospecific stands of *P. cordata* in the littoral zone. As this continually grows and dies back, a layer of organic sediments (muck) accumulates. This layer is normally considered detrimental to wildlife
and leaves many lake-users disappointed with the aesthetics and limited boat access. This muck and dense vegetation is continually targeted for removal by lake managers, to offset succession of lake degradation as well as to promote recreational use.

Aresco and Gunzburger (2004) comment on the ‘misconception’ that this organic sedimentation is unnatural and harmful to wildlife, water quality and recreation. In fact, some herps are actually associated with this muck such as sirens, amphiumas and snapping turtles (Carr 1940, Bancroft et al. 1983, Aresco and Gunzburger 2004). Macrophytes and the flocculent layer of organic detritus provides a rich source of invertebrate prey (Butler et. al 1992, Schramm et al. 1987, Schramm and Jirka 1989) and also permits burrowing to escape drought.

Amidst the plethora of studies documenting the benefits of management projects to sportfishes and overall habitat improvements, there is scant information regarding the impacts on herp species. One study, by Aresco and Gunzburger (2004) noted dramatically high mortality rates of several species during sediment removal operations. Johnson (2005) identify major gaps in the understanding of herps in Lake Okeechobee, the most well-studied lake in Florida. Their review discovered little information other than species lists. In fact, the only relevant literature available was regarding alligators (Alligator mississippiensis). If draw downs and scraping continue to be used, then it is imperative to understand how this affects native herp residents. The large scale management that took place at Lake Toho in 2003-2004 provided a unique opportunity to address these knowledge gaps.

**Species Accounts**

Sirens (Siren spp.) and amphiumas (Amphiuma means) are large aquatic salamanders commonly associated with muddy and “weed-choked ditches”, heavily vegetated ponds and lakes, or sluggish streams (Funderburg and Lee 1967, Bancroft et al. 1983 Franz 1995). Both are capable of surviving droughts by burrowing into the mud where they can survive for about 2-5
years without food or water (Martof 1969, Etheridge 1990). Bancroft et al. (1983) found increased amphiuma and siren densities with increasing sediment depth, and absence from bare sandy shorelines. Both are mainly nocturnal, benthic creatures feeding on a diversity of prey items such as crayfish, snails, adult and larval insects, tadpoles, salamander larvae and fish (Hanlin 1978, Martof et al. 1980). They are considered top predators in wetlands and because they are so similar in diet and life histories, they may be restricted by competition or predator-prey interactions (Snodgrass et al. 1999). Hypothesis 2 stated that these species, because of their mud-burrowing habits will decline in occupancy after management operations.

Pig frogs (Rana grylio) are highly aquatic animals, known to occupy marshes containing emergent vegetation, which they use for cover, foraging and substrate for egg-laying (Carr 1940, Wright and Wright 1949, Dundee and Rossman 1989, Conant and Collins 1991). They survive droughts by burrowing into mud and peat (Ligas 1960, Wood et al. 1998), however drought conditions can be detrimental. Ugarte (2004) documented an Everglades drought event in which all juveniles died or immigrated.

Many aquatic turtles and water snakes are also known to inhabit shallow wetlands with emergent vegetation. They tend to be better dispersers than amphibians, although movements are infrequent, occurring only out of necessity such as changing environmental conditions or mate searching (Roe et al. 2003). The stinkpot (Sternotherus odoratus) is highly aquatic and rarely ventures onto land except to lay eggs. They are almost always found in the water, and their drought response is to follow the receding water. Should all the water disappear, they will burrow into the mud. Extreme conditions result in temporary cessation of reproduction (Gibbons et al. 1983). The striped mud turtle (Kinosternon baurii) is more terrestrial, foraging through murky shallow water looking for seeds, mollusks, fish and algae (VGDF 2008). They do spend
more time traversing land, and so drying events would probably result in migration from the area to somewhere more suitable for the time being.

Both the Florida banded water snake (*Nerodia fasciata pictiventris*) and Florida green water snake (*Nerodia floridana*) prefer shallow and still wetlands (Ernst and Ernst 2003). Florida banded water snakes less than 50cm are thought to prey on fish, but then switch mainly to frogs once past this size class (Mushinsky et al. 1982). The Florida green water snake feeds primarily on fish but also opportunistically on frogs, salamanders, tadpoles, small turtles and invertebrates (Ernst and Ernst 2003). As with other herpetofauna, the extent to which these animals will travel and utilize terrestrial, littoral and pelagic habitats depends on specific life history traits. Their movements are probably influenced by the distribution of wetlands and preferred prey in the landscape (Roe et al. 2004).

**Results**

Before management in 2002, all species were estimated to occupy a high percentage of the littoral zone, ranging from approximately 60-90%. Following management all declined significantly. In fact, they were captured in such low numbers that they were not even statistically useful in the first year post-management, with the exception of the stinkpot. Amphiumas were not encountered at all in this time, and so rarely in the second year that analysis was not possible. Throughout the post-management sampling, both water snake species had so few captures that they were inestimable. By the second sampling year, all other species showed slightly increased occupancy, with sirens and striped mud turtles showing a marked increase.

The following occupancy estimates for each species are reported as proportion of area occupied with the standard error following in brackets. The first percentage listed is from 2002,
the next from 2006-2007 for all species, since 2005-2006 estimates were not possible. The only exception was the stinkpot. Figures 3-1 through 3-7 display these results.

- Amphiuma occupancy was 0.84(0.14), afterwards inestimable. Figure 3-1.
- Siren occupancy was 0.74(0.07), then 0.48(0.13). Figure 3-2.
- Pig frog occupancy was 0.88(0.10) then 0.29(0.21). Figure 3-3.
- Banded water snake occupancy was 0.84(0.12), afterwards inestimable. Figure 3-4.
- Green water snake occupancy was 0.59(0.22), afterwards inestimable. Figure 3-5.
- Stinkpot occupancy was 0.69(0.25) then 0.21(0.08), then 0.35(0.09). Figure 3.6.
- Striped mud turtle occupancy was 0.91(0.08) then 0.74(0.15). Figure 3-7.

Although some animals did not appear in the estimates due to low detection, they are worth briefly noting. Prior to management mud snakes, peninsula cooters and striped crayfish snakes were captured, but were never encountered afterwards. Other species such as cottonmouth, snapping turtle, Florida softshell turtle and leopard frog were also captured prior to management, but were only encountered sparingly afterwards. Leopard frogs were captured 15 separate times in 2002, not once in 2005-2006, and only 6 times in 2006-2007. Refer to Table A-1- in Appendix A for species list.

**Discussion**

The response of the herpetofauna was clear—all species’ occupancy declined dramatically after management. Except for the stinkpot, all species were caught in so few numbers in the first sample year that estimates were not even possible, with amphiumas going completely undetected the entire first year. The second year post management showed some improvement for pig frogs, stinkpots, striped mud turtles and sirens, particularly the later two. However, both water snake species as well as amphiumas were still inestimable due to low detection. Many of the scraped areas were essentially barren for the first three years (Brush et al. 2008), which probably relates
to the lag in recovery for most species. All species had relatively high occupancy pre-
management thus our hypothesis stating that animals with high pre-management occupancy
would decline afterwards is strengthened. We could not test for the hypothesis that these animals
would also occupy control habitats, as data were too sparse and hurricanes effectively reduced
control habitats in the first year..

**Amphibians**

The increase in siren occupancy implies that organic sedimentation and vegetation had
attained suitable levels, as this species is highly dependent on both. Oddly, amphiumas did not
show increased occupancy by the second year despite their similar habits to the siren. Both
salamander species burrowed into the muck during drawdown and had the scraping not occurred,
would probably have survived, as they are capable of doing so for long periods of time. The
scraping process unearths aestivating animals and crushes them with heavy machinery (Aresco
and Gunzberger 2004, Muench 2004). Potentially large numbers were uncovered and killed,
leaving behind a depleted population left to recover in extremely different habitat.

Surmising reasons for the discrepancy between siren and amphiuma recovery is purely
speculative since little is known about either species. Their similar morphologies, life histories
and diets might lead to competitive exclusion. Snodgrass et al. (1999) suggest that these two can
coexist under favorable conditions such as long hydroperiods absent from complete drying, less
wetland isolation, low disturbance frequency and high immigration rates. Prior to management
they did coexist, occupying a large proportion of the lake. The ample habitat on such a large
lake probably allowed for mutual coexistence. There are many possibilities as to why
amphiumas might have failed to respond similar to sirens. Perhaps they are less tolerant of
adverse conditions, have lower foraging efficiency, lower reproductive capacity, increased
susceptibility to predation or vulnerability to disease. Johnson (2005) cites amphibian disease as
a major stressor for animal populations. Any of these factors might have been exacerbated by stressful conditions, contributing to their failure to recover.

Pig frogs declined significantly after management, plummeting from very high occupancy with only a moderate recovery in 2006-2007. It is likely that lack of vegetative substrate protection, reproduction and resulting diminished prey base were to blame for the decreased occupancy of these frogs. On a positive note, evidence from one study suggests that these frogs (and perhaps leopard frogs which were captured infrequently throughout the entirety of the study) might find permanent retention ponds in residential developments to be satisfactory habitat (Delis et al. 1996). It is possible that during the management project, they escaped to and found refuge in nearby urban ponds. If they can avoid the other calamities afflicting amphibians, they might eventually rebound at Lake Tohopekaliga and even flourish in adjacent ‘wetlands’.

Amphibians are particularly susceptible in today’s world of climate change and rapid development. Globally, they are experiencing declining populations and extinctions (Houlahan et al. 2000, Stuart et al. 2004). There are several possible reasons including low vagility, narrow habitat tolerance, habitat loss and degradation, edge effects, high vulnerability to pathogens, invasive species, climate change, UV-B exposure and pollution (Stuart et al. 2004, Cushman 2006). Furthermore, many life history characteristics such as delayed sexual maturity, low reproductive rates and high juvenile mortality render many herpetofaunal species even more vulnerable to environmental changes. These traits can exacerbate an already stressed population that has lost many adults, making recovery very difficult (Congdon et al 1994, Snodgrass et al. 1999, Klemens 2000). Given that amphibians are considered important components of aquatic food webs and ecosystems, are globally considered an at risk animal group, and are sometimes
commercially and recreationally important, it is crucial that management decisions consider these animals.

**Reptiles**

Reptiles were similarly affected by the management. However, striped mud turtles made a significant recovery, and perhaps will completely recover in the years to come. The reason for such resilience is probably reflected in their more terrestrial nature. Striped mud turtles are semi-aquatic, but spend much of their time on land. Unlike some animals that burrowed into the mud to survive the dry-down, and were then subsequently removed during sediment removal operations, the striped mud turtles might have been able to leave during the process. Perhaps they were able to survive in the first year despite the bare littoral zone, opportunistically foraging in both water and on land. By the second year, substantial vegetative re-growth might have provided better cover and food resources to encourage re-colonization, boosting occupancy levels close to that which they once were.

Stinkpots were probably still estimable in the first year after management because of their highly aquatic nature. Although they are also known to inhabit shallow muddy water, they tend to occur more in permanent aquatic habitats and rarely leave the water except to lay eggs (Gibbons et al. 1983, VDGIF 2008). Although they too are mud burrowers, they tend to follow the receding water (Gibbons et al. 1983) before burrowing and thus some might have remained in the wet portions of the otherwise dry lake, escaping excavation. They were still impacted by the process though, as indicated by their low occupancy estimates in the two years after management. Although only half what they were in 2006-2007, it is still a substantial improvement in a short time.

Marchand and Litvaitis (2004) found more turtles in ponds with organic substrate and shoreline vegetation. They suggest that organic substrate provides better foraging opportunities
compared to hard substrates. However they also note that dense vegetation might reduce the suitability of the habitat by restricting turtle movement. This is probably more relevant for larger bodied turtles not studied in this project. Smaller turtles might navigate dense vegetation easier - indeed they had high occupancy estimates prior to management.

The Bigger Picture

There are a wide range of issues affecting herp populations that extend beyond the lake ecosystem itself, and also beyond the scope of this study. Some were mentioned earlier, such as sensitivity to climate, UV, pollution and disease. Additionally, the surrounding terrestrial landscape will have huge impacts on the lake and its resident fauna. Lake Toho has increasingly become isolated from other wetlands due to rapid urbanization including road networks and buildings. In addition to how asphalt jungles can negatively impact water quality, they also restrict animal movement between wetlands, lakes and retention ponds nearby. Many aquatic animals emerge from the water to migrate over land, in response to drought or other disturbances, or for reproductive purposes. The extent to which reptiles and amphibians will migrate depends on the species and surrounding landscape (Semlitsch and Bodie 2003). Some snake and turtle species rarely leave their aquatic habitats, whereas others will routinely travel great distances. Most amphibians will not traverse land, except for very short distances to lay eggs. Upland ‘habitat’ that surrounds Lake Toho is risky business for many animals that will encounter automobiles (Aresco 2002, 2005). Female turtles are most vulnerable because they routinely make this journey, which in the long term can skew sex ratios (Marchand and Litvaitis 2004). Johnson (2005) suggests that development and roads within several hundred meters of lake edges can have long term effects on aquatic turtle populations.
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Figure 3-1: Occupancy estimates with standard error for amphiuma, one year pre-management and two years post-management
Figure 3-2: Occupancy estimates with standard error for siren, one year pre-management and two years post-management
Figure 3-3: Occupancy estimates with standard error for pig frog, one year pre-management and two years post-management
Figure 3-4: Occupancy estimates with standard error for Florida banded water snake, one year pre-management and two years post-management
Figure 3-5: Occupancy estimates with standard error for Florida green water snake, one year pre-management and two years post-management.
Figure 3-6: Occupancy estimates with standard error for stinkpot, one year pre-management and two years post-management
Figure 3-7: Occupancy estimates with standard error for striped mud turtle, one year pre-management and two years post-management
CHAPTER 4
RESPONSE OF FISH COMMUNITIES

Introduction

Years of water restrictions coupled with cultural eutrophication and rapid succession of lake ecosystems have promoted conditions favorable to dense monospecific stands of *P. cordata* and *Typha spp.* Dense vegetation can alter the abiotic properties of the surrounding water, resulting in sub-optimal environment for fauna. This can include lethally low levels of dissolved oxygen, temperatures beyond survival thresholds, reduced foraging efficiency due to visual and swimming barriers, and damagingly low pH from plant respiration (Crowder and Cooper 1982, Savino and Stein 1982, Sculthorpe 1985, Frodge et al. 1990, Moyer et al. 1995, Miranda et al. 2000, Miranda and Hodges 2000, Allen and Tugend 2002). These conditions have been linked to stunted fish populations exhibiting reduced growth and overall condition (Colle and Shireman 1980, Crowder and Cooper 1982, Savino and Stein 1982, Bettoli et al. 1992). Offshore berm formation also occurs, which not only imparts the same adverse environment as dense vegetation, but is also thought to restrict fish access to shoreline spawning habitat (Moyer et al. 1995).

These negative attributes have drawn much attention from fisheries managers and scientists, because freshwater recreational fishing is a booming business in Florida. A 2003 report indicated that it has an economic impact of nearly $2 billion per year to the state (Harding 2003). Not surprisingly, maintaining a habitat that supports both juvenile and adult stages of sportfish, and facilitates boater access is an important goal for many lake management projects. Common methods used to achieve such conditions are routine lake draw downs, mechanical sediment removal and herbicide application. Draw downs are intended to mimic natural drought conditions that expose the littoral zone, killing off excessive plants, promoting oxidation,

The unsuitable nature of dense aquatic vegetation has more or less been considered fact for a long time. However, plants are also recognized as providing valuable habitat to juvenile fish and small forage species (Shireman et al. 1981, Crowder and Cooper 1982, Wiley et al. 1984, Killgore et al. 1989, Chick and McIvor 1994, Dewey et al. 1997, Miranda and Pugh 1997, Miranda et al. 2000, Miranda and Hodges 2000, Pelicice et al. 2005, MacRae and Jackson 2006, Bunch et al. 2008). “Largemouth bass anglers, fisheries biologists and other professionals involved in lake management often strongly support the statement that aquatic macrophytes are essential for largemouth bass populations” (Hoyer and Canfield 1996). The precise threshold beyond which aquatic macrophytes are considered problematic is highly controversial, with reference only to an ‘intermediate’ macrophyte density (Crowder and Cooper 1982, Wiley et al. 1984, Olson et al. 1998, Miranda and Hodges 2000). Trebitz et al. (1997) suggest about 20-30 percent removal and not over 50 percent in order to promote bluegill and largemouth bass populations based on computer simulation models. Hoyer and Canfield (1996) refer to a ballpark figure of 20-30 percent of lake area covered (PAC), which is thought to strike a healthy balance for fish in small water bodies (Hoyer and Canfield 1996). But they maintain that any justification for a specific level will remain controversial.
Most studies have been supportive of lake management and its benefit for sportfish (Moyer et al. 1995, Olson et al. 1998, FFWCC 2001, Tugend 2001, Allen and Tugend 2002, Allen et al. 2003). However, Allen et al. (2003) report that while draw downs and muck removals support recreational activities and improve habitat, it is difficult to clearly decipher the impacts on adult largemouth bass. They suggest instead that efforts be focused on recreational benefits rather than towards a single species. To complicate the matter further, many traditional fish sampling techniques fail to adequately sample within dense vegetation and ignore detection probabilities, a critical factor in animal studies (see Introduction- a need for a new approach to sampling and statistical analyses).

Given that recreational fishing is such an important industry, and that there is some uncertainty as to how fishes, even largemouth bass respond to management, some work was needed to address the issue. This study was an important first step in understanding how littoral zone fish communities responded to a large-scale lake management program. Focus needed to extend beyond bass and bluegill in order to understand how other species that comprise littoral food webs are similarly affected. It was also an opportunity to utilize new methods more appropriate for sampling small fish in dense vegetation, as well as to employ some modern statistical analyses.

Results

The response of fishes was variable. Of the 13 estimable species, 5 (bluegill, dollar sunfish, largemouth bass, Seminole killifish and sailfin catfish) showed increased occupancy post-management. One, the redear sunfish did not show much appreciable change. The other 7 species showed notably lower occupancy estimates, in some cases having such low captures that estimates were not possible. The hypothesis stating that animals with high occupancy would decline after management was generally true, except for bluegill and largemouth bass which
were expected to increase, regardless of pre-management occupancy estimates. Both bass and Seminole killifish had higher occupancy in treatment habitats which was also expected, but the bluegill did not, thus the hypothesis was not entirely supported.

The following occupancy estimates for each species are reported as proportion of area occupied with the standard error following in brackets. The first percentage listed is from 2002, the next from 2005-2006 and the last from 2006-2007.

- Bluegill occupancy was 0.38(0.18), 0.85(0.07) then 0.75(0.09). Figure 4-1.
- Dollar sunfish occupancy was 0.18(0.05), 0.47(0.06) then 0.56(0.07). Figure 4-3.
- Largemouth bass occupancy was 0.40(0.27), 0.71(0.17) then 0.64(0.30). Figure 4-5.
- Seminole killifish occupancy prior to management was inestimable, then 0.70(0.20) and 0.73(0.11). Figure 4-7.
- Sailfin catfish occupancy prior to management and in the first year after was not estimable, then rose 0.26(0.11). Figure 4-9.
- Redear sunfish occupancy was 0.59(0.12), 0.49(0.23), then 0.54(0.13). Figure 4-10.
- Armored catfish occupancy was 0.83(0.09), afterwards inestimable. Figure 4-11.
- Blue-spotted sunfish occupancy was 0.77(0.08), 0.28(0.07), then 0.39(0.12). Figure 4-12.
- Chubsucker occupancy was 0.43(0.21), afterwards inestimable. Figure 4-13.
- Gar occupancy was 0.67(0.09), inestimable, then 0.41(0.13). Figure 4-14.
- Sailfin molly occupancy was 0.36(0.12), afterwards inestimable. Figure 4-16.
- Spotted sunfish occupancy was 0.31(0.15), afterwards inestimable. Figure 4-17.
- Warmouth occupancy was 0.84(0.08), 0.53(0.07), then 0.66(0.08). Figure 4-18.

Although many species were not captured enough to allow statistical estimates throughout the study they are worth mentioning. Bowfin, brown bullhead, chain pickerel, flagfish, golden shiner, golden topminnow, redfin pickerel, tadpole madtom and threadfin shad were all captured at some point throughout the study but not consistently enough to allow for estimates.
Interestingly, two exotic species not detected prior to management that appeared sporadically afterwards were blue tilapia and Mayan cichlid. The Mayan cichlid had not been documented here before and a specimen was donated to the Florida Museum of Natural History’s Ichthyology Department.

**Effects of Habitat**

Six out of 13 fish species indicated that habitat influenced site occupancy estimates, including bluegill, gar, warmouth, dollar sunfish, largemouth bass and Seminole killifish. Of the 6, 4 had higher occupancy estimates in control sites. Both largemouth bass and Seminole killifish had higher estimates in all-lake sites. The following occupancy estimates for each species are reported as proportion of area occupied with the standard error following in brackets for each habitat type. These values are also reflected in the figures at the end of the chapter.

Bluegill only indicated influence of habitat on occupancy in 2006-2007, with 0.72(0.10) in control, 0.57(0.10) in treatment and 0.59(0.11) in all-lake habitats. This was the third possible model with a ΔAIC of 1.03 and model weight of 17%, thus it is a plausible model, but it is also likely that occupancy was constant. Either way habitat was influential on this species to some degree because there was also a 20% chance that habitat influenced colonization, this model had ΔAIC of 0.62, with a colonization probability of 0.28(0.13) in control, 0.54(0.21) in treatment and 0.55(0.18) in all-lake habitats. See Figure 4-2.

Gar only indicated influence of habitat in 2006-2007, with 1.0(0.0) in control, 0.32(0.13) in treatment and 0.37(0.15) in all-lake habitats. There is great support for habitat occupancy models, with 2 habitat models ranking as the first two models with a combined weight of 73%. Although the estimate of 1.0 with 0 standard error would initially cause concern for model estimates, upon examining capture data for control habitats, it seems reasonable as captures were
very frequent. Sometimes inflated estimates with low error are reported in cases where detection histories are sparse, but this does not seem to be the case. See Figure 4-15.

Warmouth indicated influence of habitat in both post management years. In 2005-2006, occupancy was estimated at 0.79(0.12) in control, 0.36(0.12) in treatment and 0.46(0.10) in all-lake habitats. There was approximately 75% support for habitat occupancy models in this year. In 2006-2007 occupancy rose in all habitat types, with 0.98(0.18) in control, 0.52(0.09) in treatment and 0.55(0.10) in all lake habitats. Habitat occupancy differences are even more supported in this year at nearly 99% weight. See Figure 4-19.

Dollar sunfish only indicated influence of habitat in 2006-2007, with 0.66(0.12) in control, 0.60(0.13) in treatment and 0.37(0.10) in all-lake habitats. This was the second of two possible models, with a ΔAIC of 0.66 and a model weight of 38%. Thus there is more support for the model indicating constant occupancy across habitats, but the habitat model is still plausible. See Figure 4-4.

Largemouth bass only indicated influence of habitat in 2005-2006, with 0.30(0.26) in control, 0.41(0.28) in treatment and 0.72(0.26) in all-lake habitats. This was the third ranked model with a ΔAIC of 0.41 and model weight of 24%. There is more support (57%) for constant occupancy across habitats, but it is still plausible that habitat influenced occupancy. See Figure 4-6.

Seminole killifish indicated influence of habitat in both post management years. In 2005-2006, occupancy was estimated at 0.41(0.13) in control, 0.56(0.17) in treatment and 0.84(0.22) in all-lake habitats. There was the top ranked model with a weight of 43%. In 2006-2007 occupancy rose in all habitat types, with 0.45(0.15) in control, 1.0(0.0) in treatment and
0.89(0.12) in all lake habitats. Habitat occupancy differences are even more supported in this year with approximately 77% weight, ranked again as the top model. See Figure 4-8.

Figure 4-1: Occupancy estimates with standard error for bluegill, one year pre-management and two years post-management
Figure 4-2: Habitat site occupancy estimates with standard error for bluegill 2006-2007
Figure 4-3: Occupancy estimates with standard error for dollar sunfish, one year pre-management and two years post-management
Figure 4-4: Habitat site occupancy estimates with standard error for dollar sunfish 2006-2007
Figure 4-5: Occupancy estimates with standard error for largemouth bass, one year pre-management and two years post-management
Figure 4-6: Habitat site occupancy estimates with standard error for largemouth bass for 2005-2006
Figure 4-7: Occupancy estimates with standard error for Seminole killifish, one year pre-management and two years post-management
Figure 4-8: Habitat site occupancy estimates with standard error for Seminole killifish for 2005-2007
Figure 4-9: Occupancy estimates with standard error for sailfin catfish, one year pre-management and two years post-management.
Figure 4-10: Occupancy estimates with standard error for redear, one year pre-management and two years post-management
Figure 4-11: Occupancy estimates with standard error for armored catfish, one year pre-management and two years post-management.
Figure 4-12: Occupancy estimates with standard error for blue-spotted sunfish, one year pre-management and two years post-management
Figure 4-13: Occupancy estimates with standard error for chubsucker, one year pre-management and two years post-management.
Figure 4-14: Occupancy estimates with standard error for gar, one year pre-management and two years post-management

* inestimable
Figure 4-15: Habitat site occupancy estimates with standard error for gar 2006-2007
Figure 4-16: Occupancy estimates with standard error for sailfin molly, one year pre-management and two years post-management
Figure 4-17: Occupancy estimates with standard error for spotted sunfish, one year pre-management and two years post-management

* inestimable
Figure 4-18: Occupancy estimates with standard error for warmouth, one year pre-management and two years post-management
Figure 4-19: Habitat site occupancy estimates with standard error for warmouth 2005-2007
Discussion

Previous fisheries studies linking the effects of lake management to fish populations had three major shortcomings. First, traditional fish and herpetofaunal sampling methods could not sufficiently characterize occurrence of species in dense vegetation. Second, all studies have either employed raw count data to infer abundance and population estimates, or creel surveys which provide only angler access information, rather than any real clues as to habitat quality or use. Finally, no studies have ever examined the impacts of management on species other than sport fishes.

This study has addressed all three of these concerns. The trapping protocol effectively sampled within dense vegetation stands including dense floating mats, and it also performed well in open water habitat in the following management. Raw counts, abundances and angler surveys were abandoned for modern statistical and methodological approaches using site occupancy and detection probability estimations. Perhaps most importantly, we were able to document changes in occupancy of several species including sport fishes, before and after management.

The response of littoral fish communities varied by species. Of the thirteen estimable species, five showed a positive response after management including two important sportfish, the largemouth bass and bluegill as well as dollar sunfish, Seminole killifish and sailfin catfish.

The three other sportfish, redear, warmouth and spotted sunfish declined after management. Redear declined only slightly followed by a slight increase, the change was minimal compared most other species. Three forage species also declined including blue-spotted sunfish, chubsucker and sailfin molly.

Unfortunately the hurricane effects from 2005 complicated our desire to understand post-management treatment effects. Control and treatment habitats were similar at least in the first year afterwards, with controls slowly recovering by the second year and treatments remaining
relatively bare (Brush et al. 2008). In the absence of hurricanes, we would have expected more differences in habitat occupancy than the mere three species observed in the first year post-management (bass, warmouth, Seminole killifish). By the second year, as habitats were becoming more distinguishable, four species showed a difference. It wasn’t until the final June 2008 vegetation sample that relatively stable control plots were evident as well as treated areas colonized with eelgrass communities (Brush et al. 2008).

Hurricane effects were probably not the only limiting factor affecting estimation possibilities. Low captures of many animals coupled with relatively few site replicates limits complex covariate modeling. In most cases, only constant occupancy models were possible. Despite not being able to tease out the finer details of habitat differences, overall occupancy changes are still very telling of how a species responded to management, confounded by hurricanes at the lake wide scale.

**Largemouth Bass**

The long studied and valuable largemouth bass have been reported to benefit from management activities (Moyer et al. 1995, Olson et al. 1998, FFWCC 2001, Tugend 2001, Allen and Tugend 2002, Allen et al. 2003). This provides incentive for managers to continue with draw downs and muck removals as they seem to improve habitat. In this study, juvenile bass increased occupancy immediately following management. There is evidence that habitat influenced either detection probability, occupancy, or perhaps both to some degree in the first year following management. The most likely model indicates that habitat influenced detection, and occupancy remained constant. Detection probability was lowest in control, highest in treatment then all-lake sites, reiterating why detection probabilities are so important to consider. Count statistics would have implied that abundance or density estimates were different across habitats, when in fact they were not.
An alternative but less likely possibility is that occupancy was in fact influenced by habitat. Occupancy was similarly affected as detection probability. That is, lowest in control, then treatment and all-lake. Interestingly, both overall occupancy estimates for each year as well as each habitat, had fairly large standard errors, just under 30%. These less precise estimates potentially make differences between years and habitats less striking than they first appear.

Either way, if abundance is the underlying cause of higher occupancy and/or detections in treated areas, then we might infer that more juvenile bass occupied the lake after management, particularly in treated areas in the first year. This would corroborate earlier studies that found increased bass abundance in enhanced lakes (Moyer et al. 1995, Allen et al. 2003).

Juvenile fish are typically associated with shallow vegetated habitat that they use for forage and cover (Hoyer and Canfield 1996, Miranda and Pugh 1997). An apparent increase in occupancy after scraping seems perplexing, because open water provides neither forage nor cover. The explanation might lie in the availability of sandy substrate for spawning adults, resulting in more opportunity to reproduce. Less heavily vegetated habitats have been associated with increased recruitment and growth rates, as this type of habitat is thought to strike a balance between protection and prey availability (Miranda and Pugh 1997).

The immediate increase in occupancy seen in 2005-2006, was not sustained the following year, nor the effect of habitat on occupancy and detection probability. Occupancy may continue to decline approaching pre-management levels as control plots stabilize and *P. cordata* threatens to dominate again. However, if the *V. americana* community currently established in scraped areas remains competitive, it might support both juvenile and adult bass populations. Further monitoring will help us to understand the long term management effects, including how recovering plant communities impact resident fishes.
There was no concurrent study looking at the effects of management on adult bass. However, a similar management activity conducted on Lake Kissimmee in 1995-1996, was unable to detect any increase in adult bass (Allen et al. 2003). The goal of the project was to improve largemouth bass fishing, but electrofishing and angler catch rates could not confirm this. In fact, fishing effort actually declined.

**Bluegill**

Bluegill also increased in occupancy quite dramatically following management, perhaps for similar reasons as largemouth bass. Although habitat differences were not readily evident until the second year, the species clearly benefited from enhanced habitat. Highest occupancy was in control, then all-lake and treatment habitats (treatment and all-lake were very similar). Interestingly, there was also support for habitat colonization differences which did not occur for any other species. In this scenario, colonization probability was highest in all-lake then treatment and control. At first, this seems to contradict the habitat occupancy model. But the contending models might not be mutually exclusive. Perhaps preference for control sites reflected by high occupancy, results in lower colonization rates due to saturation of the habitat. As vegetation recovers in scraped areas, more vegetated habitat becomes available and individuals rapidly colonize.

**Other Fish Species**

Besides bluegill and largemouth bass, four other species including dollar sunfish, gar, Seminole killifish and warmouth showed habitat occupancy differences. All but the Seminole killifish had highest occupancy in control sites.

The other three sport fish encountered in this study, the spotted sunfish, redear and warmouth did not show any positive response to management. While the spotted sunfish and warmouth declined, redear barely changed, remaining relatively stable prior to and after
management. The main food sources of redear are small snails and mussels, supplemented also with aquatic insects (Carlander 1977, Lee 1980), thus the common name “shell-cracker”. Perhaps these food sources were still plentiful post-management, such that both adults and juveniles were able to cope with the changed environment. Also, they do not seem particular as to the type of substrate they nest in, having been documented in both sandy and soft muddy bottoms as well as in aquatic vegetation (Wilbur 1969). Thus they were not limited in nesting availability, nor benefited from more open sandy bottoms, perhaps resulting in their indifference to the management.

The apparent partiality for control habitats in both post-management years exhibited by warmouths, in addition to their overall decline is not surprising given their known preference and/or tolerance for sluggish swampy conditions (Lee 1980, FFWCC 2008). Males are thought to construct nests near clumps of vegetation (Larimore 1957), which was lacking immediately after management. This may have reduced spawning or maybe just increased juvenile susceptibility, as they are not guarded past 5 or 6 days after spawning (Carlander 1977). Their increase by the second year suggests the return of pre-management vegetation conditions, more suitable to their habitat preferences and nesting requirements.

Spotted sunfish are thought to inhabit sluggish, heavily vegetated environments with sandy gravel bottoms (FFWCC 2008). They are a ubiquitous species, however their relatively low occupancy on Lake Toho prior to management as well as continually low captures afterwards, will likely prevent them from becoming a common species on this lake.

Similar to the warmouth with preference for swampy, muddy, heavily vegetated habitats, gar also declined after management and indicated a preference for control habitat in the second year post-management. Gar are particularly adaptable to anoxic conditions, with their ability to
take in oxygen through their air bladder (FFWCC 2008). This species is also showing recovery which indicates return of their preferred habitat conditions.

Seminole killifish showed higher occupancy in all-lake and treated sites for both post-management years. This is not surprising, as this species has commonly been associated with open sandy habitats, indicative of successful restoration (Wegener and Williams 1975, Moyer et al. 1995, Tugend 2001).

Dollar sunfish would be expected to prefer vegetated habitat (Bauer 1980, Hassan-Williams and Bonner 2007), which they do indicate, but their overall occupancy rates increased after the management. They seem to have varied diets and are both benthic and surface feeders (Goldstein and Simon 1999). This versatility might enable them to be opportunistic, able to exploit both open and vegetated areas. Due to their small size, they might have preference for vegetated habitats, but likewise capitalize on open water environments as well.

Small forage fish that are typically associated with shallow, vegetated still waters, such as the lake chubsucker, sailfin molly and blue-spotted sunfish all responded negatively to management. The chubsucker and sailfin molly have not yet shown any signs of recovery, but it would be anticipated given the increase in vegetation biomass returning to the lake. Some fish and herp species are also signaling that pre-management habitat conditions are returning, which should eventually benefit these small forage species.

The armored catfish (brown hoplo) declined in occupancy after management, while the sailfin catfish increased. The armored catfish is a highly adaptable creature, surviving hypoxic conditions with their ability to breathe air (Brauner et al. 1995, Affonso and Rantin 2005). They are also aggressive nest guarders (Nico et al. 1996). These traits likely allowed its persistence in the pre-management habitats, and we would expect such an aggressive species to continue...
flourishing. Maybe they were out-competed by other species afterwards or were directly
affected by the scraping process. We would expect this species to recover based on its highly
competitive nature, especially if the original vegetation communities begin to establish.

Sailfin catfish only showed an increase in the second year, and were captured so
infrequently prior to management and in the first year after, that it is impossible to conclude
whether or not their increase was a direct result of the management. Similar to many exotic
species, they exhibit hardy traits allowing them to survive and compete with natives. They are
known to construct burrows into mud, which are then used for reproduction but also allow them
to survive drought conditions (Hoover et al. 2004). They have been found in dried-out burrows
appearing dead, but are in fact alive and once returned to the water, revive quickly (Hoover et al.
2004). This adaptation to extreme conditions has likely led to their spread and ability to survive
and compete.

Two other exotic species, the blue tilapia (*Oreochromis aureus*) and Mayan cichlid
(*Cichlasoma urophthalmus*) were not encountered prior to management, but were found in low
numbers afterward. Blue tilapias have been established in southern and central Florida for
sometime, and are known to inhabit Lake Toho. The Mayan cichlid had not been documented at
Lake Toho and specimens were submitted to the Florida Museum of Natural History in
Gainesville, FL for cataloging. Whether or not the appearance of the fish are related to
management is unclear. It is possible that the disturbance allowed them to compete with natives,
but it could also be a coincidence in that they first appeared post-management.

**Summary**

Overall it appears that only a fraction of fishes benefited from the management, while the
rest declined. Two important sport fishes (bass and bluegill) that showed increased occupancy in
the first year, declined somewhat in the second year. Three other sport fish (reedar, spotted
sunfish and warmouth) did not respond positively. It would be useful and worthwhile to conduct further studies to assess the longevity of management effects on all fishes. The short term study as well as lack of adult sampling leaves some ambiguity as to the real and long-term impacts of management on various fish species. Thus, concurrent with Allen et al. (2003), it is very difficult to detect the true response of bass to management, as well as other species. Results would have been more comprehensive if we knew whether or not the increase in juveniles actually resulted in successful recruitment into the adult population. Of course the hurricanes also complicate the results.

According to vegetation studies (Brush et al. 2008), control areas have stabilized with recovery of *P. cordata* and in scraped areas, *V. americana* has become quite abundant. This has been accompanied by the recovery of warmouth and gar, two species that naturally tolerant of densely vegetated, soft-bottom environments. As noted in chapter 3, sirens also recovered substantially by the second year, another species indicative of pre-management conditions.

Although focus is typically directed towards recreational sportfish for management studies, also understanding how other species are affected by management is important. Small forage species provide vital links between smaller epiphytic fauna and larger predatory fish, herps and birds. Although single species management for largemouth bass will probably remain the underlying motivation for many future activities, it is important to have a holistic ecosystem approach. Species diversity is often touted as the key to healthy ecosystems, which in the end is a net benefit to aquatic fauna, managers and lake users. Continued monitoring should be implemented, to track how the succession of lake vegetation continues to shape faunal communities.
CHAPTER 5
RESPONSE OF NATIVE AND EXOTIC APPLE SNAIL COMMUNITIES

Introduction

Although apple snails were not intended to be studied, they were captured and recorded throughout our sampling. The exotic island apple snail (*Pomacea insularum*) appeared on the lake shortly before management, providing an opportunity to document the response of both the native Florida apple snail (*Pomacea paludosa*) and the exotic to the management. The presence of the exotic apple snail from South America presents stiff competition for the native species.

They have a voracious appetite, consuming almost any aquatic plant in sight (Power 2007). Egg masses are large with the average clutch containing over 2000 eggs, and a field hatching success estimated at 70% (Barnes et al. 2008). In the warm months typical of the southeastern US, a new clutch every week is possible (Barnes et al. 2008). Not surprisingly, it is considered a major pest in many parts of the world where it has been introduced. Its threats to Florida’s wetland ecosystems are serious. This aggressive and rapid reproducing species will likely compete with the native apple snail, a critical prey item for the snail kite (*Rostrhamus sociabilis*). It might also incur damage to wetland plant communities.

Although the native Florida apple snail is well adapted to the dynamic Florida ecosystem in which it evolved, competition with a much larger and more actively reproducing species could be a serious problem. Natural drying events are survived for weeks or months at a time by aestivating, since they are not capable of moving far. Snails are stranded on dry ground and will wait for the water to return (Darby et al. 2002). The chances of surviving such conditions are much higher for adults than juveniles.
Individual eggs are much larger than that of the exotic species, but clutches contain far fewer eggs, about 10-80 on average (Brown 2005). While drying processes are a natural part of the Florida ecosystem and are beneficial to apple snail habitat and reproduction, prolonged drying or flooding events and the timing in which they occur is critical to this species. This species is considered an annual breeder, with the peak from April-June. Essentially they have one opportunity to successfully reproduce (Darby et al. 2008). If conditions are excessively dry or wet, reproduction will fail that year.

Although the native Florida apple snail is well-adapted to its environment, the presence of a potentially aggressive competitor could cause problems. It is possible that the management created a disturbance that might favor the establishment of the exotic. Here we present occupancy trends over time for both species before and after the management.

**Results**

The following occupancy and colonization results are indicated in percentages, with standard error following in brackets. Exotic apple snails were not detected prior to management, but in the first year after, were estimated to occupy 45\%(0.10) of the lake, further increasing the following year to 83\%(0.06) (Figure 5-1). In the first year control and treatments had nearly equal occupancies, 53\%(0.15) and 52\%(0.15) respectively, with lower occupancy occurring in the all-lake sites at 20\%(0.12) (Figure 5-2). In the second year, habitat occupancy models were not successful, instead a colonization model indicated differences between habitats. The probability of colonization was much higher in control, at 76\%(0.21) compared to both treatment and all-lake areas, 3\%(0.10) and 6\%(0.06) respectively (Figure 5-5).

Native apple snails declined dramatically in the first year after the scraping, but increased beyond pre-management levels by 2006-2007 (Figure 5-1). In 2002 occupancy was estimated at 66\%(0.11), compared to 13\%(0.0427) then 80\%(0.08) in the succeeding post-management
years. Differences in occupancy between habitats was only evident in the second year, with highest occupancy estimated in control sites at 90 % (0.08), then treatment at 69% (0.12) and all-lake at 53% (0.23) (Figure 5-3). In 2002 colonization probability was fairly low at 15 % (0.09), dropped in 2005-2006 to 7% (0.03), and then increased dramatically in the final sample year to 43% (0.14) (Figure 5-4).

In the second year, overall occupancy estimates for both species were quite high and close to one and other, with exotics indicating only a slightly higher occupancy. Native occupancy was estimated at 80% (.08) and 83% (0.06) for exotics (Figure 5-1).

Figure 5-1: Occupancy estimates with standard error for both snail species, one year pre-management and two years post-management
Figure 5-2: Habitat site occupancy site estimates with standard error for exotic apple snails 2005-2006
Figure 5-3: Habitat site occupancy estimates with standard error for native apple snails 2006-2007
Figure 5-4: Colonization estimates with standard error for both apple snail species, one year pre-management and two years post-management
Figure 5-5: Habitat colonization estimates with standard error for exotic apple snails 2006-2007
Discussion

It is not certain when exactly the exotic apple snail arrived at Lake Toho, only that it was observed in 2003 in Goblets cove (field staff, personal communication). Its establishment has been remarkable, with a spike from no detection at all in 2002 to an estimated 45% lake wide occupancy in 2005-2006, then nearly doubling the next year to 83%. This rapid increase confirms their ability to rapidly reproduce. One individual female can produce thousands of eggs per season (Kolar and Lodge 2001, Barnes et al. 2008), with an estimated successful hatching rate of over 70%. Their successful spread within a short time period will probably lead to competitive interaction with the native apple snail (Halwart 1994), and there may also be the potential that the two will hybridize (Rhymer and Simberloff 1996).

As of this writing, both species have rapidly colonized since the management and are estimated to occupy just over 80% of the lake. We have no idea how this will play out, whether or not the exotic will continue to occupy the entire lake out-competing the native, or if the highly resilient native will still be able to successfully compete. Rawlings et al. (2007) noted a decline of the Florida apple snail in the presence of this exotic species.

In the first post-management sampling year, control and treatment habitats were occupied roughly the same, with lower occupancy in the all-lake habitats. This is probably an artifact of the active 2004 hurricane season that essentially rendered control and treatments areas the same, especially in the south end of the lake. The all-lake areas were completely devoid of any vegetation or organic sediments and thus would have been unsuitable habitat for either species, with no means by which to emerge from the water to breathe or lay eggs. The fact that they were captured at all in such barren areas might be the result of the trap’s presence, offering a substrate for both of these important activities.
By the second year, colonization of the exotic species was extremely high in control plots at 76%, with a mere 3% and 6% for treatment and all-lake respectively. Although this species was capable of early establishment despite sparse plant communities, it is evident that vegetated areas are preferred. We have three pieces of evidence to support this. First, near equal occupancy in the first year between control and treatment habitats, compared to much lower occupancy in all-lake areas. Second, their high colonization probability in control sites in the second year. Third, the near doubling of occupancy by the second year which was probably at least in part due to recovering vegetation communities.

The island apple snails’ persistence in the first year post-management is testament to their ability to survive adverse conditions. Not only did they endure the management operations, but they were able to establish in a lake ecosystem almost completely devoid of vegetation, an important component of apple snail habitat.

The native Florida apple snail appears highly resilient. Prior to management it was fairly widespread on Lake Toho estimated to occupy about 66% of the lake. The management greatly affected this species, as reflected by a sharp decline in occupancy to 13%, coincidentally synchronous with the population explosion of exotic snails. The natives were captured too infrequently to offer any clues as to habitat effects. However their dramatic drop in occupancy is telling of their vulnerability to the management operations and/or resultant habitat. Despite this, they appear highly resilient, showing an even larger site occupancy estimate by the second year than they did prior to management (80% compared to 66%). Although they still lagged slightly behind the exotics, this was a significant recovery in a short time period. This resilience is also reflected in the much higher colonization in the second year (43%), compared both to pre-management (15%) and the first year after (3%).
By the second year we were able to detect differences in occupancy between habitats, with control being the most highly occupied, then treatment and all-lake. This apparent preference for vegetated habitat is concurrent with Karunaratne et al. (2006) who found that native apple snail densities were much greater in wet prairie habitats than sloughs, the main difference being the presence of emergent vegetation in the wet prairies and none in the sloughs. The authors attribute this pattern to the snails’ use of emergent vegetation for oviposition and for climbing out of the water for aerial respiration.

Eggs must be laid several centimeters above the water (Turner 1996), thus requiring tall emergent vegetation and precluding floating leaf communities as suitable egg deposition sites. However, the density of emergent plants is likely to influence habitat usage. Karunaratne et al. (2006) noted that dense stands of *Eleocharis* had lower snail densities and the authors surmise that dense vegetation might hinder horizontal movements of snails as well as their ability to climb vegetation. They noticed less periphyton (a major food source) in dense *Eleocharis* stands as well. This might explain the higher occupancy noted in 2006-2007 compared to pre-management. Pickerelweed communities were very dense prior to the management, but had not yet reached such intensity after management.

Peak egg laying and hatching occurs in the dry season in April-May with hatchlings reaching adult size by June. With the drawdown on Lake Toho beginning in November 2003, peak reproduction had already occurred and most individuals were adults, better able to survive drought conditions (Darby et al. 2002). Thus the native snail was likely spared imminent danger by having many adults in the population, that were either able to follow the receding water or perhaps evade removal by remaining stranded far enough on shore to miss the bulldozers.
This corroborates accumulated evidence that the Florida apple snail is tolerant and adaptable to periodic drying events. Not only that, they were able to make a significant recovery in a short time, despite a sudden decline in occupancy immediately after the management. The reduction of high density plant stands may also have contributed to their success.

The exotic island apple snail appears a very aggressive competitor, able to spread quickly and occupy nearly half the lake within a very short time. This was accomplished in an environment containing very sparse emergent vegetation and exposed shorelines subject to wind and wave action. During this time the natives were captured infrequently, as they were attempting to recover from the management and perhaps also competing with the exotic. The native was able to make a strong recovery, and by the last year was essentially at par with the exotic that had nearly doubled its occupancy within a year. Perhaps the differences in diet (exotics consume vegetation, natives periphyton) will allow the two to co-exist even if the exotic remains widespread. Knowing how the native will respond is unclear without further monitoring.

As vegetation continues to recover, the snail populations will probably react. Continued monitoring would be useful to reveal how the native versus exotic species saga plays out, and how the long-term effects of the management project continue to influence snail occupancy.
CHAPTER 6
FINAL CONCLUSION AND IMPLICATIONS FOR FUTURE MANAGEMENT

As Florida’s wetlands continue to disappear and become impacted by human development, it becomes even more important to understand how local fauna respond to management actions. Most remaining wetlands including large lakes such as Lake Toho are becoming isolated in the landscape, fragmented by road networks and development, effectively reducing habitat availability for many animals. Ironically the purposeful human alteration of water bodies has had such major unintended consequences, that even more human involvement is required to ameliorate the problems. Management actions act only as band aids temporarily offsetting the problems, because the actual cure is not possible. Intelligent decisions are needed that try to strike a balance between the needs of several species, maintenance of a viable sportfish industry and other recreational uses, and flood control. This study was a first step in understanding the impacts of management on vegetation, birds, herps, fish, and apple snails (Muench 2004, Welch 2004, Brush 2006, Brush et al. 2008?).

The main goals of Florida lake management are typically to offset or stall succession, remove undesirable accumulations of plants and muck, promote quality habitat and open up boater access. Arguably, the main impetus of all these is to preserve sport fish populations. However the ambiguity involved in whether or not important species such as largemouth bass are actually benefiting from management practices has led to the idea that management goals really ought to be more about the habitat quality and recreational value rather than targeted toward a single species (Allen et al. 2003).

This study documented an immediate increase in occupancy of juvenile largemouth bass and bluegill, but it also showed that many other species of fish as well as all resident reptiles and amphibians were negatively affected. Without long term studies to follow this trend, we are left
again with an unclear perspective on how management actions are performing. For example, it is unclear whether the increased occupancy of some fishes actually benefited the population as a whole. We have no idea if increased occupancy of young largemouth bass and bluegill actually recruited into the adult population. We are also unsure how long it will take for some of the reptiles and amphibians to rebound because of limiting life history characteristics such as delayed sexual maturity, high juvenile mortality and/or low vagility. These traits are likely to prohibit immediate response to habitat change, it will take time to re-establish some populations but without long term data we have no idea how long that will take, if it happens at all. This study was a first step towards adopting a more holistic view to lake management, but still leaves questions and further work. Regardless of the direction in which management moves, it is realistic to expect the preservation of a reputable sport fishing industry as an important goal, because of the huge economic stimulus it brings to the state of Florida.

**Landscape Level Considerations**

Managing a lake should naturally extend beyond the lake itself, because neighboring environments influence the lake ecosystem and many animals utilize both habitats. The more wetlands become fragmented in the landscape, the more vulnerable aquatic and semi-aquatic animals become. Depending on the severity of the situation, if localized extinctions occur, re-colonization might not be possible (Brown and Kodric-Brown 1977, Semlitsch and Bodie 1998; Cushman 2006).

Part of this solution might be the maintenance of substantial wetland buffers that not only improve water quality, but also provide valuable habitats critical for life-history functions such as nesting, basking, foraging and refuge (Semlitsch and Bodie 2003). If development is continually permitted in buffer zones surrounding lakes, the success of many species will likely decline.
**Water Level Considerations**

Water level fluctuations have powerful impacts on vegetation and animal communities. Various studies have noted that water level fluctuations during critical bass spawning times can reduce hatching success and year-class strength (Mitchell 1982, Kohler et al. 1993, Waters and Noble 2004). Lake stage determines aquatic plant communities, which in turn influences fish, herp, bird and invertebrate community dynamics via changes in habitat structure, predator protection, prey availability and nesting and spawning substrate. Havens et al. (2005) documented bass recruitment changes resulting from lake stage and vegetation structure variations. Prolonged high water reduced plant biomass and coverage, coinciding with failed bass recruitment. Return of moderate water levels triggered a structurally diverse vegetation community that coincided with years of strong bass recruitment. Simply put, water level regulations that permit a diverse plant community, will promote a diverse fish community as well (Johnson et al. 2007).

In their review paper, Johnson et al. (2007), note that many lake and reservoir studies cite high water levels as drivers of increased year-class strength of largemouth bass. High water levels are associated with increased spawning substrate, protective cover and invertebrate production. In this case high water resulted in more inundated area, thus more available wetland habitat.

Additionally, fluctuating water levels have been implicated in failed recruitment. If fluctuation occurs during spawning season, this can reduce hatching success. Perhaps high and stable water levels during important stages such as spawning and nursery seasons are important to largemouth bass. Too much fluctuation in this time might impact successful reproduction, while maintaining high water too long can adversely affect plant communities. The effects of
plant communities cascade through the ecosystem affecting everything from water quality, dissolved oxygen, epiphytic and invertebrate communities and all higher fauna.

Many herp species would also be influenced by water stage as it relates to critical life stages. Johnson (2005) recommends that a slow decline in lake stage during the spring and summer followed by a reversal in the fall would most benefit amphiuma nesting success. For aquatic turtles Johnson suggests stable or slowly declining levels in the spring and summer, continuing into the fall. Any lake stage regime that results in significant increase in water levels over a short period of time prior to September would likely drown some nests. However variation in upland habitat and elevation of nesting sites are important to protecting some nests from inundation. Thus lake hydrology and upland habitat quality are two important stressors influencing turtle population, and most likely many other herpetofauna as well.

When necessary draw downs are performed, the welfare of several species ought to be considered. If draw downs happen when they are convenient for people, but at a time when animals are sensitive to change, unintended consequences such as failed reproduction are likely to occur.

Considering the powerful impact that water levels have on plant growth and animal communities, it should be more carefully considered. A comprehensive review of critical nesting, spawning, migration seasons of all aquatic vertebrates should be compiled and examined. Perhaps a suitable water schedule that would encourage favorable plant growth and animal use that is still compatible with flood control needs.

Vegetation Management Considerations

Prior to management, the dense stands of vegetation consisted mostly of pickerelweed and cattail, and were targeted for removal because past research indicated that it was poor quality habitat. While there is no doubt that extremely dense plant stands create difficult conditions such
as hypoxia and limited access to various faunal species, dense plants might have a place in the ecosystem. This study documented high occupancy of many reptiles and amphibians as well as a host of fish species. Other studies also provide evidence that dense plant stands provide pockets of refugia to small fishes (Miranda et al. 2000, Bunch et al. 2008). Differences among plant species and their structural complexity will affect the surrounding environment and habitat suitability. For example, Bunch et al. (2008) found highest species richness in cattail, pickerelweeed and torpedograss when compared to smartweed and water primrose. But even high density torpedograss and primrose communities provided habitat for high abundances of stress-tolerant fishes. The authors suggest that in the interest of maintaining high fish diversity, managers should prevent large areas of dense plant coverage, but recognize that these communities to provide important habitat for fishes as well as other wildlife such as birds, reptiles and amphibians.

Pelicice et al. (2005) found that in a tropical reservoir, high macrophyte density was beneficial to fish assemblages. They conclude that routine vegetation removal for multiple use purposes would be deleterious to the fishes dependent on dense macrophytes, reducing littoral fish density, biomass and species richness.

Some literature has referred to the usefulness of vegetation edges (Trebitz et al. 1997, Miranda and Hodges 2000). The mowing or scraping of narrow channels through vegetation could permit edge use by species that benefit from both open water habitat and vegetation. It might strike a balance between predator and prey interactions and provide oxygenated microhabitat amidst dense vegetation. Mowed channels would also benefit boater access. Experimenting with minimal plant removal techniques that achieve goals of boater access, while opening up some edge habitat might be useful.
Final Thoughts and Summary

The results from this study document only the immediate response of animals to habitat change. This is hardly conclusive information with which to recommend specific management actions. Lake managers and scientists working on these systems understand them best and the realities of what is possible, given the highly interdisciplinary nature of lake management. Continuing with studies that document faunal response to management will help shape our understanding of lake ecosystems, and support a adaptive and flexible outlook in light of new information.

Although the immediate effects documented here are important to understand, it does not paint a complete picture of long term response to management. As the vegetation continues to recover, faunal communities will change. Currently, most control sites have stabilized to communities similar to pre-management conditions, while scraped areas are still changing. Currently in these areas, carpets of \textit{V. americana} are dominating scraped areas (Brush et al. 2008). But it is unknown whether this community will continue to flourish or eventually be out-competed by \textit{P. cordata}.

Although such a large scale management project is not likely to take place again (FWC, personal communication), there have been valuable lessons learned. Management will always be needed in lakes because they have been so far removed from their natural state. This precludes restoration to historical conditions, and these highly urbanized and regulated lakes must be managed realistically. It might not be necessary to spend too much time, effort and money eradicating plant communities that are considered unacceptable and tend to return quickly after removal. Plus, in this study these communities were used by many animals that subsequently lost ground after the management. Probably no one would argue in favor of dense plants and
floating mats to dominate an entire lake, especially in small lakes where this could rapidly progress into a dried out depression.

By using drawdowns and minimal plant removal at appropriate times, managers might be able to establish lake habitats that are supportive of many species and also permit boater access. The realistic situation is that no one will ever be happy. Lakefront property owners will always despise dense plants and tall cattail stands, because it isn’t what a lake is “supposed to look like”. Many Florida lakes are naturally eutrophic which promotes these conditions and the continual influence of humans will continue to drive lakes into undesirable states. Fisherman do not appreciate dense hydrilla, but are tolerant of some because of its purported benefit to fishes. Nor do they like such dense emergent plants that they cannot access the shorelines. They do want some plants though, as most fishermen realize that where there are plants, there are fish. Herpetophiles will want “weedy mucky” habitats that support these critters. Birders who enjoy limpkins, rails, herons and gallinules will similarly want these vegetated lakes with some floating tussocks.

Lake managers are challenged to maintain healthy ecosystems that promote economically viable sportfishing, as many other species as possible and satisfying lake residents. All this in the face of climate change, continued urbanization and heightened demands on wetland ecosystems.
### APPENDIX A

**HERPETOFAUNA SPECIES LIST AND OCCUPANCY MODEL OUTPUTS**

Table A-1: Herpetofauna species list

<table>
<thead>
<tr>
<th></th>
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<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Amphiuma</td>
<td><em>Amphiuma means</em></td>
<td>✓</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Cottonmouth</td>
<td><em>Agkistrodon piscivorous conanti</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fl. banded water snake</td>
<td><em>Nerodia fasciata pictiventris</em></td>
<td>✓</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Fl. green water snake</td>
<td><em>Nerodia floridana</em></td>
<td>✓</td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Fl. snapping turtle</td>
<td><em>Chelydra serpentina osceola</em></td>
<td>×</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Fl. softshell turtle</td>
<td><em>Apalone ferox</em></td>
<td>×</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Leopard frog</td>
<td><em>Rana sphenocephala</em></td>
<td>×</td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Mud snake</td>
<td><em>Farancia abacura abacura</em></td>
<td>×</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Peninsula cooter</td>
<td><em>Pseudemys floridana peninsularis</em></td>
<td>×</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Pig frog</td>
<td><em>Rana grylio</em></td>
<td>✓</td>
<td>x</td>
<td>✓</td>
</tr>
<tr>
<td>Siren</td>
<td><em>Siren spp.</em></td>
<td>✓</td>
<td>x</td>
<td>✓</td>
</tr>
<tr>
<td>Stinkpot</td>
<td><em>Sternotherus odoratus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Striped crayfish snake</td>
<td><em>Regina alleni</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Striped mud turtle</td>
<td><em>Kinosternon baurii</em></td>
<td>✓</td>
<td>x</td>
<td>✓</td>
</tr>
</tbody>
</table>

A check indicates the species was captured enough to provide occupancy analyses, an x it was captured but not enough to provide analyses, no mark indicates the species was not captured at all in that year.
Table A-2: All occupancy models within 3 Δ AIC for each herpetofaunal species

<table>
<thead>
<tr>
<th>Year</th>
<th>Model</th>
<th>ΔAIC</th>
<th>wi</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Amphiuma</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>ψ(.), γ(.), p(temperature)</td>
<td>0</td>
<td>0.70</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Not captured</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Florida Banded Water Snake</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>ψ(.), γ(.), p(season)</td>
<td>0</td>
<td>0.52</td>
</tr>
<tr>
<td></td>
<td>ψ(temperature), γ(.), p(season)</td>
<td>1.53</td>
<td>0.24</td>
</tr>
<tr>
<td></td>
<td>ψ(lake stage), γ(.), p(season)</td>
<td>1.58</td>
<td>0.24</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Florida Green Water Snake</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>ψ(.), γ(.), p(lake stage)</td>
<td>0</td>
<td>0.84</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Pig Frog</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>ψ(.), γ(.), p(temperature)</td>
<td>0</td>
<td>0.46</td>
</tr>
<tr>
<td></td>
<td>ψ(.), γ(.), p(.)</td>
<td>0.32</td>
<td>0.39</td>
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<td></td>
<td>ψ(.), γ(.), p(lake stage)</td>
<td>2.16</td>
<td>0.15</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>ψ(.), γ(.), p(temperature)</td>
<td>0</td>
<td>0.79</td>
</tr>
<tr>
<td><strong>Siren</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>ψ(lake stage), γ(.), p(lake stage)</td>
<td>0</td>
<td>0.22</td>
</tr>
<tr>
<td></td>
<td>ψ(.), γ(.), p(lake stage)</td>
<td>0.33</td>
<td>0.18</td>
</tr>
<tr>
<td></td>
<td>ψ(.), γ(.), p(temperature)</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>ψ(lake stage &amp; temperature), γ(.), p(lake stage)</td>
<td>0.55</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>ψ(location), γ(.), p(lake stage)</td>
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<td>0.14</td>
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<td></td>
<td>ψ(temperature), γ(.), p(lake stage)</td>
<td>1.06</td>
<td>0.13</td>
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<td>ψ(.), γ(.), p(lake stage &amp; temperature)</td>
<td>2.19</td>
<td>0.07</td>
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<td>2005-2006</td>
<td>Inestimable</td>
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<tr>
<td>2006-2007</td>
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<td>0.85</td>
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<tr>
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<td>0.67</td>
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<td>ψ(lake stage), γ(.), p(lake stage)</td>
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<td>2005-2006</td>
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<td>0.45</td>
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<td>ψ(lake stage), γ(.), p(habitat)</td>
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<td>ψ(.), γ(.), p(lake stage)</td>
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<td>0.13</td>
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<td>2006-2007</td>
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<td>ψ(lake stage), γ(.), p(lake stage)</td>
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<td><strong>Striped Mud Turtle</strong></td>
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<td></td>
</tr>
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<td>2002</td>
<td>ψ(location), γ(.), p(.)</td>
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<td>0.68</td>
</tr>
<tr>
<td></td>
<td>ψ(.), γ(.), p(season)</td>
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<td>0.18</td>
</tr>
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<td>2005-2006</td>
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<td></td>
</tr>
<tr>
<td>2006-2007</td>
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<td>0.51</td>
</tr>
<tr>
<td></td>
<td>ψ(.), γ(.), p(temperature)</td>
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<td>0.40</td>
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</tbody>
</table>

Also indicated are the model weights (wi). Where there were not enough captures for estimates, it is indicated “inestimable”. Where the species was not captured at all that sample year, it is indicated “not captured”.
APPENDIX B
FISH SPECIES LIST AND OCCUPANCY MODEL OUTPUTS

Table B-1: Fish species list

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Armored catfish</td>
<td><em>Hoplosternum littorale</em></td>
<td>✓</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Black crappie</td>
<td><em>Pomoxis nigromaculatus</em></td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Bluegill</td>
<td><em>Lepomis macrochirus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Blue-spotted sunfish</td>
<td><em>Enneacanthus gloriosus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Blue tilapia</td>
<td><em>Oreochromis aureus</em></td>
<td></td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Bowfin</td>
<td><em>Amia calva</em></td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Chain pickerel</td>
<td><em>Esox niger</em></td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Chubsucker</td>
<td><em>Erimyzon spp.</em></td>
<td>✓</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Dollar sunfish</td>
<td><em>Lepomis marginatus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Flagfish</td>
<td><em>Jordanella floridæ</em></td>
<td>x</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Gar</td>
<td><em>Lepisosteus spp.</em></td>
<td>✓</td>
<td>x</td>
<td>✓</td>
</tr>
<tr>
<td>Golden shiner</td>
<td><em>Notemigonus crysoleucas</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Golden topminnow</td>
<td><em>Fundulus chrysaleus</em></td>
<td>x</td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Largemouth bass</td>
<td><em>Micropterus salmoides</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Mayan cichlid</td>
<td><em>Cichlasoma urophthalmus</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sailfin catfish</td>
<td><em>Pterygoplichthys spp.</em></td>
<td>x</td>
<td></td>
<td>✓</td>
</tr>
<tr>
<td>Redear sunfish</td>
<td><em>Lepomis microlophus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Redfin pickerel</td>
<td><em>Esox americanus</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sailfin molly</td>
<td><em>Poecilia latipinna</em></td>
<td>✓</td>
<td></td>
<td>x</td>
</tr>
<tr>
<td>Seminole killifish</td>
<td><em>Fundulus seminolius</em></td>
<td>x</td>
<td>✓</td>
<td>✓</td>
</tr>
<tr>
<td>Spotted sunfish</td>
<td><em>Lepomis punctatus</em></td>
<td>✓</td>
<td>x</td>
<td>x</td>
</tr>
<tr>
<td>Threadfin shad</td>
<td><em>Dorosoma petenense</em></td>
<td>x</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Warmouth</td>
<td><em>Lepomis gulosus</em></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
</tr>
</tbody>
</table>

A check indicates the species was captured enough to provide occupancy analyses, an x it was captured but not enough to provide analyses, no mark indicates the species was not captured at all in that year.
Table B-2: All occupancy models within 3 Δ AIC for each fish species

<table>
<thead>
<tr>
<th>Year</th>
<th>Model</th>
<th>ΔAIC</th>
<th>$w_i$</th>
</tr>
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<tbody>
<tr>
<td><strong>Armored Catfish</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(season)$</td>
<td>0</td>
<td>0.47</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Bluegill</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage &amp; temperature)$</td>
<td>0</td>
<td>0.66</td>
</tr>
<tr>
<td></td>
<td>$\Psi(lake stage), \gamma(\cdot), p(lake stage &amp; temperature)$</td>
<td>1.93</td>
<td>0.25</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(location)$</td>
<td>0</td>
<td>0.33</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(habitat)$</td>
<td>1.67</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>$\Psi(habitat), \gamma(\cdot), p(habitat &amp; location)$</td>
<td>2.30</td>
<td>0.10</td>
</tr>
<tr>
<td></td>
<td>$\Psi(habitat), \gamma(\cdot), p(location)$</td>
<td>2.31</td>
<td>0.10</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(temperature &amp; time)$</td>
<td>0</td>
<td>0.29</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(habitat), p(temperature)$</td>
<td>0.62</td>
<td>0.21</td>
</tr>
<tr>
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<td>$\Psi(habitat), \gamma(\cdot), p(temperature)$</td>
<td>1.03</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(habitat)$</td>
<td>2.00</td>
<td>0.11</td>
</tr>
<tr>
<td><strong>Blue-spotted Sunfish</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(location), \gamma(\cdot), p(temperature)$</td>
<td>0</td>
<td>0.80</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage)$</td>
<td>0</td>
<td>0.75</td>
</tr>
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<td></td>
<td>$\Psi(\cdot), \gamma(habitat), p(lake stage)$</td>
<td>2.22</td>
<td>0.25</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage)$</td>
<td>0</td>
<td>0.59</td>
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<tr>
<td></td>
<td>$\Psi(habitat), \gamma(\cdot), p(lake stage)$</td>
<td>2.72</td>
<td>0.15</td>
</tr>
<tr>
<td><strong>Chubsucker</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(temperature)$</td>
<td>0</td>
<td>0.34</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(.)$</td>
<td>1.43</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>$\Psi(lake stage), \gamma(\cdot), p(temperature)$</td>
<td>1.75</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage &amp; temperature)$</td>
<td>1.81</td>
<td>0.14</td>
</tr>
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<td>$\Psi(temperature), \gamma(\cdot), p(temperature)$</td>
<td>1.81</td>
<td>0.14</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Not captured</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Not captured</td>
<td></td>
<td></td>
</tr>
<tr>
<td><strong>Dollar Sunfish</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(.)$</td>
<td>0</td>
<td>0.2</td>
</tr>
<tr>
<td></td>
<td>$\Psi(temperature), \gamma(\cdot), p(temperature)$</td>
<td>0.08</td>
<td>0.19</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(temperature)$</td>
<td>0.33</td>
<td>0.17</td>
</tr>
<tr>
<td></td>
<td>$\Psi(lake stage &amp; temperature), \gamma(\cdot), p(temperature)$</td>
<td>0.62</td>
<td>0.14</td>
</tr>
<tr>
<td></td>
<td>$\Psi(lake stage), \gamma(\cdot), p(temperature)$</td>
<td>1.4</td>
<td>0.10</td>
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<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage)$</td>
<td>1.59</td>
<td>0.09</td>
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<tr>
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<td>$\Psi(\cdot), \gamma(\cdot), p(lake stage &amp; temperature)$</td>
<td>2.15</td>
<td>0.07</td>
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<tr>
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<td>$\Psi(lake stage &amp; temp), \gamma(\cdot), p(lake stage &amp; temp)$</td>
<td>2.61</td>
<td>0.05</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(season)$</td>
<td>0</td>
<td>0.63</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(temperature)$</td>
<td>0</td>
<td>0.52</td>
</tr>
<tr>
<td></td>
<td>$\Psi(habitat), \gamma(\cdot), p(temperature)$</td>
<td>0.66</td>
<td>0.38</td>
</tr>
</tbody>
</table>

Also indicated are the model weights ($w_i$). Where there were not enough captures for estimates, it is indicated “inestimable”. Where the species was not captured at all that sample year, it is indicated “not captured”.
<table>
<thead>
<tr>
<th>Year</th>
<th>Model</th>
<th>ΔAIC</th>
<th>wi</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Gar</strong></td>
<td><strong>Largemouth Bass</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
<td>0</td>
<td>0.69</td>
</tr>
<tr>
<td>2005-2006</td>
<td><strong>Inestimable</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\text{habitat}), \gamma(\cdot), p(\text{temperature})$</td>
<td>0</td>
<td>0.53</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\text{habitat}), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
<td>1.99</td>
<td>0.20</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature})$</td>
<td>2.72</td>
<td>0.14</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature} &amp; \text{ habitat})$</td>
<td>0</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{habitat})$</td>
<td>0.16</td>
<td>0.27</td>
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<tr>
<td></td>
<td>$\Psi(\text{habitat}), \gamma(\cdot), p(\text{temperature})$</td>
<td>0.41</td>
<td>0.24</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
<td>0</td>
<td>0.72</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature})$</td>
<td>2.99</td>
<td>0.16</td>
</tr>
<tr>
<td><strong>Redear Sunfish</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{season})$</td>
<td>0</td>
<td>0.68</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature})$</td>
<td>0.78</td>
<td>0.25</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{lake stage})$</td>
<td>1.95</td>
<td>0.14</td>
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<td>$\Psi(\cdot), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
<td>2.37</td>
<td>0.11</td>
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<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature})$</td>
<td>2.72</td>
<td>0.09</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{lake stage})$</td>
<td>0</td>
<td>0.34</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
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<tr>
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<td>0.15</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{habitats})$</td>
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<td>0.11</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\text{habitat}), \gamma(\cdot), p(\text{lake stage})$</td>
<td>2.91</td>
<td>0.08</td>
</tr>
<tr>
<td><strong>Sailfin Catfish</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>Not captured</td>
<td>0</td>
<td>0.36</td>
</tr>
<tr>
<td>2005-2006</td>
<td><strong>Inestimable</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\cdot)$</td>
<td>0</td>
<td>0.36</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{temperature})$</td>
<td>0.12</td>
<td>0.34</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{lake stage})$</td>
<td>1.99</td>
<td>0.13</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\cdot), \gamma(\cdot), p(lake \text{ stage} &amp; \text{ temperature})$</td>
<td>2.04</td>
<td>0.13</td>
</tr>
<tr>
<td><strong>Sailfin Molly</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(\cdot), \gamma(\cdot), p(\text{lake stage})$</td>
<td>0</td>
<td>0.53</td>
</tr>
<tr>
<td></td>
<td>$\Psi(\text{temperature}), \gamma(\cdot), p(\text{lake stage})$</td>
<td>1.9</td>
<td>0.21</td>
</tr>
<tr>
<td>Year</td>
<td>Model</td>
<td>ΔAIC</td>
<td>(w_i)</td>
</tr>
<tr>
<td>--------------</td>
<td>--------------------------------------------</td>
<td>------</td>
<td>---------</td>
</tr>
<tr>
<td></td>
<td>(\Psi(\text{lake stage}), \gamma(\cdot), p(\text{lake stage}))</td>
<td>1.97</td>
<td>0.20</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Seminole Killifish</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2005-2006</td>
<td>(\Psi(\text{habitat}), \gamma(\cdot), p(\text{season}))</td>
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<td>0.43</td>
</tr>
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<td></td>
<td>(\Psi(\cdot), \gamma(\cdot), p(\text{season}))</td>
<td>0.72</td>
<td>0.30</td>
</tr>
<tr>
<td></td>
<td>(\Psi(\cdot), \gamma(\cdot), p(\text{lake stage &amp; habitat}))</td>
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<td>0.16</td>
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<td>2006-2007</td>
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</tr>
<tr>
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<td>(\Psi(\cdot), \gamma(\cdot), p(\text{season}))</td>
<td>2.47</td>
<td>0.22</td>
</tr>
<tr>
<td></td>
<td>Spotted Sunfish</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>(\Psi(\cdot), \gamma(\cdot), p(\text{lake stage}))</td>
<td>0</td>
<td>0.53</td>
</tr>
<tr>
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<td>(\Psi(\text{lake stage}), \gamma(\cdot), p(\text{lake stage}))</td>
<td>1.84</td>
<td>0.21</td>
</tr>
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<td>(\Psi(\cdot), \gamma(\cdot), p(\cdot))</td>
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<td>0.13</td>
</tr>
<tr>
<td>2005-2006</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2006-2007</td>
<td>Inestimable</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>Warmouth</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>(\Psi(\text{temp}), \gamma(\cdot), p(\text{temperature}))</td>
<td>0</td>
<td>0.41</td>
</tr>
<tr>
<td></td>
<td>(\Psi(\cdot), \gamma(\cdot), p(\text{temperature}))</td>
<td>0.74</td>
<td>0.28</td>
</tr>
<tr>
<td></td>
<td>(\Psi(\cdot), \gamma(\cdot), p(\text{lake stage &amp; temperature}))</td>
<td>2.69</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>(\Psi(\text{lake stage}), \gamma(\cdot), p(\text{temperature}))</td>
<td>2.72</td>
<td>0.11</td>
</tr>
<tr>
<td>2005-2006</td>
<td>(\Psi(\text{habitat}), \gamma(\cdot), p(\text{lake stage}))</td>
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<td>2006-2007</td>
<td>(\Psi(\text{habitat}), \gamma(\cdot), p(\text{temperature}))</td>
<td>0</td>
<td>0.99</td>
</tr>
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</table>
## Table C-1: All occupancy models within 2 Δ AIC for both exotic and native species

<table>
<thead>
<tr>
<th>Year</th>
<th>Model</th>
<th>ΔAIC</th>
<th>wi</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Exotic Apple Snail</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>Not applicable</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(.), \gamma(.), p(location)$</td>
<td>0</td>
<td>0.64</td>
</tr>
<tr>
<td></td>
<td>$\Psi(habitat), \gamma(.), p(location)$</td>
<td>1.18</td>
<td>0.36</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(.), \gamma(habitat), p(temperature &amp; location)$</td>
<td>0</td>
<td>0.89</td>
</tr>
<tr>
<td><strong>Native Apple Snail</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>2002</td>
<td>$\Psi(.), \gamma(.), p(location)$</td>
<td>0</td>
<td>0.55</td>
</tr>
<tr>
<td></td>
<td>$\Psi(.), \gamma(.), p(location &amp; temperature)$</td>
<td>1.07</td>
<td>0.32</td>
</tr>
<tr>
<td>2005-2006</td>
<td>$\Psi(.), \gamma(.), p(lake stage)$</td>
<td>0</td>
<td>0.52</td>
</tr>
<tr>
<td></td>
<td>$\Psi(.), \gamma(.), p(lake stage &amp; temperature)$</td>
<td>0</td>
<td>0.40</td>
</tr>
<tr>
<td>2006-2007</td>
<td>$\Psi(.), \gamma(.), p(location)$</td>
<td>0</td>
<td>0.58</td>
</tr>
<tr>
<td></td>
<td>$\Psi(habitat), \gamma(.), p(location)$</td>
<td>1.29</td>
<td>0.31</td>
</tr>
</tbody>
</table>

Also indicated are the model weights ($w_i$)
APPENDIX D
FUTURE MONITORING RECOMMENDATIONS

For continued monitoring of the herpetofauna and fish communities, sampling in the same manner as in the last year (2006-2007), with 18 transects evenly distributed in the control, treatment and all-lake sites is recommended. The sampling effort should be adjusted, such that traps are in place only at specific times throughout the year, rather than being deployed year-round. This requires too much time and investment in trap maintenance and field personnel. If traps are active for about 7 days, this would be the approximate time period elapsed in one ‘season’ for this study. But this is not too important, as long as there are two or more consecutive trap checks (the more the better for increased accuracy). Traps should be checked every day, to reduce stress and mortality of trapped animals. Perhaps once per month or as little as 6 months spread throughout the year would be sufficient to encompass the extremes of the lake environment that affect detection probabilities. For example, trapping would ideally take place during cold winter weather with high lake stage, moderate weather and lake stage and very hot summer temperatures with lower lake stages. Although the protocol would be slightly different, it would still provide reliable occupancy estimates for comparison. In fact, many of the species will almost certainly be more abundant than they were in the two years following management, which increases detections, providing better precision estimates.
APPENDIX E
IMPORTANCE OF DETECTION PROBABILITY: SOME EXAMPLES

A famous fisheries saying is something to effect of, “studying fish is like studying trees, except that they move around and are invisible” (M. Rogers personal communication). This emphasizes why studying animal populations without considering detection probabilities is unwise (Mackenzie 2005, Mazerolle et al. 2007). Studies that report raw counts without considering detection probability are likely reporting results that reflect some random environmental or observer variable. This was part of the reasoning for using the program PRESENCE.

In this study we chose five covariates suspected of influencing the study animals. The three (continuous) environmental variables lake stage, temperature, and time were incorporated as well as two (categorical) habitat variables location on lake and habitat. As it turns out, most animals were influenced by temperature and/or lake stage. In fact when plotting detection probabilities against these two covariates, there were clear patterns. Such variation underscores their importance in animal studies. The following are just a few examples.
Figure E-1: Fish detection probabilities from 2002 that were influenced by temperature
Figure E-2: Fish detection probabilities from 2002 that were influenced by lake stage and temperature. The temperature curve does not correlate directly to the y-axis, it is superimposed onto the figure for reference.
Figure E-3: Bass detection probabilities from 2005-2006 that were influenced by temperature
Figure E-4: Fish detection probabilities from 2006-2007 that were influenced by lake stage and temperature
Figure E-5: Herpetofauna detection probabilities from 2002 that were influenced by lake stage
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BIOGRAPHICAL SKETCH

Melissa DeSa received two bachelor’s degrees: one in biological sciences from Brock University in St. Catherine’s, Ontario; and the other in wildlife biology from the University of Guelph in Guelph, Ontario. After receiving these degrees, she pursued field work in this area for a couple of years before beginning graduate school. This marks over 4 years of a greatly privileged opportunity and experience at the Florida Fish and Wildlife Cooperative Research Unit in Gainesville, Florida. Upon completion Melissa hopes to pursue a career involving the human dimensions of environmental decision-making.