EFFECTS OF COMMERCIAL GILL NET BYCATCH ON THE BLACK CRAPPIE FISHERY AT LAKE DORA, FLORIDA

By

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To my parents, Randy and Lynn Dotson; my grandmother, Jo Anne Dotson; and my sister, Jennifer Dotson: thank you so much for all of your love, support, and guidance. I would not be the person that I am today without you all.

To my late grandfather, Paul Dotson: you are dearly missed and I know that you are proud of my accomplishments.

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Commercial bycatch can potentially cause population-level effects and represents serious concerns for sustainability and efficiency of fisheries. A commercial gill net fishery for gizzard shad *Dorosoma cepedianum* took place during 2005 and 2006 at Lake Dora, Florida. The primary bycatch of the gill net fishery was reproductively mature black crappie *Pomoxis nigromaculatus*, which also support the primary sport fishery at the lake. I assessed total black crappie bycatch, mortality rates of black crappie entangled in gill nets, and quantified recreational fishing effort and harvest for 2005 and 2006, and estimated exploitation for the recreational and commercial (bycatch) fisheries in 2006. I utilized age-structured population dynamics modeling techniques to investigate potential population-level impacts of bycatch. Onboard observer data of commercial fishing activity showed that approximately 17,000 and 30,000 black crappie were captured in gill nets in 2005 and 2006, respectively. Estimates from a pen experiment revealed that about 30% and 47% of black crappie experienced 72-h mortality due to entanglement in gill nets in 2005 and 2006, respectively. Recreational exploitation (u_{rec}) was estimated to be 42% based on tag returns, and commercial exploitation (u_{com}) was estimated to be 16% based on the number of black crappie that died due to gill netting in 2006.

Simulations were performed from a stock reduction analysis (SRA) population dynamics model for three exploitation scenarios to investigate the potential of recruitment overfishing and simulations were performed from a yield-per-recruit model with varying exploitation rates to investigate the potential for growth overfishing. Results suggested that the current level of recreational exploitation is operating near a target SPR goal of 0.3 to 0.35 and additional exploitation from the recreational or commercial fishery could risk recruitment overfishing. Given the current vulnerability to harvest schedule growth overfishing is not of concern, however a shift in vulnerability towards smaller fish could increase the risk of growth overfishing. The greatest risk for recruitment overfishing via bycatch occurs when recreational exploitation is also high (e.g., this work). My study revealed a trade off, where potential benefits of biomanipulation via gizzard shad harvesting must be weighed against bycatch impacts to recreational fisheries.

CHAPTER 1
INTRODUCTION

Bycatch, the incidental catch of non-target species with fishing gear, occurs in almost all commercial fisheries, and has become a central resource management concern throughout the world (Diamond et al. 2000; Crowder and Murawski 1998; Pikitch et al.1998). Many studies have attempted to assess total bycatch in commercial fisheries (Hale et al. 1981; Hale et al. 1983; Renfro et al. 1989; Hale et al. 1996; Clark and Hare 1998; Pikitch et al. 1998; Stein et al. 2004), assess mortality of incidental bycatch (Hale et al. 1981; Hale et al. 1983; Clark and Hare 1998; Belda and Sanchez 2001; Beerkircher et al. 2002; Stein et al. 2004), and ultimately address population-level effects (Crouse et al. 1987; Mangel 1993; Crowder et al. 1994; Caswell et al. 1998; Diamond et al. 1999; Diamond et al. 2000; Tuck et al. 2001; Majluf et al. 2002). Prior to 1998, hypotheses about population-level impacts rarely had been tested (Crowder and Murawski 1998) and Diamond et al. (2000) noted that population-level effects of bycatch have been difficult to quantify.

Observations made on board commercial fishing vessels have estimated the proportion of total landings made up of bycatch and bycatch initial mortality rates (Hale et al. 1981; Hale et al. 1983; Hale et al. 1996; Clark and Hare 1998; Pikitch et al. 1998; Beerkircher et al. 2002; Stein et al. 2004). Hale et al. (1983) observed pound net fishing operations in the St. Johns River, Florida, and estimated game fish total bycatch and initial mortality with estimates of fishing effort, area fished, and game fish catch rate. Pikitch et al. (1998) used on-board observer data to estimate bycatch of Pacific halibut *Hippoglossus stenolepis* in Washington, Oregon, and California bottom trawl fisheries to test differences in catch rates of trawl types and time of year. Stein et al. (2004) tested for differences in total bycatch and mortality of Atlantic sturgeon *Acipenser oxyrinchus* among three gear types (trawl and two gill nets). Beerkircher et al. (2002)

quantified shark bycatch by species and initial mortality rates in the Southeast United States pelagic longline fishery with nine years of fisheries observer data. Onboard observations can provide useful information for measuring the proportion of total landings made up of bycatch and can provide estimates of initial mortality due to fishing.

Total bycatch mortality includes initial mortality occurring as part of the capture process and secondary mortality, which occurs following release from fishing gear. Initial mortality is most often calculated directly onboard as part of observer programs, whereas secondary mortality is estimated via pen studies or tagging programs. Total bycatch mortality is difficult to measure due to the long observation periods required after fish capture. Total mortality may result from chronic effects such as injury or infection, or increased vulnerability to predation (Crowder and Murawski 1998). Crowder and Murawski (1998) argued that secondary and total mortality should be considered in bycatch management, and appropriate survival studies should be conducted.

Total bycatch and bycatch mortality estimates provide useful information to aid in optimizing gear choice, fishing areas, and fishing seasons, but these estimates alone do not quantify population effects of bycatch. Catch of non-target species in fisheries can have implications at the population level (Crowder and Murawski 1998), and there are concerns about impacts to fish populations (Murray et al. 1992) and marine fauna such as sea turtles, seabirds, sharks, and mammals (Lewison et al. 2004). Methods to determine the population impacts of bycatch typically involve field estimates and population modeling. Age-and-stage-structured modeling techniques have been applied successfully to examine bycatch population implications for a variety of species including sea turtles (Crouse et al. 1987; Crowder et al. 1994), wandering albatross *Diomedea exulans* (Tuck et al. 2001), humboldt penguins *Spheniscus humboldti*

(Majluf et al. 2002), right whale dolphins *Lissodelphis borealis* (Mangel 1993), and harbor porpoises *Phocoena phocoena* (Caswell et al. 1998). Diamond et al. (1999) explored the population level effects of catch and bycatch on Atlantic croaker *Micropogonias undulatus* in the Gulf of Mexico and the Atlantic Ocean.

Lake Dora was recently selected by Florida resource management agencies for biomanipulation via intensive commercial fishing with gill nets. Gizzard shad Dorosoma cepedianum are an omnivorous fish with the potential to influence lake nutrient cycling. Gizzard shad can greatly reduce large crustacean zooplankton density (DeVries and Stein 1992; Stein et al. 1995) and can also consume benthic detritus when zooplankton resources are low (Stein et al. 1995; Irwin et al. 2003). Density and biomass of gizzard shad increase with trophic state, and gizzard shad often occupy the majority of total fish biomass in hypereutrophic systems (Bachmann et al. 1996; Allen et al. 2000). Because gizzard shad have the potential to influence zooplankton abundance and influence nutrient cycling between the sediment and the water column (Schauss and Vanni 2000; Schaus et al. 2002; Gido 2003), gizzard shad at Lake Dora were targeted for removal.

Gill nets are size selective and not species specific; thus, adult sport fish bycatch associated with the commercial gill net fishery for gizzard shad at Lake Dora is of concern to state agency scientists and anglers. Black crappie *Pomoxis nigromaculatus* comprise some of the most popular sport fisheries throughout North America (Hooe 1991; Allen and Miranda 1998) and represent the primary recreational fishery on Lake Dora, Florida (Benton 2005). Bycatch of black crappie is of concern to lake managers because significant bycatch mortality could have deleterious impacts on recreational fisheries. Thus, there is a need to evaluate whether bycatch could influence black crappie fisheries, which would elucidate policy trade-offs between

potential benefits of gizzard shad removal and impacts of commercial gill net bycatch on recreational fisheries.

The objectives of this study were to (1) estimate total black crappie bycatch in commercial gill nets, (2) estimate bycatch mortality (initial and secondary) from commercial gill nets on black crappie, (3) assess recreational fishing effort and harvest of black crappie, and (4) address population-level effects that bycatch could have on the black crappie fishery at Lake Dora. I assessed the population-level impacts of black crappie bycatch from the gizzard shad gill net fishery at Lake Dora, Florida by investigating the potential for recruitment overfishing via a stock reduction analysis (SRA) model and evaluating the potential for growth overfishing with a yield-per-recruit model. Growth overfishing occurs when fish are being harvested at an average size that is less than the size that produces maximum yield per recruit, and usually results from excessive effort and a selectivity schedule where small fish are vulnerable to harvest and not allowed to reach their maximum growth potential. Recruitment overfishing occurs when fishing mortality rates are so high that the adult population does not have the reproductive capacity to replace itself. Recruitment overfishing is less common but is of serious concern because it can lead to stock depletion and collapse. If selectivity schedules are skewed towards larger fish that have passed the age at sexual maturity recruitment overfishing may occur where growth overfishing is not a concern.

CHAPTER 2
METHODS

Study Site

Lakes Dora and Beauclair are part of the Upper Ocklawaha River Basin, located approximately 30 miles northwest of Orlando in central Florida (Magley 2003, Figure 2-1). Lakes Dora (1,774 ha) and Beauclair have a combined surface area of 2,211 ha and they are connected by a short open canal, commercial fishing for gizzard shad was permitted on both water bodies, and both systems have similar trophic status and fish communities (Table 2-1). I considered Lakes Dora and Beauclair as one water body for the purposes of this study and will refer to the system collectively as Lake Dora. Surface outflow from Lake Dora is through the Dora Canal into Lake Eustis (3,139 ha) (Figure 2-1).

Commercial Fishing

Permits were issued by the Florida Fish and Wildlife Conservation Commission (FWC) for 28 commercial fishers to remove gizzard shad from Lake Dora in 2005 and 2006. The fishery was regulated in an effort to minimize bycatch mortality as much as possible with the following restrictions. A maximum of two gill nets, not to total more than 1,097 meters could be used simultaneously by each boat, and gill net specifications were a minimum stretch mesh size of 10.2 cm. The maximum allowable length of one net was 549 meters, and nets were allowed 2 hours maximum soak time. There was no restriction on the maximum number of nets fished daily, as long as all other guidelines were followed. Floating and sinking gill nets were used. Commercial fishing was allowed only during daylight hours in open water areas at least 90 meters from shore during open seasons. Commercial fishers harvested gizzard shad, Florida gar *Lepisosteus platyrhincus*, longnose gar *Lepisosteus osseus*, blue tilapia *Oreochromis aurea*, and

the nonnative sailfin catfish *Liposarcus multiradiatus*. All other fish species caught in gill nets were required to be returned to the water immediately after removal from the nets.

Total Bycatch Assessment

Gill net operations during the gizzard shad removal were monitored by St. Johns River Water Management District (SJRWMD) observers. Monitoring was conducted at least twice per week during the commercial seasons and consisted of random observations of gill net fishing operations. Observers reported catch numbers, species composition, mesh size, net type (floating or sinking), and net length. An observation day consisted of at least six gill net sets. If there was no commercial gill net activity or weather prohibited observations, an attempt was made to average 12 gill net set observations per week and two sampling days per week over a one-month period. Subsamples of crappie bycatch were measured for total length (TL) weekly until a maximum of 100 fish per species was recorded each month. The first four weeks of fishing in 2006 required increased monitoring as follows; observations were conducted at least three days per week, at least 18 gill net sets were observed per week, and all black crappie encountered were measured until a maximum of 200 were recorded. The SJRWMD was required to follow these methods set forth in the sampling permit for the shad removal project issued by FWC.

Bycatch Mortality

To evaluate bycatch mortality of black crappie I collected fish from commercial fishing vessels as gill nets were being retrieved in both years. After black crappie were removed from gill nets by commercial fishers, I transferred the fish to a research vessel where they were measured to the nearest mm TL and placed in a 190 liter cooler with aerators used to maintain dissolved oxygen levels over 5 mg/L. Dissolved oxygen levels were recorded in the cooler to assure that they exceeded 5 mg/L at all times. Any initial mortality of fish from gill nets was recorded. I considered a fish to be alive when the net was pulled if there was opercular

movement (Kwak and Henry 1995). I recorded gill net mesh size and style (sinking or floating) for each sample fish were collected from.

I estimated secondary mortality of black crappie entangled in gill nets. Secondary mortality has been effectively measured for largemouth bass in live-release tournaments (Schramm et al. 1987; Kwak and Henry 1995; Weathers and Newman 1997; Neal and Lopez-Clayton 2001; Edwards et al. 2004) using pens to hold fish that were captured during hook-and-line tournaments. Holding time ranged from two to 21 days (Schramm et al. 1987; Kwak and Henry 1995; Weathers and Newman 1997; Neal and Lopez-Clayton 2001; Edwards et al. 2004), and Edwards et al. (2004) considered the three-day observation period adequate compared to other studies. Secondary mortality was measured using replicates of fish held in pens for 72 hours.

After fish were collected from the commercial fishers, they were transported to holding pens placed in the lake. The pens used were large hoop nets measuring 4.57 meters long, 1.22 meter diameter, and 50.8 mm stretch mesh nylon. A total of four hoop nets were used, and the nets were placed in three meters of water on a hard sand substrate bottom and marked with University of Florida research buoys. All net replicates were performed in the same area of Lake Dora during both commercial seasons. A minimum of 10 and maximum of 20 fish were placed in each pen. If a minimum of 10 fish could not be collected within 30 minutes of net pull time with the fishers, any fish that had been collected were transported to the pens to avoid further stress. All fish exhibiting opercular movement were placed in the pens for measures of secondary mortality. After the 72 hour treatment all fish were released, and any dead fish were measured to the nearest mm TL. Consistent with Hale et al. (1981) and Hale et al. (1983), we considered a fish to be dead if it was unable to swim away after 72 hours.

Pollock and Pine (2007) recognized the need for control replications in assessing delayed mortality for catch and release studies. It is not possible to obtain an unbiased estimate of fish captured in gill nets alone unless one assumes that there is no handling mortality (Pollock and Pine 2007). This is most likely not a reasonable assumption, hence control fish are necessary to account for handing mortality. Control fish were collected via electrofishing and hoop net gear during the 2006 season. Replicates of control fish placed in pens were used to account for potential mortality effects from transporting and holding fish. The same methods were applied during replications of control fish as described for treatment replications.

Water temperature and dissolved oxygen are critical factors influencing secondary mortality of fishes (Schramm et al. 1987; Gallinat et al. 1997; Weathers and Newman 1997; Wilde et al. 2000; Edwards et al. 2004). A temperature logger was placed at our pen holding site to record temperature every four hours during the course of the experiment. Dissolved oxygen (mg/L) was also measured each time a pen was set and retrieved, and in 2006 a dissolved oxygen logger was placed at my pen holding site to record dissolved oxygen levels every four hours during the course of the experiment to measure oxygen levels throughout the 72-hour treatment period.

Recreational Fishing Effort and Harvest

Roving creel surveys were conducted by the FWC on Lake Dora from November 2004 to June 2005, November 2005 to May 2006, and November 2006 to March 2007, respectively (three fishing seasons) to measure angling effort, harvest, and catch rates. Each survey was conducted on ten randomly selected days (six weekdays and four weekend days) for each 28-day period (Benton 2005). Using a randomly selected time, lake section, and direction of travel on each sample day, a clerk completed a survey of the entire lake by taking an instantaneous count of all anglers actively fishing on the lake to determine fishing effort (man-hour) (Benton 2005).

The clerk also interviewed anglers about their target species (if any species were specified by the angler), the number of each species caught, and how much time was spent fishing to determine fishing success (fish/hour) (Benton 2005). Catch from the angler interviews was extrapolated to angler effort estimates from the instantaneous counts to estimate total harvest at each lake in both years (Malvestuto et al. 1978; Malvestuto 1996; Benton 2005). Measurements of TL were recorded for a subsample of the black crappie catches during the three survey periods.

Tagging Study

A tagging study was conducted in 2006 for a direct estimate of exploitation from the recreational fishery (μ_{rec}). Lake Dora was divided into four areas and an approximately equal number of fish were tagged in each area. Area one encompassed Lake Beauclair, and areas two through four encompassed Lake Dora; the three areas of Lake Dora were the east lobe (2), middle lobe (3), and west lobe (4) (Figure 2-1). Fish were collected for tagging with a boat electrofisher, hoop nets, and an otter trawl. All fish captured were measured to the nearest mm TL, and fish 230 mm TL and greater were tagged and released into approximately the same area they were captured. Although there was no minimum size limit in place, I assumed that all fish 230 mm TL and greater had recruited to the fishery based on creel survey data.

All black crappie were tagged with dart tags with a yellow streamer containing information specifying the tag specific identification number, monetary reward value, and return address. Tags were inserted into the body of the fish below the dorsal fin rays using a hollow needle. When injected the streamer of each tag extended in a posterior direction at a 45° angle to the body. All black crappie were tagged from November 2005 to January 2006 to obtain an estimate of exploitation for the 2006 fishing season. All fish were single tagged with either a standard tag ($5) or a higher value reward tag ($50). The tagging reward study allowed for estimates of reporting rates (described below).

Age and Growth

Age and growth of black crappie at Lake Dora was estimated using fish collected from the recreational fishery from January through March 2005 to 2007, which is when black crappie angling effort peaks (Benton 2005; FWC 2005). Lake Dora has numerous fish camps where anglers clean harvested fish daily and these camps were the source of fish for age samples. Collecting recreationally harvested fish is an efficient way to gather age information and has been utilized for many marine species (Potts et al. 1998; Potts and Manooch 1999; Patterson et al. 2001; Fischer et al. 2004; Fischer et al. 2005), although like all sampling gears is subject to size and age selectivity.

Coolers with ice were placed at fish cleaning stations for three camps. Information signs were also posted at the fish cleaning stations explaining the purpose of the project. Some anglers may fish multiple lakes on a given day and thus, I asked anglers not to donate black crappie if they had fished more than one lake in an effort to assure all black crappie ages represented the correct population. Coolers were left for two to three days before retrieval. All black crappie collected from recreational anglers were brought back to the lab where they were measured to the nearest mm TL and sagittal otoliths were removed from ten randomly selected fish for each centimeter group. Because fish larger than 330 mm TL were rare, all black crappie greater than this size were aged.

Ages of fish collected from the recreational fishery were determined by counting annuli on whole otoliths with the aid of a dissecting microscope. The use of otoliths to determine ages of black crappie has been verified (Hammers and Miranda 1991; Ross et al. 2005). Two independent readers aged each fish. Schramm and Doerzbacher (1982) found that black crappie have relatively thin otoliths that had clearly visible bands present in patterns expected for annual marks. Older fish (fish showing four or more opaque bands) have thicker otoliths, and therefore

are more likely to have bands masked in whole view (Schramm and Doerzbacher 1982). Thus, any otoliths showing four or more opaque bands, and any otolith disagreements from whole view readings were sectioned for verification of aging accuracy. One otolith was sectioned transversally using a South Bay Technology, Inc. low speed diamond wheel saw. Two transverse sections, 0.5 mm wide, were cut from each otolith and mounted on a labeled glass slide using ThermoShandon Synthetic Mountant for reading. Two independent readers used a dissecting microscope to read the sections. A third independent reader reexamined all disagreements and the majority reading was recorded as number of annuli. Not all black crappie form new opaque bands on their otoliths at the same time during spring, although opaque bands on otoliths from all age classes should be formed by June 1st in Florida (Schramm and Doerzbacher 1982). I used an arbitrary birth date of June 1st, so that all fish collected prior to June 1st were assigned ages corresponding to the number of annuli observed plus one.

Analyses

Total Bycatch Assessment

I obtained estimates of total black crappie bycatch from the commercial fishery using a stratified sampling design (see Krebs 1999). Onboard observer data were stratified into three time strata (A, B, and C) for both commercial fishing seasons. The strata represented periods of high, moderate, and low fishing effort, and were grouped such that the variance of bycatch observed was homogeneous within and heterogeneous among strata. The total bycatch estimate and variance on this total were determined using the equations for a stratified design from Pollock et al. (1994):

$$\hat{X}_{ST} = N\overline{X}_{ST} \tag{2-1}$$

$$VAR(\hat{X}_{ST}) = N_h^{\,2} \times VAR(\overline{X}_{ST}) \tag{2-2}$$

where,

$\hat{X}_{ST}$ = total bycatch estimate,

N = number of total possible fishing days in a season,

$\overline{X}_{ST}$ = stratified bycatch mean per fishing day.

h = stratum number (A, B, C)

and,

N_h = total possible fishing days in stratum

Bycatch Mortality

I measured the mortality rate for each pen replication in each year as the number of dead black crappie observed per pen divided by the total number of black crappie held in each pen. I then estimated the annual mean bycatch mortality rate as the average mortality rate across all replications for each year, with uncertainty expressed as the standard error around the yearly means. Mean and variance were also estimated for control replications.

I used the annual mean bycatch mortality rate multiplied by our estimate of total bycatch for black crappie in each year to achieve total commercial fishing mortality of black crappie by year given by the equation:

$$GD = GC \times GM \tag{2-3}$$

where,

GD = estimated total number of black crappie that died from gill net mortality,

GC = estimated total number of black crappie caught by gill nets,

and,

GM = total gill net mortality rate.

Recreational Fishing Effort and Harvest

All data were entered and analyzed in a creel survey analysis program developed by FWC (version 2, Conner and Sheaffer 2000) and were stored in a Microsoft Access® database on an FWC regional server (Benton 2005). Data was lost overboard from one 28-day period in 2006. We approximated the missing time period in 2006 using the percentage of effort for that period during 2005, assuming that the percentage of effort during that period in 2005 would serve as the best model to reconstruct the missing data in 2006.

Tagging Study

Tag returns were adjusted for tag-related mortality, tag loss, and non-reporting prior to estimating exploitation. I assumed 5 – 10% tagging mortality and tag loss for all black crappie tagged. Reporting rates of higher value reward tags ($50) in 2006 were estimated based on a linear-logistic model created by Nichols et al. (1991):

$$\lambda_H = \frac{e^{(-0.0045+0.0283(H))}}{\left(1+e^{(-0.0045+0.0283(H))}\right)} \tag{2-4}$$

where,

H = the dollar value of higher value reward tags,

and,

λ_H = the reporting rate of tags from higher reward value fish.

The reward values (H) were converted from 2006 standards to the 1988 monetary equivalents based on the Consumer Price Index. The 1988 monetary equivalents used in equation 2-4 were $30.29 for $50 rewards (U.S. Department of Labor 2006). Reporting rate estimates calculated from equation 2-4 were most precise at higher reward values (Nichols et al. 1991) and thus, I used equation 2-4 to estimate reporting rates of high-reward tag fish and then estimated the reporting rate of standard tags based on the assumption that all tagged fish had an equal

probability of recapture regardless of reward value. Alternate methods for estimating reporting rate, such as those presented in Taylor et al. (2006) assume 100% reporting rate of higher value tags in order to estimate the reporting rate of standard tags. I felt that a $50 tag value was not sufficient to make the assumption that all higher value reward tags were returned.

I estimated the total number of high value reward tag fish caught in 2006 using the equation:

$$\hat{C}_H = \frac{R_H}{\lambda_H} \tag{2-5}$$

where,

$\hat{C}_H$ = estimated number of higher value reward tag fish caught,

and,

R_H = total number of tags returned in 2006 from fish tagged with a higher reward value.

I assumed that standard tags and higher reward value tags had an equal probability of capture by anglers and estimated the total number of standard tag fish caught in 2006 using the ratio:

$$\frac{\hat{C}_H}{T_H} = \frac{\hat{C}_S}{T_S} \tag{2-6}$$

where,

S = the dollar value of a fish tagged with a standard tag,

$\hat{C}_S$ = estimated number of standard tag fish caught,

T_S = original number of fish tagged with standard reward tags,

and,

T_H = original number of fish tagged with higher value reward tags.

I estimated the reporting rates of standard reward tags ($5) in 2006 using the equation

$$\lambda_S = \frac{R_S}{\hat{C}_S} \tag{2-7}$$

Reporting rate estimates for high-value reward tags were varied to evaluate how uncertainty in λ_H would influence the exploitation rate.

Estimates of exploitation for the recreation fishery (μ_{REC}) were estimated using the equation:

$$\mu_{REC} = \frac{(\hat{C}_S + \hat{C}_H)}{(T_S \times 1 - (TM + TL)) + (T_H \times 1 - (TM + TL))} \tag{2-8}$$

where TM = tagging mortality and TL = tag loss.

The instantaneous rate of fishing mortality for the recreational fishery (F_{rec}) was estimated using the equation:

$$F_{REC} = -LN(1 - \mu_{REC}) \tag{2-9}$$

Estimates of exploitation for the commercial fishery (μ_{COM}) could not be obtained directly from tagging data because a reliable reporting rate could not be calculated. There was evidence that vulnerability with fish size to gill nets was similar to recreational angling, but commercial fishers had an incentive not to return tags. Thus, I was unable to use Nichol's equation to estimate commercial reporting rate. To estimate commercial exploitation I first estimated the vulnerable black crappie population size with the equation:

$$\hat{N} = \frac{C_{REC}}{\mu_{REC}} \tag{2-10}$$

where,

$\hat{N}$ = the number of vulnerable black crappie in the population,

and,

C_{REC} = recreational catch from creel survey data.

I estimated the exploitation rate from the commercial fishery (μCOM) as:

$$\mu_{com} = \frac{GD}{\hat{N}} \tag{2-11}$$

The instantaneous fishing mortality for the commercial fishery (F_{com}) was estimated as:

$$F_{com} = -LN(1 - \mu_{COM}) \tag{2-12}$$

I simulated changes in F_{COM} by changing the gill net mortality rate (*GM*), which changed the number of black crappie that died from gill nets (*GD*). The instantaneous fishing mortality for the commercial and recreational fisheries were estimated with varying levels of reporting rates, tag loss, tagging mortality, recreational catch, and total gillnet bycatch mortality to evaluate uncertainty in F values for a range of input parameters.

Age and Growth

Data collected from the recreational fishery (carcasses and creel) was used to estimate growth rates for black crappie. I created an age-length key from a subsample of black crappie aged from recreationally harvested carcasses and assigned an age to each individual from the entire sample of carcasses and the recreational creel measurements in order to obtain age and size structure of the population. Mean-length-at-age (MLA) and its associated variance (σ2) were found by equations for fixed-length subsamples presented by DeVries and Frie (1996). I used the Von Bertalanffy growth model (Ricker 1975) to describe growth rates. Von Bertalanffy parameter estimates (L∞, k, and t0) were obtained using Procedure NLIN (SAS 9.1).

Population-Level Impacts of Exploitation

I used Microsoft Excel® to construct a stock reduction analysis (SRA) with stochastic recruitment (see Walters et al. 2006) in order to evaluate the potential of recruitment overfishing occurring at varying exploitation rates. The basic idea of an SRA is to construct an age-

structured population dynamics model that consists of leading parameters (e.g., Bo and recK in this study) that describe the underlying production and carrying capacity and subtract known removals from the population over time (Walters et al. 2006). When leading parameter estimates produce a stock size that is too low to have sustained historical catches, the model predicts that the population should have disappeared prior to today (Walters et al. 2006). When leading parameters estimates produce a stock size that is too high, it predicts too little fishing impact and a current population size that is much too large to fit recent estimates (Walters et al. 2006).

The SRA I created reconstructed the historic stock size of black crappie in order to match model predicted estimates of exploitation and vulnerable biomass in 2006 to empirical estimates of exploitation and vulnerable biomass in 2006, given estimates of the leading parameters Bo and recK. Typically the leading parameter Bo is a measure of vulnerable biomass in the unfished condition. However, in this study Bo represents an estimate of vulnerable biomass far enough back in time to achieve a stable age distribution in the simulated population prior to this study (2005). The leading parameter recK is the Goodyear recruitment compensation ratio (Goodyear 1980) and is a measure of the juvenile survival at extremely low stock size relative to juvenile survival in the unfished condition. The parameter recK examines relationships between maximum recruitment at low stock size and the density dependence of recruitment at high stock size or the unfished condition (Goodwin et al. 2006). The two leading parameters are correlated in the sense that a lower Bo and higher recK can produce the same stock size as a higher Bo and lower recK. SRA models often have an exorbitant amount of combinations of Bo and recK that can explain the same stock size. The best combination of recK and Bo chosen must be supported statistically and biologically so that the parameter estimates are logical.

My empirical estimate of vulnerable biomass in 2006 in the fished condition was estimated as the vulnerable biomass per acre times the surface area (acres) of Lakes Dora and Beauclair combined. Vulnerable biomass per acre was estimated as the vulnerable number of black crappie per acre ($\frac{\hat{N}}{acres}$) times the average weight of a vulnerable black crappie, where the average weight of a vulnerable black crappie was estimated using a standard weight equation for black crappie (Anderson and Neumann 1996) with an average length of vulnerable black crappie harvested in 2006 (given from carcass and creel measurements). My empirical estimate of exploitation in 2006 was estimated for the recreational and commercial fisheries using equations 2-8 and 2-11, respectively.

I solved for my leading parameters (Bo and recK) by fitting the model predicted values of vulnerable biomass and exploitation in 2006 to empirical estimates in 2006 given by the log likelihood of the lognormal distribution:

$$MLE = -\ln((\ln(06predu_{total}) - \ln(06estu_{total}))^2 + (\ln(06estVB) - \ln(06predVB))^2) \qquad (2\text{-}13)$$

where,

MLE = the maximum likelihood estimate,

$06predu_{total}$ = 2006 model predicted estimate of total exploitation,

$06estu_{total}$ = 2006 empirical estimate of total exploitation,

06estVB = 2006 empirical estimate of vulnerable biomass (kg),

and,

06predVB = 2006 model predicted estimate of vulnerable biomass (kg).

I used Excel® table function to construct a maximum likelihood profile for a range of Bo and recK values that made sense biologically in order to determine combinations of parameter estimates that were supported statistically. Considering a review of maximum reproductive rates

of fish at low population sizes by Myers et al. (1999), black crappie most likely have a recK value between five and 20 based on fish species with similar life history characteristics. Estimates of Bo were considered from 70,000 to 100,000 kg, which were supported by my empirical estimates of adult fish density and fishing mortality rates.

When solving for leading parameter estimates, my model was very sensitive to starting values because of the correlation between leading parameters and multiple possible combinations. Thus, I was not able to solve for Bo and recK simultaneously. This phenomenon is very common in SRA model fitting. Therefore, I fixed Bo and solved for recK, because Bo exhibited much less variability than recK in the maximum likelihood profile and I had data for black crappie at Lake Dora that supported my estimate. Once reasonable parameter estimates were obtained the model was used to predict how the black crappie stock would respond in the future under different scenarios of exploitation. The output metrics of interest were vulnerable biomass (kg), total harvest (numbers) and weighted transitional spawning potential ratio (SPR).

The SRA required estimates of mean length at age, weight at age, fishing and natural mortalities, fecundity, and a vulnerability to harvest schedule in order to function. Fishing mortalities were separated into F_{REC} and F_{COM}, as described above. Estimates of total length-at-age were obtained from the Von Bertalanffy growth model and age specific weight was calculated using a standard weight equation for black crappie (Anderson and Neumann 1996). Equal vulnerability schedules were assumed for the recreational and commercial fisheries, based on the length frequencies from the recreational and commercial fisheries. Vulnerabilities at age were estimated using a cumulative normal distribution, which predicted expected catches at age in a yield-per-recruit model simulation that approximated the observed age structure of the catch. Fecundity was calculated as the weight at age minus weight at maturity (W_{mat}). Walters et al.

(2007) noted that fecundity is typically proportional to body weight above the weight at maturity. Weight at maturity was assumed to be the weight predicted at age 2, given that black crappie mature at approximately age 2 in this system (FWC 2005).

Survivorship at age in the unfished condition ($Survivorship0_a$) was calculated as survivorship in the previous year multiplied by survivorship in the absence of fishing (S_0). The instantaneous rate of natural mortality (M) was assumed to be 0.4 for all simulations, which is similar to values found in a review of black crappies (*Pomoxis spp.*) from Allen et al. (1998). Survival from natural mortality was found by $S_0 = e^{-M}$. Survivorship at age a in the fished condition ($SurvivorshipF_a$) was calculated as:

$$SurvivorshipF_a = SurvivorshipF_{a-1} \times S_0 \times \left(1 - \mu_{total} \times vul_{a-1}\right) \tag{2-14}$$

where survivorship at age one was assumed to be 1, the first age in the model.

Expected numbers were assumed to change over a ages and t years according to the survival equation (Walters et al. 2006):

$$N_{a+1,t+1} = N_{a,t} \times S_0 \times \left(1 - vul_{a,t} \times u_{total,t}\right) \tag{2-15}$$

I used an accounting scheme with 8 ages from 1961 – 2050 (N = 90). Expected numbers at age in the initial year were calculated as:

$$N_{a,t=1} = R_0 \times \sum_a survivorship0 \tag{2-16}$$

where Ro is the recruitment abundance in the unfished condition estimated as:

$$R_0 = \frac{B_0}{\Phi_{vb0}} \tag{2-17}$$

The Botsford incidence function for vulnerable biomass per recruit in the unfished condition was calculated as (Box 3.1, Walters and Martell 2004):

$$\Phi VB_0 = \sum_a wt_a, vul_a, survivorship0_a \tag{2-18}$$

Vulnerable biomass was determined annually with the equation:

$$\hat{B}_t = \sum_a N_{a,t}, vul_a, wt_a \tag{2-19}$$

The model required exploitation ($\mu_{total,\,t}$) and recruitment time series for all years after 1961. For each year the total exploitation rate was estimated as:

$$\mu_{total,t} = \frac{HARV_{total,t}}{\sum_a N_{a,t} vul_a} \tag{2-20}$$

Total harvest was estimated from historical creel data from 1977 to 1981 and from creel and commercial landings data in 2005 and 2006. Logical estimates of total harvest were simulated for the remaining years from 1961 to 2006. For future projections, estimates of exploitation were assumed under different fishing scenarios and total harvest estimates were calculated as:

$$HARV_{total} = \hat{N} \times \mu_{total,t} \tag{2-21}$$

This allowed the model to explore a range of assumed exploitation rates in the future and determine the expected vulnerable biomass, total harvest, and SPR given an exploitation rate.

Recruitment rates were predicted from estimates of annual egg production (Et) as:

$$E_t = \sum_a N_{a,t}, fec_a \tag{2-22}$$

using a Beverton and Holt stock-recruit relationship with recruitment variability of the form of the relationship (Walters et al. 2006):

$$N_{1,t+1} = \frac{\alpha E_t}{1 + \beta E_t} \times rand_t \tag{2-23}$$

where the alpha and beta Beverton and Holt parameters are described by the relationships:

$$\alpha = recK \times \frac{R_0}{E_0} \tag{2-24}$$

$$\beta = \frac{recK - 1}{E_0} \tag{2-25}$$

Variability around recruitment at time t ($rand_t$) was accounted for with a random number that was determined with PopTools in Microsoft Excel® by using a log normal distribution with a mean of 1.0 and recruitment coefficient of variation of 0.4. Allen (1997) observed black crappie recruitment coefficient of variation values ranging from 0.55 to 0.84 for 6 populations in Southeast and Midwest reservoirs, but there is evidence that recruitment variation in this system is considerably lower based on age-0 black crappie catch rates in bottom trawls (M. Hale, FWC, unpublished data).

Recruitment variability was added to the model simulations for future projections once estimates of the leading parameters were obtained via equation 2-13 in order to explore how abundance, catch, and spawning potential ratio varied through time with different exploitation rates. A weighted transitional SPR was used as a biological reference point to investigate the potential for recruitment overfishing at various exploitation scenarios. A weighted transitional SPR allows fishing mortality to vary by age and year and accounts for changes in the numbers at age over years. The SPR was estimated with the equation:

$$SPR_{t+1,2,3...89} = \frac{\sum_a N_{a,t+1,2,3...89}, fec_a}{\sum_a N_{a,t=1}, fec_a} \tag{2-26}$$

I determined the uncertainty in my terminal year SPR (2050) by using Monte Carlo analysis with 1,000 iterations to determine a terminal year mean SPR and 95% confidence limits around the mean. The same methods were applied to total harvest and vulnerable biomass estimates. I also used Monte Carlo analysis with 100 iterations to determine mean annual SPR values and 95%

confidence intervals for the entire model time series to show how the SPR would be expected to vary with variation in recruitment.

Future projections were simulated from 2007 through the terminal year 2050 under three exploitation scenarios; (1) $\mu_{total} = 0.42$, (2) $\mu_{total} = 0.51$, and (3) $\mu_{total} = 0.60$. Exploitation scenario one was chosen because it was the empirical estimate of μ_{rec} in 2006, scenario two was chosen because it was the empirical estimate of μ_{total} in 2006 and scenario three was chosen as an arbitrary increase in exploitation either due to recreational fishing, bycatch mortality, or both. The model simulations examined the three different exploitation scenarios and the implications they have on black crappie abundance, total harvest, and SPR if they were sustained through the terminal year 2050.

In order to investigate the potential for growth overfishing, I constructed a yield-per-recruit model in Excel®. Yield-per-recruit (kg) was determined as:

$$YPR = \Phi VB_F \times u_{total} \tag{2-27}$$

where,

The Botsford incidence function for vulnerable biomass per recruit in the fished condition (ΦVB_F) was calculated as (Box 3.1, Walters and Martell 2004):

$$\Phi VB_F = \sum_a wt_a, vul_a, survivorshipF_a \tag{2-28}$$

To investigate if growth overfishing was a concern I used Excel® table function to profile *YPR* values at total exploitation (μ_{total}) scenarios ranging from 0.2 to 1.0.

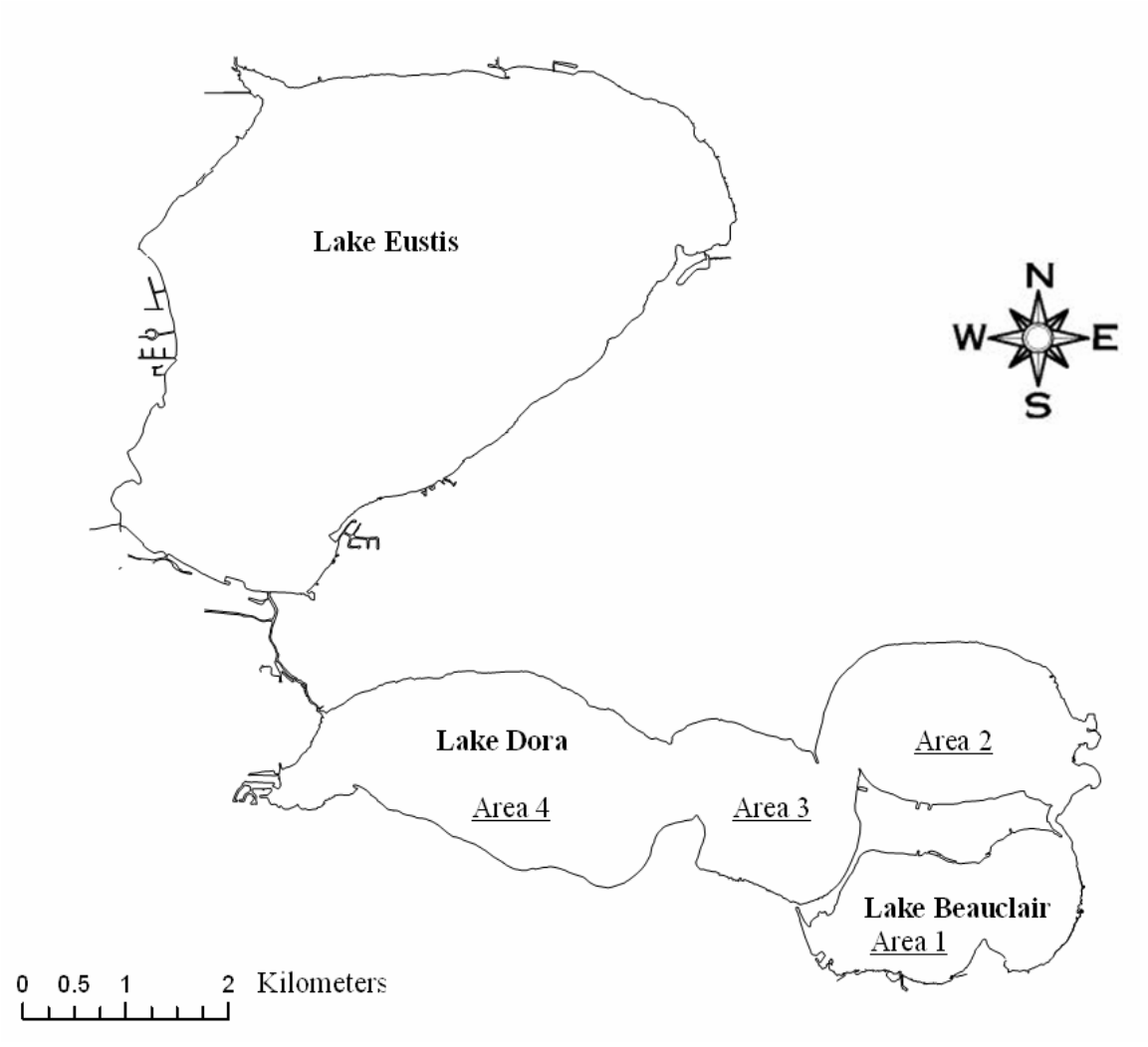

Figure 2-1. Map of Lakes Eustis, Dora, and Beauclair located in the Upper Ocklawaha River Basin in Lake County, Florida. Areas 1 – 4 represent designated capture and release areas for tagging study.

Table 2-1. Mean water quality parameters for Lakes Dora (east and west) and Beauclair (Florida LAKEWATCH 2004). Water quality parameters include total phosphorous (TP (µg/L), total nitrogen TN (µg/L), chlorophyll CHL (µg/L), secchi depths (meters), and trophic state and reflect the annual average for 2004.

LAKE	TP (µg/L)	TN (µg/L)	CHL (µg/L)	SECCHI (meters)	Trophic State
Dora east	55	2941	102.7	0.4	hypereutrophic
Dora west	50	2889	99.4	0.43	hypereutrophic
Beauclair	81	2971	100.5	0.4	hypereutrophic

*Trophic state based on Forsburg and Ryding (1980).

CHAPTER 3
RESULTS

Commercial Fishing

Commercial fishing occurred from March 1 to April 22, 2005 and from January 3 to March 28, 2006. Fishing was not permitted until March 1, 2005 because pre-harvest data were being collected for the gizzard shad population. Generally, there were two permitted fishermen per fishing vessel; there was a maximum of 16 vessels and a minimum of 1 vessel per fishing day during the 2005 and 2006 commercial fishing seasons. Total commercial effort was 258 boat days in 2005 and 251 boat days in 2006 (Figure 3-1) with an average of six boats per fishing day in 2005 and five boats per fishing day in 2006.

Total Bycatch Assessment

Black crappie bycatch was higher in 2006 than 2005 (Table 3-1). For 2005, there were a total of 487 black crappie observed during gillnet operations, 294 in stratum A (March 1 to March 14), 156 in stratum B (March 15 to Mar 31), and 37 in stratum C (April 1 to April 22). The average total daily bycatch per stratum ($\bar{x}_h$) was 595, 488, and 26 for strata A, B, and C, respectively. The total bycatch estimate ($\hat{X}_{ST}$) for 2005 was 17,199 black crappie and the 95% confidence intervals were 8,777 to 25,622. For 2006, there were a total of 2,109 black crappie observed during gillnet operations, 1,375 in stratum A (January 3 to January 31), 545 in stratum B (February 1 to February 28), and 189 in stratum C (March 1 to March 28). The average total daily bycatch per stratum was 979, 498, and 265 for strata A, B, and C, respectively. The total bycatch estimate ($\hat{X}_{ST}$) for 2006 was 30,258 black crappie, and the 95% confidence intervals were 19,048 to 41,469. Total daily bycatch of black crappie is reported in Figure 3-1 for days with onboard observer data in 2005 and 2006.

Bycatch Mortality

I conducted 17 pen replications from March 1 to April 8 during the 2005 commercial gill net season, and 23 pen replications from January 3 to March 15 during the 2006 season. Six control replications were made with fish caught in hoop nets, and four pen replications were made with fish caught with electrofishing gear in 2006 from January 13 to January 29. In 2005, bycatch mortality rates ranged from 0 to 0.75 during the treatment period with a mean of 0.31 (GM_{2005}) and a standard error of 0.06. In 2006, bycatch mortality rates ranged from 0.05 to 1 during the treatment period with a mean of 0.47 (GM_{2006}) and a standard error of 0.07. In 2006, control replications of fish collected with hoop nets (n = 6) ranged in mortality from 0 to 0.35 during the treatment period with a mean of 0.10 and a standard error of 0.05; control replications of fish collected with electrofishing gear (n = 4) had zero mortality. Results are summarized in Table 3-2. Estimates of bycatch mortality were not adjusted for pen related mortality due to low mortality estimates from control replicates.

I combined the mortality estimation and total bycatch estimates to estimate the number of black crappie deaths via bycatch each year. The estimated mean number of black crappie that died from gill net mortality in 2005 (GD_{2005}) was 5,332 with a range from 2,194 to 9,480 considering the range in estimates of *GM* and *GC*. The mean number of bycatch deaths in 2006 (GD_{2006}) was estimated at 14,221 with a range of 7,619 to 22,393 given the range in estimates in *GM* and *GC*.

Recreational Fishing Effort and Harvest

Comparison of the existing creel survey data at the lake suggest that recreational fishing effort and harvest have increased at Lake Dora. The annual fishing effort for black crappie at Lake Dora historically (survey data from 1977 to 1981) ranged from 14,208 to 26,233 hours constituting 25 to 39% of total angling effort (Benton 2005), and catch ranged from 16,603 to

41,745 black crappie per year (Benton 2005). The current surveys were only during the peak fishing season from November 2004 to June 2005, November 2005 to May 2006, and November 2006 to March 2007. Directed black crappie effort ranged from approximately 27,000 to 29,000 hours and harvest ranged from about 32,000 to 39,000 from 2004/2005 through 2006/2007 (Figure 3-2). Black crappie angling effort accounted for 80 to 94% of the total fishing effort for the three survey periods. No standard error could be calculated for the estimates from the 2005/2006 survey period because of missing data for one 28-day period that was estimated by substituting the mean value of fishing effort from the same time period the previous year.

Tagging Study

Tagging was conducted from November 3, 2005 to January 13, 2006 during sixteen sampling trips at Lakes Dora and Beauclair. A total of 514 black crappie were single-tagged with standard and higher reward floy tags, 197 fish were captured with electrofishing gear (38%), 214 fish were captured with hoop nets (42%), and the remaining 105 fish were captured with an otter trawl (20%). Totals of 125, 118, 133, and 132 fish were tagged in areas 1 through 4, respectively (tagging location of six fish were not recorded). A total of 413 black crappie were tagged with $5 standard reward tags and 101 black crappie were tagged with $50 higher value reward tags.

A total of 69 tags were returned, 40 $5 tags (10% of available $5 reward tags – 34 from recreational anglers and six from commercial fishers) and 29 $50 tags (29% of available $50 reward tags – 27 from recreational anglers and two from commercial fishers); recreational anglers accounted for 88% of total tag returns (61 of 69 returns) and commercial fishermen only accounted for 12% of total tag returns (8 of 69 returns). All tags were recaptured from December 7, 2005 to April 7, 2006, and recapture location was obtained from 55 of the 69 returned tags. We received six returns from area 1 (11%), nine returns from area 2 (16%), nine

returns from area 3 (16%), 23 returns from area 4 (42%), and eight returns from outside our study area in adjoining canals (15%). Although 15% percent of tag returns were from outside the study area in adjoining canals, all canals had locks that prevented fish escapement from the system.

Estimates of exploitation for the recreational fishery included adjustments for tag loss, tagging mortality, and reporting rate. Tag loss and tagging mortality were simulated at values from 5 to 10%. I assumed 5% tag loss and tagging mortality for the average estimate of exploitation for model simulations; Miranda et al. (2003) estimated tag loss for black and white black crappie to be 4.6% within 24 hours of tagging using t-bar tags, and there was a significant effect of time on tag loss. Henry (2003) estimated tag loss for largemouth bass to be approximately 5% using dart tags. I felt that 5% tag loss was a reasonable estimate, based on the short amount of time between tagging and recaptures and results from other studies. Miranda et al. (2003) estimated tagging mortality for black and white black crappie to be 11% (SE = 7.2%) for fish captured with electrofishing gear and trap nets. Henry (2003) estimated tagging mortality for largemouth bass to be 0% for fish collected with electrofishing gear and hook-and-line. Results from control replications of black crappie greater than 230 mm TL captured with hoop nets and electrofishing gear on Lake Dora (not tagged) had a mortality rate of 10% and 0%, respectively, and control replicates of black crappie greater than 180 mm TL captured with an otter trawl (pelvic fin clip) at Lake Jeffords, Florida had a mortality rate of 1% (G. Binion, UF, unpublished data). I felt that 5% tagging mortality was a reasonable estimate, based on our control replications of fish captured with hoop nets, an otter trawl, and electrofishing gear, and results from similar studies. The expected reporting rate of tags from higher value reward tag

fish (λ_H) was 70% (H = \$50) based on equation 2-4, and the expected reporting rate of standard tags was 22% based on equation 2-7 (Table 3-3).

The recreational exploitation rate (μ_{REC}) was 42% (TM = 0.05, TL = 0.05, λ_H = 0.7), the instantaneous rate of fishing mortality for the recreational fishery (F_{rec}) was 0.55. The commercial exploitation rate (μ_{COM}) was 16%, and the instantaneous rate of fishing mortality for the commercial fishery (F_{com}) was 0.17. I simulated a range of higher value reward tag reporting rates from 0.5 to 1.0 by intervals of 0.1 and tag loss/tagging mortality from 5 to 10% to analyze the effects of reporting rate on exploitation (Table 3-3). Lower reporting rates and higher tag loss/tagging mortality increase estimates of recreational exploitation and higher reporting rates and lower tag loss/tagging mortality decrease estimates of recreational exploitation. I simulated a range of the total number of black crappie that died from gill net mortality in 2006 (GD_{2006}) from 7,000 to 22,000, and the number of black crappie harvested in the recreational fishery in 2006 from 25,000 to 39,000 to evaluate effects on the instantaneous rate of fishing mortality for the commercial fishery (F_{com}). As expected, F_{com} values were highest at low recreational catch and high gillnet deaths, and lowest at high recreational catch and low gillnet deaths.

Age and Growth

A total of 882, 664, and 723 black crappie were collected and measured from the recreational fishery (whole sample – carcasses and creel) in 2005, 2006, and 2007, respectively. Sub-samples of carcasses (N = 183, 158, and 153 in 2005, 2006, and 2007) ranging from approximately 18 to 37 cm TL were analyzed to determine age annually. The size and age frequencies from the recreational catch (whole sample) in 2005, 2006, and 2007 are reported in Figures 3-3 and 3-4. Ages ranged from 2 to 8 years old for all three years. Mean length-at-age and associated variance and growth for the whole sample for each year were determined. Mean length-at-age and growth were similar for black crappie in all years. Results from 2006 were

used in model simulations and are reported in Figure 3-5. Ages were applied to 145 and 362 black crappie collected from the commercial gillnet fishery in 2005 and 2006, respectively. The size and age frequencies from the commercial bycatch in 2005 and 2006 are shown in Figures 3-4 and 3-5.

Age-structured Population Model Simulations

Estimates of historical harvest, vulnerable biomass, and exploitation from 1961 to 2006 are presented in Figure 3-6. Values of the total number of black crappie harvested from 1977 to 1981 were from historical creel data collected by FWC, values of harvest in 2005 and 2006 were estimates of total harvest from the commercial (estimated from onboard observations) and recreational fishery (estimated from creel survey) combined, and the remaining years were logical estimates of total harvest based on limited creel survey data. The maximum likelihood profile for Bo and recK is presented in Figure 3-7. I simulated a range of Bo values from 70,000 to 100,000 kg and a range of recK values from 5 to 20. Given the life history and known population characteristics of black crappie in Lake Dora, the ranges of Bo and recK values that were simulated include the most likely range of logical possibilities.

Based on the maximum likelihood profile a Bo estimate of 80,000 kg is supported statistically and is biologically realistic given my estimates of stock size and exploitation. Thus, I fixed Bo at 80,000 kg and used equation 2-13 to solve for a recK, resulting in an estimate of 15.2. The maximum likelihood estimate occurred at a Bo of 78,000 kg and a recK of 20; however, I felt that the *MLE* was not the true best fit because it occurred at the maximum recK in the likelihood profile. The model fit the recK value at the highest possible value it was restricted to resulting in estimates that were not biologically reasonable. After model fitting with my best parameter estimates, my predicted and empirical estimates of exploitation were 0.51 in 2006, and

the model predicted vulnerable biomass in 2006 approximated my empirical estimate (Table 3-4).

Future simulated exploitation rates influenced the model predicted estimates of total harvest, vulnerable biomass, and SPR (Table 3-5). Mean total harvest slightly increased as exploitation increased in simulations; however, mean vulnerable biomass decreased with increases in exploitation. The mean weighted transitional SPR in the terminal year decreased from 0.32 (scenario one) to 0.19 (scenario three). The SPR target goal for most fish species is approximately 0.3 to 0.4, used as a biological reference point where values below the target goal increase the likelihood of recruitment overfishing (Goodyear 1993; Clark 2002). The terminal year mean weighted transitional SPR was operating near the target goal of 0.3 to 0.35 at the levels of exploitation found in 2006, and model simulations predicted that increased exploitation may cause concern of recruitment overfishing. At the highest exploitation rate simulated, the mean weighted transitional SPR was predicted to be well below the target goal (Table 3-5). Results for the annual weighted transitional SPR values with recruitment variability (0.4) are reported for the entire model time series from Monte Carlo analysis with 100 iterations to show how recruitment variation would influence SPR values (Figure 3-8).

Results from yield-per-recruit model simulations are presented in Figure 3-9. The YPR values exhibited an asymptotic relationship with exploitation, indicating that with the current vulnerability schedules the black crappie fishery is not likely to exhibit growth overfishing. The maximum YPR value was 0.13 occurring at a total exploitation rate of 1. Black crappie were not fully vulnerable to either recreational fishing or commercial bycatch until age four, and they become reproductively mature at age two, which allows enough reproduction to prevent growth

overfishing. However, at extremely high exploitation rates a shift in the size structure toward smaller, younger fish would be anticipated.

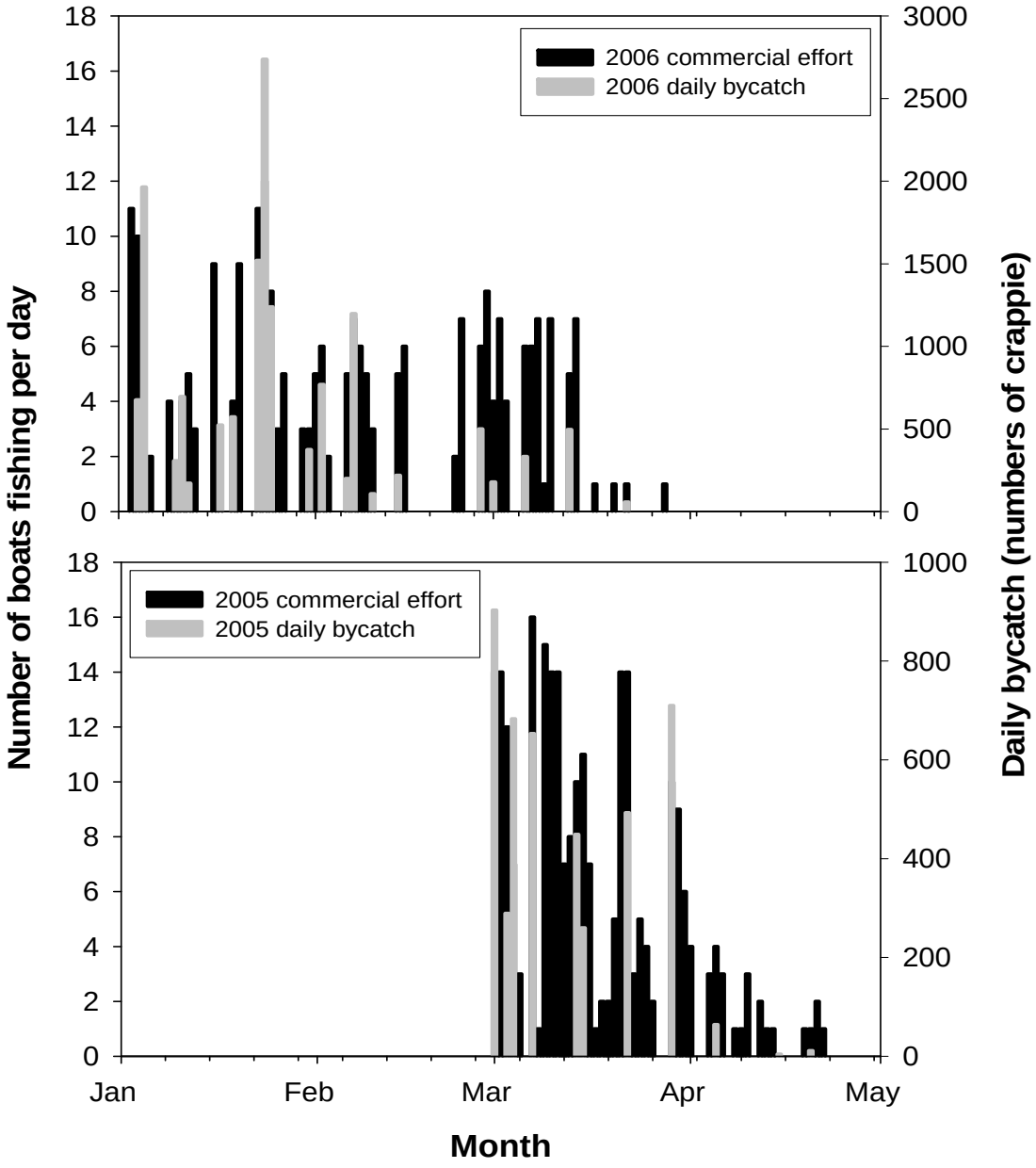

Figure 3-1. Commercial fishing effort (number of boats fishing per day) and daily black crappie bycatch (numbers) for the 2005 and 2006 commercial gill net seasons at Lake Dora. Daily bycatch estimates are shown for 2005 and 2006 for days where onboard observation data was available.

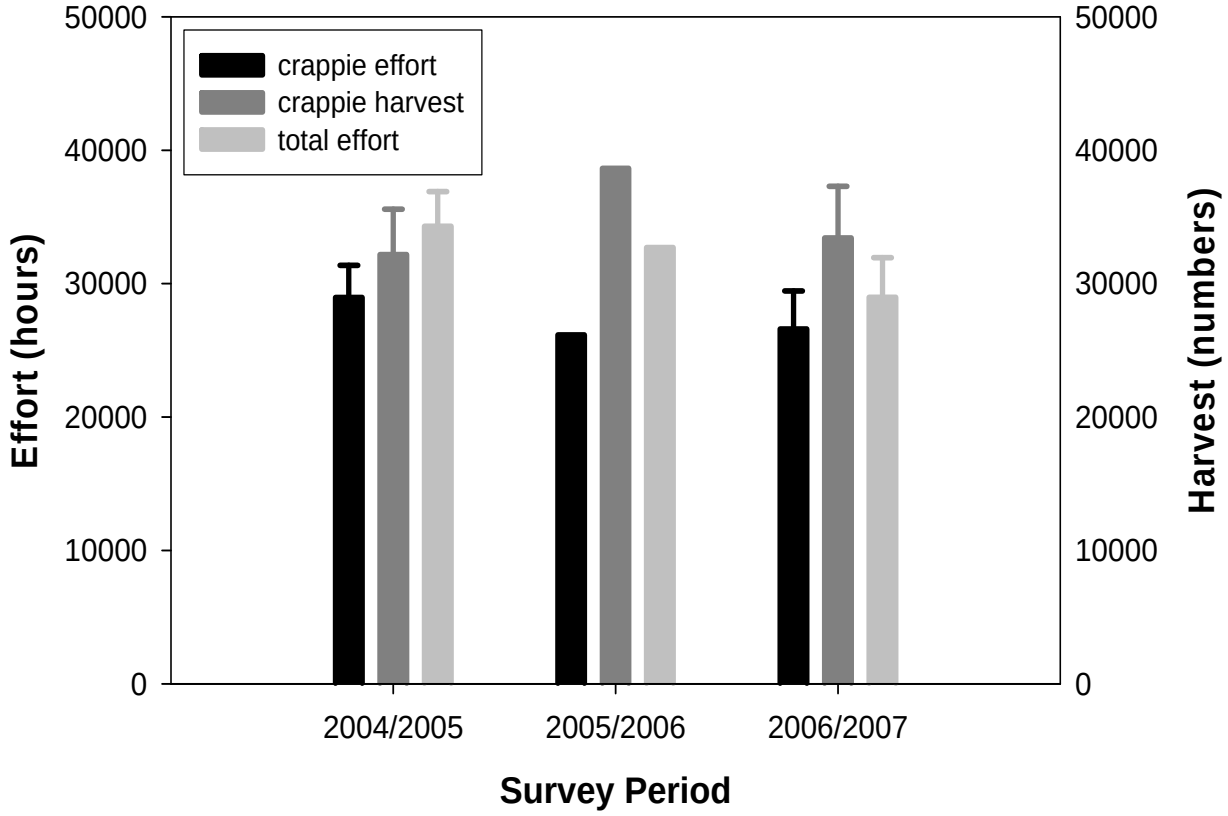

Figure 3-2. Total recreational fishing effort (hours), black crappie effort (hours), and harvest of black crappie (numbers) during the three creel survey periods at Lake Dora. The associated standard error is reported for the survey periods in 2004/2005 and 2006/2007 (no SE could be calculated in 2005/2006 due to missing data from one 28-day time period).

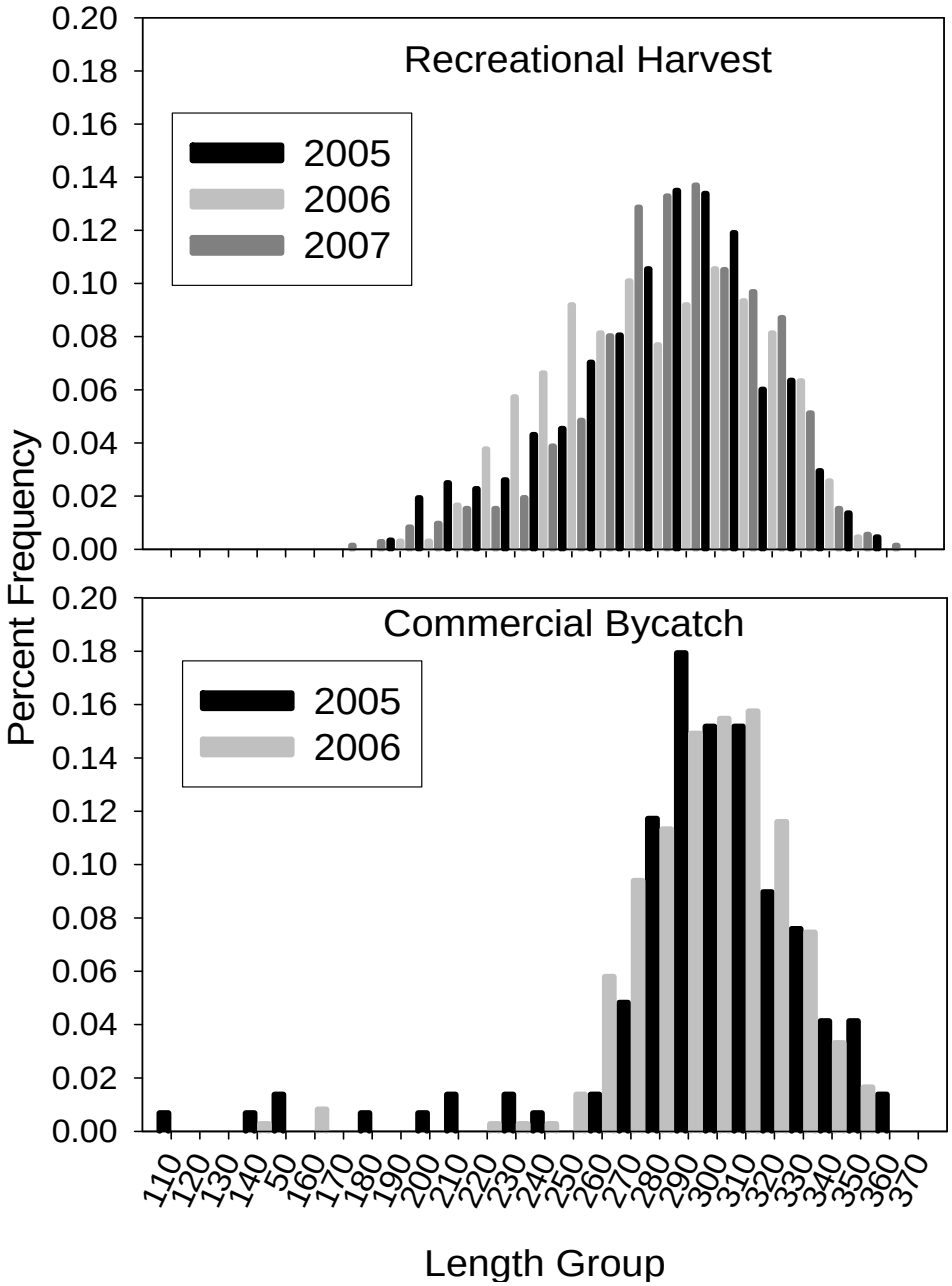

Figure 3-3. Relative length frequencies of black crappie measured from the recreational catch (carcasses and creel) and commercial gill net bycatch on Lake Dora. Measurements of black crappie were sampled from the black crappie recreational catch on Lake Dora in 2005 (N = 882), 2006 (N = 664), and 2007 (N = 723), and from commercial gill net bycatch on Lake Dora in 2005 (N = 145) and 2006 (N = 362). Length group on x-axis represents 10 mm size groups.

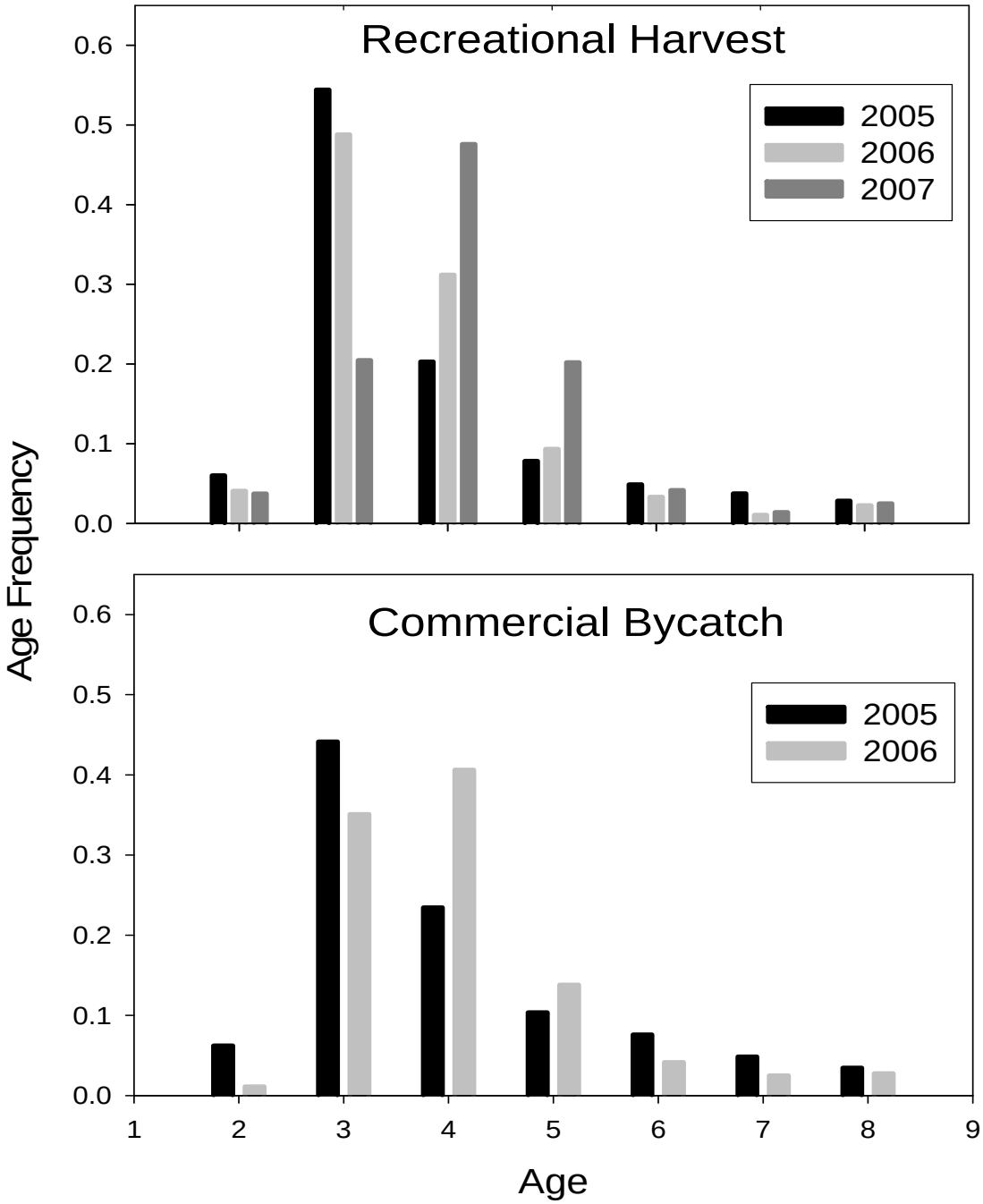

Figure 3-4. Age frequency of black crappie collected from the recreational catch (carcasses and creel) and commercial gill net bycatch on Lake Dora. Ages were determined from the recreational catch in 2005 (N = 882), 2006 (N = 664), and 2007 (N = 723), and from commercial gill net bycatch in 2005 (N = 145) and 2006 (N = 362).

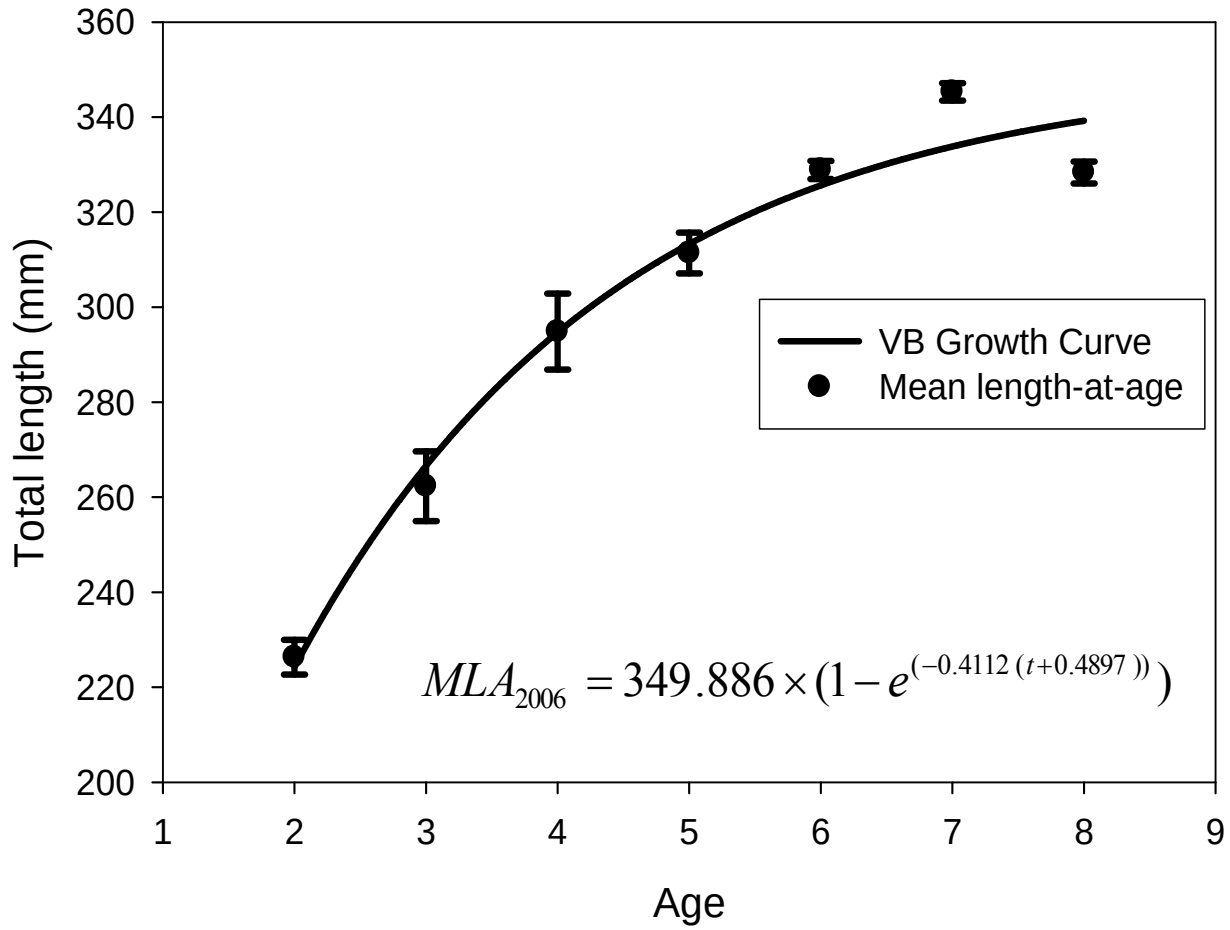

Figure 3-5. Von Bertalanffy growth curve fit to mean length-at-age values for black crappie collected from the recreational fishery (carcasses and creel) at Lake Dora in 2006. Error bars represent one standard deviation around the mean length-at-age values.

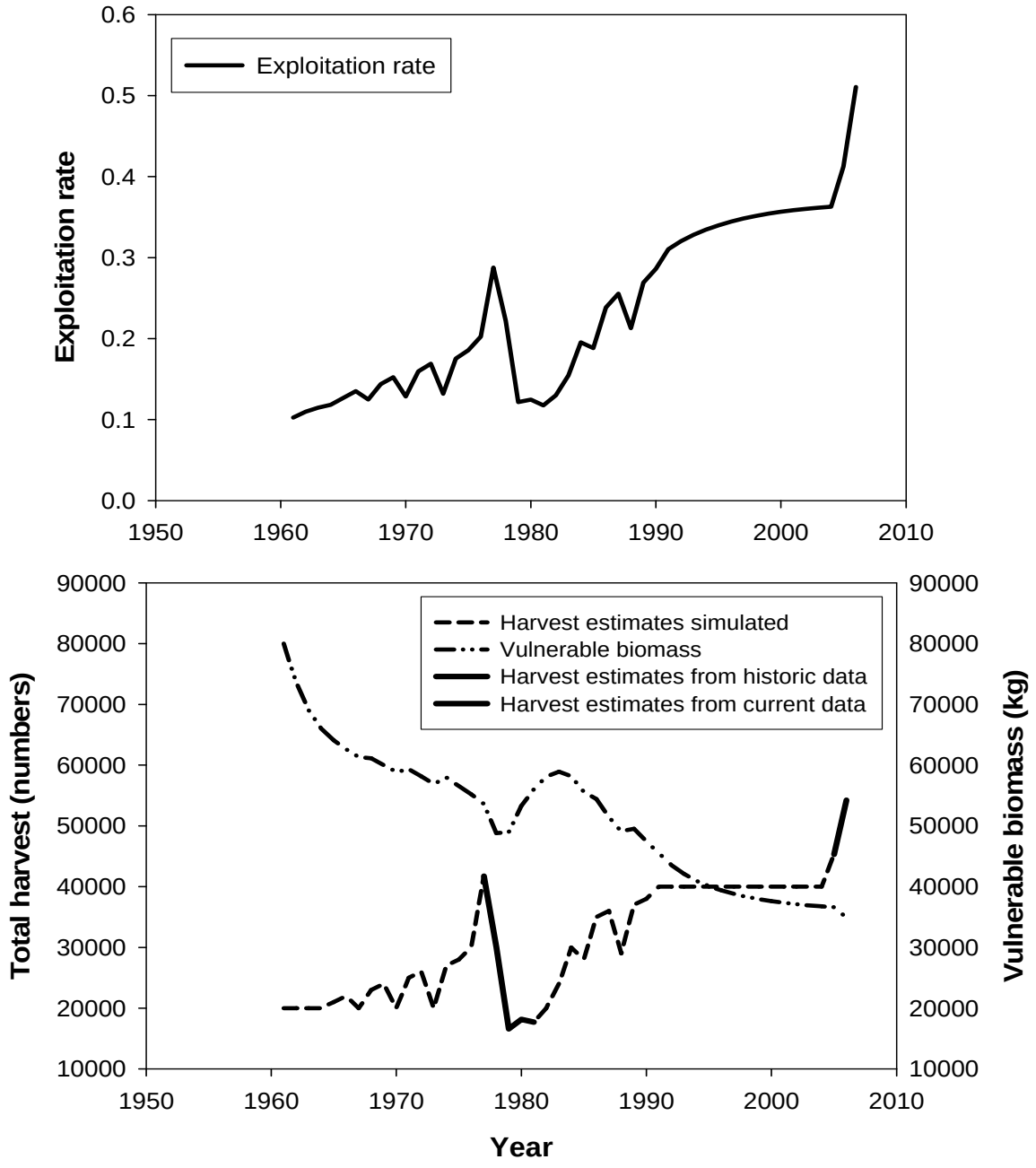

Figure 3-6. Estimates of exploitation from 1961 to 2006 and estimates of historical total harvest and vulnerable biomass from the SRA model. Values of the total number of black crappie harvested are simulated for years that harvest data is not available.

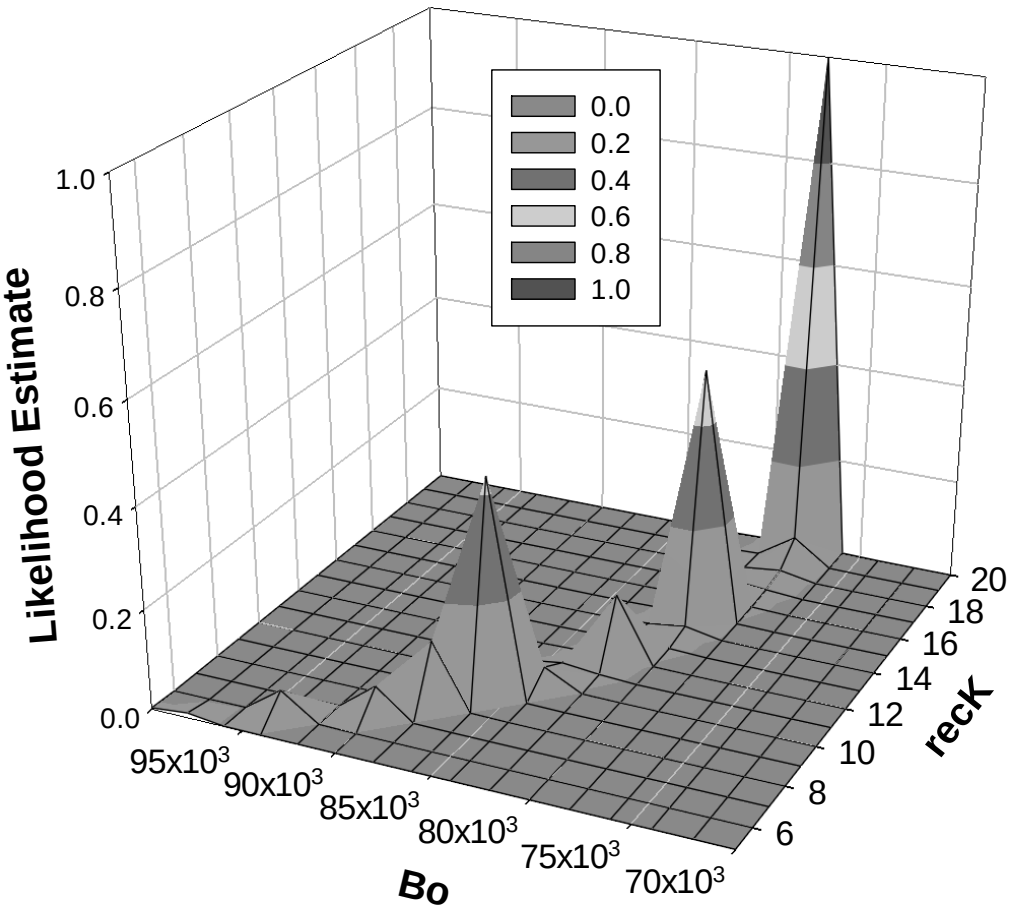

Figure 3-7. Maximum likelihood profile for recK values ranging from 5 to 20 and Bo values ranging from 70,000 to 100,000 kg.

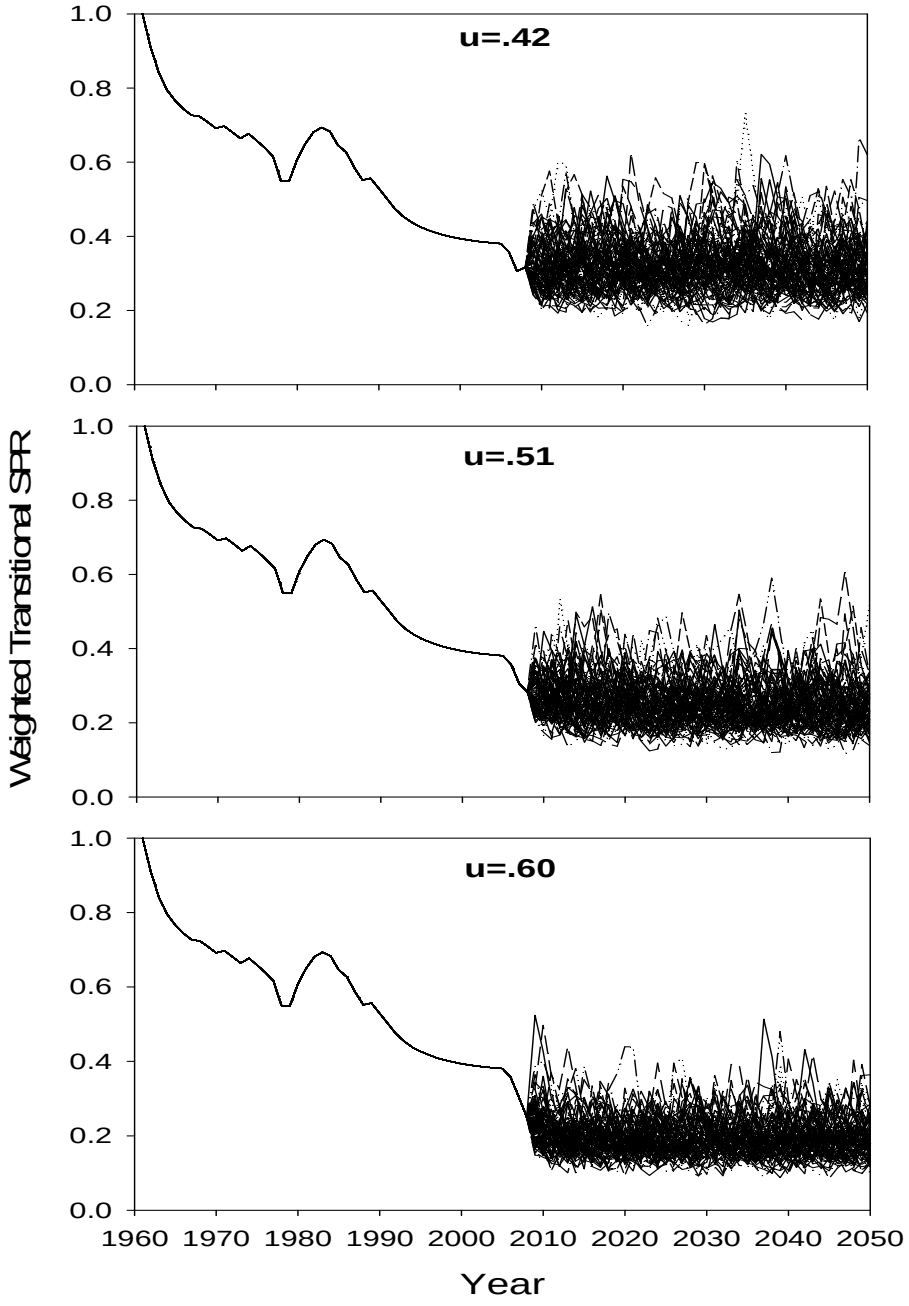

Figure 3-8. Weighted transitional SPR estimated from SRA from 1961 to 2050 with Monte Carlo simulations (100 iterations) under three exploitation scenarios. The three exploitation scenarios were μ_{total} = 0.42, 0.51, and 0.6. Recruitment variability = 0.4 from 2007 to 2050.

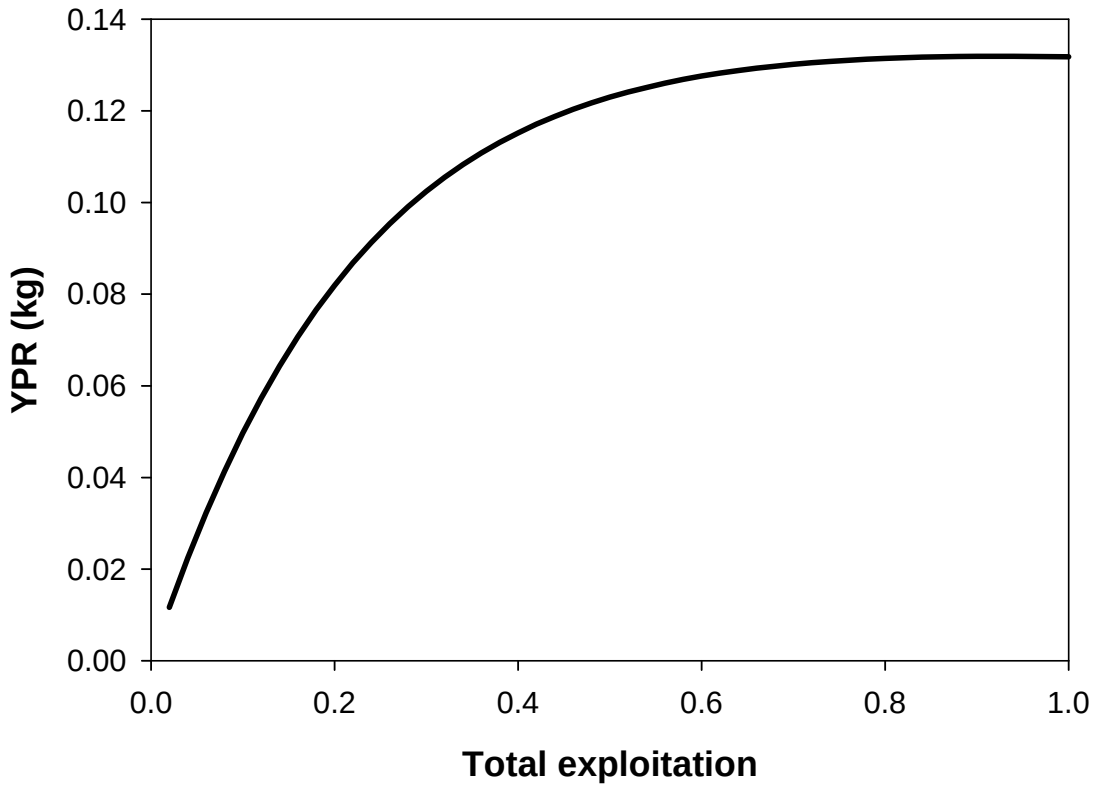

Figure 3-9. Results for YPR (kg) values at total exploitation rates from 0.2 to 1.0 from yield-per-recruit model simulations.

Table 3-1. Summary of results from stratified sampling design in 2005 and 2006. Results include stratified bycatch mean per fishing day, total bycatch estimate, variances for bycatch mean per fishing day and total bycatch estimate, 95% upper and lower confidence intervals, and the number of degrees of freedom used.

Year	$\overline{X}_{ST}$	$\hat{X}_{ST}$	$Var(\overline{X}_{ST})$	$VAR(\hat{X}_{ST})$	CI low $\hat{X}_{ST}$	CI high $\hat{X}_{ST}$	DF
2005	324.52	17,199	4,011.76	11,269,026	8,777	25,622	5.50
2006	630.38	30,258	12,357	28,469,427	19,048	41,469	17.85

Table 3-2. Summary of results from secondary mortality experiment for treatment fish in 2005 and 2006 and control fish in 2006. Year, treatment type, number of replicates, and the mean mortality and associated standard error are shown.

Year	Type	Replicates	Mean mortality	Standard error
2005	treatment	17	0.31	0.06
2006	treatment	23	0.47	0.07
2006	control (hoopnets)	6	0.10	0.05
2006	control (electrofishing)	4	0	0

Table 3-3. Estimates of recreational exploitation rate (μ_{rec}) based on values of the number of higher reward value (C_H) and standard reward tag fish caught (C_S). Tagging mortality (TM) and tag loss (TL) were simulated at 5% and 10%. The total number of higher value tag fish caught (C_H), and standard tag fish caught (C_S) were calculated based on differing values of higher value reward tag reporting rate (λ_H) from 0.5 to 1.0.

λ_H	C_H	R_H	R_L	C_L	T_L	T_H	λ_L	μ_{rec} (5%TL-5%TM)	μ_{rec} (10%TL-10%TM)
0.5	54	27	34	221	413	101	0.15	0.59	0.67
0.6	45	27	34	184	413	101	0.18	0.50	0.56
0.7	39	27	34	158	413	101	0.22	0.42	0.48
0.8	34	27	34	138	413	101	0.25	0.37	0.42
0.9	30	27	34	123	413	101	0.28	0.33	0.37
1.0	27	27	34	110	413	101	0.31	0.30	0.33

Table 3-4. Empirical estimates of vulnerable biomass (kg) and total exploitation (μ_{total}) in 2006 and model predicted values of vulnerable biomass and total exploiation in 2006. Empirical estimates in 2006 were calculated with an estimated total harvest of 54, 221 (recreational and commercial) and 2006 model predicted values of vulnerable biomass and total exploitation were derived with leading parameter estimates of Bo = 80,000 kg and recK = 15.22.

Parameter	2006 empirical estimate	2006 model predicted value
vulnerable biomass (kg)	34,912	35,080
total exploitation (μ_{total})	0.51	0.51
total harvest (numbers)	54,221	.

Table 3-5. Estimates of mean vulnerable biomass (kg), mean total harvest (numbers) and mean weighted transitional SPR in the terminal year 2050 determined from Monte Carlo simulations (1,000 iterations). Three exploitation scenarios (μ_{total} = 0.42, μ_{total} = 0.51, μ_{total} = 0.60) are shown.

Exploitation scenario	u_{total}	Mean vulnerable biomass	Mean total harvest	Mean SPR
1	0.42	32,026	41,592	0.32
2	0.51	26,359	43,491	0.25
3	0.60	21,973	44,583	0.19

CHAPTER 4
DISCUSSION

Black crappie is the primary sport fish targeted by recreational anglers at Lake Dora, and my results show that the population could be negatively impacted by increases in exploitation resulting from either the recreational fishery or bycatch from the commercial gill net fishery for gizzard shad. Currently, FWC has not defined a standard to measure impacts of bycatch and determine levels of commercial exploitation that are acceptable. I used a biological reference point (SPR) determined from an age-structured model to attempt to determine what levels of total exploitation could be sustainable without risking recruitment overfishing. I also used maximum yield per recruit to investigate the potential for growth overfishing to occur at varying total exploitation rates. It is important to realize that negative impacts such as reduced catch or decreased angler success may occur at fishing mortality rates below those which cause recruitment overfishing and changes in the vulnerability to harvest schedule may influence the potential for recruitment overfishing to occur at varying total exploitation rates.

Additionally, management decisions are still required to determine how much of the total sustainable exploitation rate is allocated to the recreational fishery versus bycatch from the gill net fishery. The total sustainable exploitation rate is approximately 0.42, which results in an SPR near the target goal of 0.3 to 0.35. The estimated recreational exploitation rate in 2006 was approximately the total sustainable exploitation rate, and increases due to recreational fishing and/or commercial bycatch greatly increase the probability of recruitment overfishing. Total exploitation in 2006 resulted in an estimated exploitation rate (0.51) that produces worrisome SPR levels and is most likely not sustainable. The exploitation via bycatch of black crappie at Lake Dora is a negative effect because it is not resulting from a directed fishery and all mortality results in waste. The gill net fishery was regulated to minimize bycatch as much as possible, but

bycatch mortality occurred at rates that cause concern for recruitment overfishing. Resource managers must evaluate policy trade-offs to consider the benefit of the gizzard shad removal and the negative impacts of bycatch mortality.

Commercial fishing occurred on Lake Dora during 2005 and 2006 and total commercial fishing effort was approximately equal during the two fishing seasons. However, the temporal range of effort differed, which may have influenced total gill net bycatch mortality among the commercial seasons. The commercial fishing season in 2005 began two months later than the commercial season in 2006, which could have resulted in differences in catchability due to differing vulnerability to capture in gill nets. This is plausible due to black crappie inshore spawning movements occurring during the later months of the fishing seasons. Total bycatch estimates in 2006 were nearly twice as high as total bycatch estimates in 2005. These results suggest that bycatch could be reduced by timing the commercial fishing season to prevent fishing during winter and early spring. Reducing total bycatch mortality is achieved by reducing the amount of total bycatch or reducing mortality resulting from bycatch. Timing of season could potentially reduce the amount of total bycatch without increasing mortality resulting from bycatch. Bycatch mortality rates would not likely increase by timing of season because I found no significant impact of water temperature or dissolved oxygen levels on bycatch mortality.

No initial mortality of bycatch was observed at Lake Dora during gill net operations, and secondary mortality was the primary mortality source for black crappie caught in commercial gill nets. This was likely due to the maximum soak time of two hours. Total mortality of black crappie captured via gill nets at Lake Apopka, Florida was estimated from 1993 to 1997 and results indicated that 87% survived the treatment (J. Crumpton, FWC, unpublished data). Similarly, secondary mortality accounted for the majority of total mortality and only a small

percentage of total mortality observed was initial mortality, which generally occurred in nets fished greater than two hours.

The potential for adverse population-level effects resulting from commercial bycatch is greatest when recreational exploitation is already high. A previous evaluation found negligible impacts from gill net bycatch for black crappie on Lake Apopka, Florida using a transitional SPR constructed from an SRA (M. Allen, UF, unpublished data), due to low recreational exploitation (~1 fish/acre/year). Conversely, commercial harvest of black crappie at Lake Okeechobee, Florida coupled with recreational harvest increased exploitation to 65%, but the effects were increased growth rates and the population did not show signs of overfishing (Schramm et al. 1985). However, the conclusions of this study were based on catches and angler success, and they did not investigate the potential for recruitment overfishing.

Other studies have assessed population-level impacts of bycatch with modeling techniques. Crouse et al. (1987) developed a stage-based matrix model that incorporated fecundity, survival, and growth rates, and used yearly iterations to make population projections for loggerhead sea turtles *Caretta caretta*. The model used seven life stages from eggs/hatchlings to mature breeders and tested the sensitivity of bycatch mortality on population growth rates. They found that reducing mortality in the large juvenile and adult life stages provided the best protection for population viability. Diamond et al. (1999) explored the population level effects of catch and bycatch on Atlantic croaker *Micropogonias undulatus* in the Gulf of Mexico and the Atlantic Ocean. Catch of Atlantic croaker, including bycatch, had historically been at least three times higher in the Gulf than the Atlantic; however, primarily juveniles are taken in the Gulf fisheries whereas fisheries in the Atlantic have targeted adult fish. Long-term intensive fishing in the Gulf caused severe declines in abundance of Atlantic croaker, but there was no change in size

distribution and age at maturity, and large fish remained common. In contrast, the Atlantic fishery targeting adult fish has caused changes in age at maturity and size structure of that population. Diamond et al. (2000) used stage-within-age based matrix models of Atlantic croaker in the Gulf of Mexico and Atlantic to investigate population-level effects of shrimp trawl bycatch. The Gulf model showed a rapidly declining population, and the Atlantic population showed only a modest decline. Results indicated that both populations were more sensitive to survival of adults than first-year survival, and reducing late juvenile and adult mortality could reverse population declines. Results from these studies support my conclusion that population-level impacts can occur, especially when targeted-fishery exploitation is also high.

Biological reference points such as spawning potential ratio are commonly used as critical metrics to measure the potential of recruitment overfishing. Goodyear (1993) defines SPR as the ratio of fished to unfished magnitude of *P* (reproductive potential of an average recruit) and is a measure of the impact of fishing on the potential productivity of a stock. Critical levels had typically been set in the range of 0.2 to 0.3, based primarily on work in the Northwest Atlantic (Goodyear 1993). SPR target values of 0.35 to 0.4 have also been suggested (Clark 2002), but the critical level for any particular species is influenced by the level of recruitment compensation for fishing mortality (Goodyear 1993). The state of Florida has adopted a target SPR of 0.35 for some heavily exploited marine species, including the spotted seatrout *Cynoscion nebulosus*, which have shown worrisome levels of SPR values due to recreational exploitation (no commercial exploitation and very limited bycatch) (Murphy et al. 1999).

Estimates of exploitation from tagging studies are always subject to uncertainty due to tag loss, tagging mortality, and reporting rate. For my model simulations, I utilized the best estimate of recreational exploitation (0.42) from tag returns corrected for tag loss of 5%, tagging mortality

of 5%, and reporting rate of higher value reward tags of 70%. My estimate of recreational exploitation in 2006 (μ_{rec} = 0.42) was comparable to estimates of exploitation for black crappie in other southeastern systems. Larson et al. (1991) estimated exploitation rates ranging from 40 to 68% in three Georgia reservoirs, Allen and Miranda (1995) estimated a mean exploitation rate of 42% for white and black crappie in 10 Southeast and Midwest lakes, and Allen et al. (1998) found that exploitation averaged 48% for 18 lakes in the Southeast and Midwest. Black crappie are one of the most heavily harvested and exploited freshwater fishes in the United States, and strong size selectivity under heavy exploitation may affect black crappie population dynamics (Miranda and Dorr 2000).

My exploitation estimate was critical for model simulations because the model was fit to my 2006 empirical estimates of exploitation and vulnerable biomass. An unbiased estimate of exploitation was additionally important to reduce parameter uncertainty, because there is also structural uncertainty in the SRA. The SRA model reduces population size based on catches alone, and does not account for other factors that may influence recruitment such as habitat changes. This is of particular importance because if the gizzard shad removal is successful, improved water clarity could result in increased aquatic macrophyte abundance thereby changing the available habitat and factors that influence black crappie recruitment and growth.

All model simulations assumed vulnerability to harvest was equal for the recreational and commercial fisheries. This is important because the vulnerability to harvest schedule directly impacts estimates of exploitation. It is likely that vulnerability between commercial and recreational fisheries were similar based on the size and age distributions of the harvest. Although recreational anglers did tend to harvest some smaller black crappie that were not fully vulnerable to the commercial fishery, Miranda and Dorr (2000) showed that recreational anglers

tend to select for fish over 250 mm TL. Additionally, much of the recreational angling effort occurs in open water areas where gill nets are fished. The number of tag returns from the commercial fishery was significantly lower possibly indicating a difference in vulnerability to harvest, however commercial fishers had incentive to not return tags and no reliable reporting rate could be obtained for the commercial fishery.

My future projections were conducted under the assumption that total exploitation remained constant through the terminal year. This scenario is unlikely, because changes in angler catch rates through fish reductions via recreational and/or commercial exploitation would probably influence recreational fishing effort. Cox et al. (2003) found that angling effort depends on the angler catch rate, and there is no reason to expect that the level of fishing effort that produces the maximum total yield will also provide maximum total satisfaction to anglers. Additionally, Walters and Martell (2004) state that most fisheries reach a bionomic equilibrium where they become "self-regulating" in the sense that further stock decline past some equilibrium caused by development of a fishery should trigger a reduction in fishing effort and mortality allowing the stock to begin recovery. Thus, it is likely that recreational effort would decline if total exploitation continued to increase and catch rates declined, due to decreased angler satisfaction and shifts in fishing effort to other systems. Under this scenario of bionomic equilibrium, commercial bycatch will probably not result in recruitment overfishing. However, decreased angler satisfaction and fishing effort is still a negative impact resulting from increased exploitation, which could occur due to bycatch mortality. Reduced recreational angler effort caused by commercial bycatch mortality warrants furture investigation because lower effort would constitute "harm" to the recreational fishery.

CHAPTER 5
MANAGEMENT IMPLICATIONS

Impact on the black crappie fishery due to bycatch mortality may be acceptable if the gizzard shad reduction is successful in improving water clarity and increasing aquatic macrophyte abundance. A management decision must be made for the future of commercial fishing with gill nets on Florida lakes that evaluates the trade-offs of the positive effects of biomanipulation and possible negative effects of bycatch on recreational fisheries. Possible management alternatives are to 1) discontinue the gill net fishery to eradicate bycatch and optimize the black crappie recreational fishery, or 2) increase commercial effort and gizzard shad exploitation to optimize the success of the biomanipulation. It is plausible that continuing the program at the current level of commercial effort will most likely not optimize either management objective.

Another alternative is to initiate an active adaptive management plan. Active management of recreational fisheries implies that a complete management procedure is in place, with clear goals or objectives for the fishery, management schemes to keep the total harvest or exploitation rates within target limits, and methods to determine whether the goals or objectives have been met (Walters 1986; Pereira and Hansen 2003). Little experience has been gained in actively managing recreational fisheries due to the extensive and diverse array of recreational fisheries, few recreational fisheries are of such singular importance that they demand the sociopolitical or economic motives, and many passive management schemes are in place in response of the need for management (Pereira and Hansen 2003). For successful active adaptive management in recreational fisheries, agencies must commit to a clear goal or objective. In the case of the Lake Dora commercial gill net fishery, possible objectives are 1) reducing the gizzard shad population enough to change the trophic structure or 2) optimizing black crappie recreational angling

satisfaction. If the goal of the Lake Dora fishery is to reduce gizzard shad abundance to levels that result in trophic structure alterations, then a long-term management plan should be implemented that involves fishing the gizzard shad intensively, measuring the levels of gizzard shad reduction, measuring levels of chlorophyll reduction, and measuring the black crappie bycatch mortality and angling success. Another consideration in the evaluation of the policy trade-off is the effect that a change in the trophic structure would have on the black crappie population. A shift in the trophic structure may result in changes in water clarity, aquatic macrophyte abundance, and fish productivity that could impact black crappie population dynamics and angling success, which is not accounted for in SRA simulations.

Fisheries management inherently requires making decisions that involve trade-offs. Management agencies often try to make decisions that optimize all alternatives, which can create a situation where none of the management alternatives are optimized. Failure to admit the severity of trade-off relationships can result in policy choices that are not beneficial for anyone (Walters and Martell 2004). The trade-offs associated with the gizzard shad biomanipulation and black crappie bycatch must be considered and clear management objectives defined. If commercial fishing continues, methods must be set forth to measure the effectiveness of the management objectives. My results show that the current size-selective removal of gizzard shad at Lake Dora could cause negative impacts to the black crappie population, with the potential for recruitment overfishing. Resource managers should consider these impacts and the trade-offs they represent when considering commercial fishing operations.

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BIOGRAPHICAL SKETCH

Jason Randall Dotson was born February 11, 1980, in Manassas, Virginia, the son of Paul Randall and Rae Lynn Dotson. He was raised with his younger sister Jennifer on Lake Jackson, a private reservoir in Northern Virginia. Under the influence of his father Randy and grandfather Paul, Jason acquired a love and appreciation for the outdoors at an early age. In 1998, Jason began his collegiate studies at Virginia Tech, where he would earn a Bachelor of Science degree in fisheries science in 2003. While at Virginia Tech, he worked on a variety of research projects involving smallmouth bass, muskellunge, striped bass, and the federally endangered Roanoke logperch. After graduation, Jason experienced a short-lived career as an insurance adjuster. In August of 2004, he decided to pursue a career in fisheries science and served as a fisheries technician for the University of Florida, where he worked on a variety of projects involving gizzard shad, American shad, black crappie, spotted sunfish, and largemouth bass. In August of 2005, Jason began his own research as a graduate student in the Department of Fisheries and Aquatic Sciences at the University of Florida. He graduated in August 2007 with a Master of Science degree, and is currently working as a fisheries biologist for Florida Fish and Wildlife Conservation Commission.