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# POTENTIAL ENVIRONMENTAL EFFECTS OF WATER WITHDRAWALS FROM THE ST. JOHNS RIVER—ICHTHYOPLANKTON ENTRAINMENT

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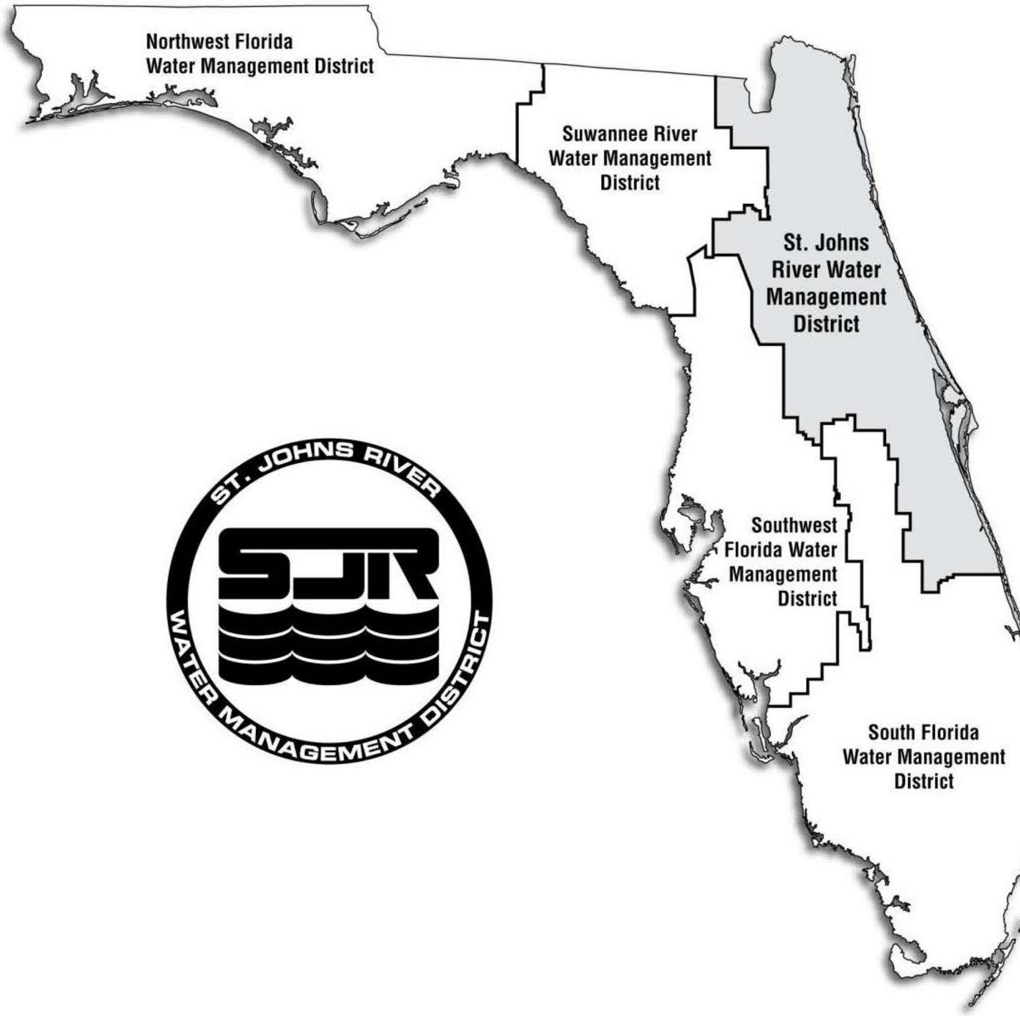


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# EXECUTIVE SUMMARY


Water withdrawals have the potential to affect St. Johns River fish populations directly by physically removing fish eggs and larvae (ichthyoplankton) through entrainment. To evaluate potential fish loss to entrainment, extensive ichthyoplankton surveys were conducted once to twice weekly from February 2008 through September 2009 within six regions under consideration as potential withdrawal sites in the Middle St. Johns River Basin. A total of 8,071 fish eggs, 709,603 fish larvae, and 19,086 juvenile/adult fishes representing 47 taxa were collected. Larval fishes were most abundant at the State Road 46 (SR 46; near the outflow of Lake Jesup) and within Lake Monroe and least abundant in the river channel at SR 50. At all locations, gizzard shad (*Dorosoma cepedianum*) and threadfin shad (*Dorosoma petenense*) larvae were numerically dominant. Other abundant species were clown goby (*Microgobius gulosus*), naked goby (*Gobiosoma bosc*), black crappie (*Pomoxis nigromaculatus*), bluegill (*Lepomis macrochirus*), and American shad (*Alosa sapidissima*). Together these species comprised >95% of the total catch. Potential entrainment losses of these seven species were calculated and extrapolated into adult equivalent losses (*NA*) using an Egg Production Model.

Data suggest that under the worst-case, maximum water withdrawal scenarios, predicted *NA* losses of gizzard shad, threadfin shad, clown goby, naked goby, bluegill, and black crappie due to larval entrainment will have negligible to minimal population effects. Over the 2-year period combined, *NA* losses of gizzard shad, on average, constituted <4% of the standing stock of this species in any of the water bodies for which comparable biomass data were available. Extrapolated to an annual basis, *NA* losses would account for <2.0% of the standing stock in these water bodies. For threadfin shad, *NA* losses over 2 years under the worst-case scenarios constituted <2.3% of their estimated standing stock in any water bodies considered.

Clown and naked gobies were not present at the Lake Poinsett and SR 50 locations but were abundant at SR 46, Lake Monroe, Yankee Lake, and SR 44. Data suggest that *NA* losses due to larval entrainment under the worst-case, full withdrawal scenarios could have localized impacts on clown and naked goby populations by reducing adult numbers up to 15% on an annual basis. However, wide interannual variation in the abundance of the larvae of these two species observed in this study may override or mask entrainment effects that might occur due to water withdrawals. Abundance of both species was significantly lower at all locations in 2009 than in 2008. Since both clown and naked gobies are considered estuarine species, these declines likely reflect population responses to the massive inflow of freshwater to the basin that occurred following the passage of Tropical Storm Fay in August 2008. More information on abundance and the ecological role of gobies in the Middle St. Johns River and on their population responses to freshwater inflow is needed if potential entrainment effects are to be fully understood.

Black crappies and bluegills are both popular sport fish in the St. Johns River. Predicted *NA* losses of black crappies under the worst-case withdrawal scenarios generally constituted <1.0% of their estimated standing stock in any of the water bodies considered. In addition, combined *NA* losses of black crappies over the 2 years generally amounted to <4% (<2% annually) of the number of crappies harvested by sport fishermen. Predicted *NA* losses of bluegills under the worst-case withdrawal scenarios was also small, generally constituting <0.1% of their estimated standing stock in any of the water bodies considered. Combined *NA* losses of bluegills over the 2 years amounted to <8% (<4% annually) of the total sport fish harvest of bream (bluegill and redear sunfish [*Lepomis microlophus*] combined). Estimates of angler harvest of bream used are likely conservative, however, because they only reflect harvest during a portion of the year.

Potential *NA* losses of American shad could have negative population impacts depending on the withdrawal site. The abundance of American shad along the east coast of the United States is at an all-time low. Currently the Atlantic States Marine Fishery Commission (ASMFC) is overseeing restoration efforts to rebuild adult stocks in many rivers. As a part of this restoration initiative, Florida is required to conduct annual surveys of American shad recreational harvest. In 2011, the Florida Fish and Wildlife Conservation Commission (FWC) submitted an American shad sustainable fishing plan for the St. Johns River to the ASMFC. The recommendation of the FWC was to maintain current harvest of American shad from the St. Johns River at the present level. Any increase in American shad harvest not accompanied by a concomitant increase in independent spawning stock abundance estimates could trigger either reduced bag limits or spatial and/or temporal closures of the recreational fishery.

In 2011 and 2012, estimates of total annual recreational American shad harvest from the St. Johns River were 198 and 232 fish, respectively. Under the worst-case scenarios, withdrawals at SR 50 alone over the 2 years combined would account for *NA* losses of 208 to 424 adult fish (104 to 221 fish annually). Combined with recreational harvest this would increase the current annual loss of adult American shad 50% to 100%. Under the best-case scenarios, SR 50 withdrawals alone combined with recreational harvest would increase current annual loss of adult American shad 11% to 22%. Based on the information presented, SR 50 is least desirable as a water withdrawal site due to potential negative impacts to American shad. Should withdrawals occur at SR 50, they should be cut off or curtailed during the December through April peak shad spawning season.

Without entrainment effects at SR 50, overall predicted *NA* losses of American shad are considerably smaller but still potentially significant. Under the worst-case scenarios modeled (low estimated lifetime fecundity), *NA* losses at the other withdrawal locations all occurring simultaneously (200 million gallons per day [mgd] total) combined with angler harvest could increase annual loss of adult American shad 17% to 33%. Under the best-case scenarios (high estimated lifetime fecundity), cumulative annual loss of adults would only increase 3% to 6%.

Potential entrainment of American shad at the SR 46 location may be considerably larger if winter-spring flow conditions are higher than those sampled during this study. Historical data on American shad spawning suggests the river reach adjacent to the SR 46 withdrawal site constitutes important American shad spawning habitat under higher flows. During ichthyoplankton sampling, average flows from December through April at SR 50 were low (282 cubic feet per second [cfs]), a condition historically associated with greater American shad spawning upstream of Lake Harney. However, during higher flows (SR 50 discharge >706 cfs), a condition that did not occur during the 2008–2009 sampling window, previous studies have found the majority of American shad spawning occurred downstream of Lake Harney near the SR 46 site. Thus, there is also a need to minimize ichthyoplankton entrainment at this location. In addition, two other anadromous herrings—hickory shad (*Alosa mediocris*) and blueback herring (*Alosa aestivalis*)—also spawn near this location.

Efforts to minimize larval fish entrainment should be included in the design of all intake structures that withdraw water from the St. Johns River. Even though withdrawal effects may be minimal for the majority of fish species, minimizing larval entrainment where possible seems prudent as well as fundamental to ensure that fish stocks in the river are protected to the greatest extent possible. Commonly utilized design features that will help minimize entrainment include (1) constructing river intakes so that passive ichthyoplankton moving downstream in the current deflect away from the withdrawal location, (2) installing wedge wire screens with small mesh sizes, and (3) limiting structure inflow velocities.

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# ACRONYMS AND ABBREVIATIONS

| | |
|---|---|
| µhos | Microohms |
| µm | Micrometer |
| ANOVA | Analysis of Variance |
| ASMFC | Atlantic States Marine Fishery Commission |
| cfs | Cubic feet per second |
| cm | Centimeter |
| d | Day |
| DO | Dissolved oxygen |
| EPA | U.S. Environmental Protection Agency |
| EPM | Egg Production Model |
| ft | Feet/foot |
| FWC | Florida Fish and Wildlife Conservation Commission |
| g | Gram |
| ha | Hectare |
| in. | Inches |
| kg | Kilograms |
| L | Liter |
| lbs | Pounds |
| m | Meter |
| mg | Milligrams |
| mgd | Million gallons per day |
| mm | Millimeters |
| *NA* | Adult equivalent losses |
| psu | Practical salinity units |
| QA/QC | Quality Assurance/Quality Control |
| s | Second |
| SD | Standard deviation |
| SJRWMD | St. Johns River Water Management District |
| SL | Standard length |
| SR | State Road |
| TL | Total length |
| USGS | U.S. Geological Survey |

# INTRODUCTION AND BACKGROUND

The St. Johns River is a potential surface water supply source for meeting the growing water needs of central Florida. Preliminary water supply yield assessments indicate that up to 155 million gallons per day (mgd), or 8% of the mean annual freshwater discharge to the ocean, is potentially available (Lowe et al. 2012). As a part of the yield assessment analysis, the St. Johns River Water Management District (SJRWMD) conducted a wide ranging study of potential impacts of surface water withdrawals on the ecology and natural resources of the river (Lowe et al. 2012). Included among these diverse studies were investigations into potential direct effects of the withdrawals on fish populations (Miller et al. 2012). In 2008, the SJRWMD contracted with the Florida Institute of Technology to conduct studies to quantify potential losses of fish eggs and larvae (ichthyoplankton) that may occur due to surface water withdrawals. This report summarizes the results of these studies and provides projections of reductions in adult numbers and standing stocks that may result from the entrainment (being drawn into water systems and subjected to thermal, physical, or chemical stresses) mortality.

Large numbers of fish are lost annually due to surface water diversions for power generation, irrigation, and industrial and domestic use (Boreman 1977; Porak and Tranquilli 1981; Post et al. 2006; EPA 2008). Such impacts include death or injury to aquatic organisms by impingement (being pinned against screens or other parts of water intake structures) or entrainment. The exposure of fishes to potential water withdrawal effects varies with fish size, life history characteristics, intake velocities, and location and design of intake structures (Gowan et al. 1999).

Most facilities that withdraw water place screens in front of their water intakes to prevent fishes and debris from entering their systems. Systems with high inflow rates and large intake sizes (e.g., power plants) often impinge juvenile and larger fishes against these screens. To minimize impingement, the U.S. Environmental Protection Agency (EPA) suggests that withdrawal intake velocities should not exceed 0.15 meters per second ($m\ s^{-1}$; 0.5 feet per second [$ft\ s^{-1}$]) (EPA 2004). Impingement of adult and juvenile fishes is not anticipated to be a significant issue with surface water withdrawals from the St. Johns River because water intake velocities to municipal facilities are generally lower than the EPA recommendation, allowing most larger, more mobile fishes to avoid the intake screens (Zeitoun et al. 1981; Weisberg et al. 1987; Gowan et al. 1999).

Fish eggs and newly hatched larvae, however, lack the mobility to avoid entrainment into water withdrawal intake structures. Entrained ichthyoplankton suffer complete mortality (Gowan et al. 1999). If entrainment losses cause an increase in overall egg and larval mortality, then entrainment losses could reduce recruitment into the adult population and cause declines in fish standing stocks (Boreman 1977; Horst 1977; Boreman et al. 1981; EPRI 1999; Van Winkle 2000; 2002; Van Winkle and Kadvany 2003). Conversely, if entrainment has no effect on overall egg and larval mortality (which is naturally high) or only removes a small proportion of the overall production of eggs and larvae in the region, then effects may be minimal and direct impacts on population dynamics of species may be difficult to detect (Saila et al. 1997; Gallaway et al. 2007).

The biological and ecological attributes of the reproductive characteristics of individual species (e.g., timing and duration of spawning and spawning locations; larval growth and mobility; habitat use) determine the vulnerability of ichthyoplankton to entrainment. Unfortunately, reproductive and early life history information for most fish species in the St. Johns River system is not completely understood or is lacking.

In 2008 and 2009, we collected data on the species composition and abundance of ichthyoplankton in the St. Johns River near proposed surface water withdrawal locations. To our knowledge, these are the first comprehensive surveys of the temporal and spatial distribution of ichthyoplankton ever conducted in the freshwater sections of the St. Johns River. Abundance estimates were used to develop quantitative predictions on potential ichthyoplankton entrainment that would occur with water withdrawals at each location. Ichthyoplankton loss predictions were further extrapolated to quantitatively estimate withdrawal effects on adult abundance. This information will be used for analyzing potential impacts as well as locating withdrawal sites and designing intake structures to minimize entrainment.

Evaluation of the impacts of the proposed water withdrawal sites on reproductive success of fishes requires an analysis of the complex and shifting dynamics of ichthyoplankton in the St. Johns River (Figure 1).

Fish reproduction in lakes and rivers may be highly variable in both time and space (Figure 1). This variable reproductive behavior can result in patches of eggs and larvae (for patchy or habitat-specific spawning efforts) or in more widely dispersed eggs and larvae (for fish that spawn throughout the region or are broadcast spawners). After spawning, eggs and larval patches (depending on the species and their individual reproductive strategies) may diffuse outward in response to turbulent hydrodynamic processes and the vertical and horizontal swimming behavior of the larvae. For species who use limnetic or mainstream flowing habitats, this patchiness is often reflected in highly variable catches made by plankton nets towed at different locations within an individual lake or river reach (Snyder 1983).

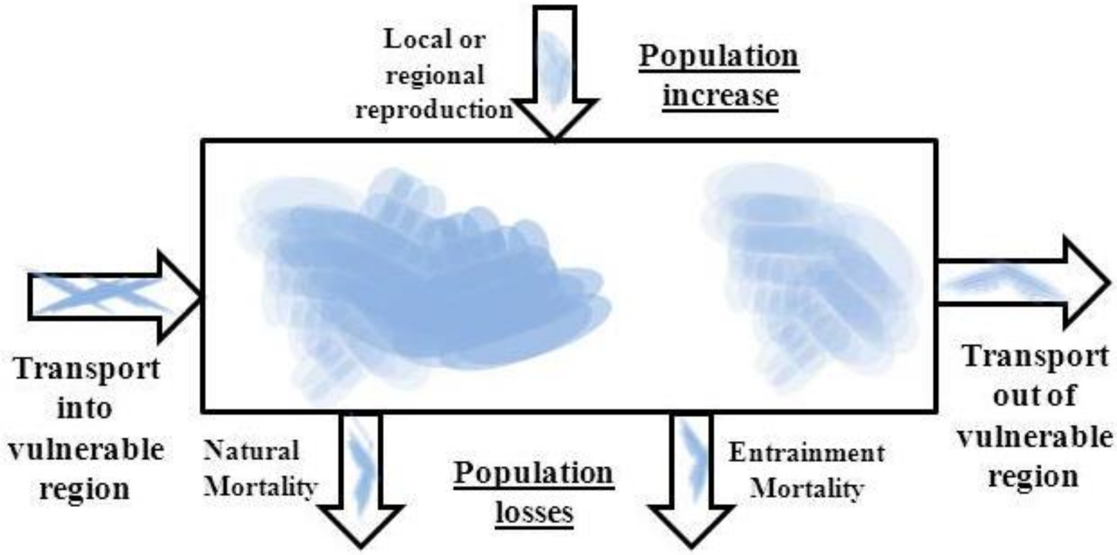



Figure 1. Schematic diagram of larval fish dynamics in a section of riverine habitat
Clouds represent patches of fish larvae.

Development of models that relate entrainment losses to adult losses requires information on natural mortality rates for the different species of concern. Unfortunately, such information is lacking for most species, and mortality rates for eggs and larval fishes are difficult to measure (McGurk 1986). Mortality of newly spawned fish eggs can be extremely high, especially in species that provide little parental care (Dahlberg 1976). As larvae grow, they undergo dramatic developmental changes that affect their morphology and behavior, and survival tends to increase. For many species, newly hatched larvae subsist on yolk for several days (endogenous feeding) and do not begin feeding until they develop complete mouth and digestive systems. These yolk-sac larvae generally have limited mobility and suffer high mortality, especially if an abundant food source is not available when the larvae begin actively feeding (exogenous feeding) (Houde 1987). Species that broadcast their eggs into the water and have drifting yolk-sac larvae generally experience the highest mortality rates (Dahlberg 1976). Other fish species build nests, lay demersal eggs, and may guard the eggs and newly hatched young (e.g., sunfishes). Demersal spawners that guard their eggs and fry typically experience lower larval mortality rates (Dahlberg 1976). Species-specific mortality rates can also vary widely. For example, Leak and Houde (1987) determined that the mortality rates of bay anchovy (*Anchoa mitchilli*) larvae between four different experimental groups varied by a factor of 20 during their first 20 days after hatching.

Growth increases the sensory capabilities of larvae and enables them to swim at higher speeds as they progress from preflexion to juvenile stages (flexion refers to a process in the morphological development of the caudal fin of larvae; see Snyder 1976 for development terminology). These enhanced capabilities enable them to capture prey, avoid predators, and select habitats. Mortality rates can still be high, however, as they become prey for different groups of predators or react to unfavorable environmental conditions.

Daily mortality rates averaged 12.3% and ranged from 6.1% to 22.2% for a number of marine and freshwater species from egg deposition through the larval development states (Dahlberg 1976). Small changes in mortality and growth rates can have substantial effects on fish recruitment (Houde 1987). As an example, survivorship curves for a cohort of larvae experiencing daily mortality rates from 5% to 25% are illustrated in Figure 2. It is not uncommon for entire cohorts of larvae to suffer complete mortality due to starvation (Houde 1987) or environmental perturbations (Kramer and Smith 1962), while larvae produced a month earlier or later or in another area might encounter conditions that promote higher survivorship. Variability in mortality is further confounded by other factors including predation (McGurk 1986), competition (DeVries et al. 1991), water quality (DeVries et al. 2009), and presence of vegetation (Dahlberg 1976; DeVries et al. 2009).

The ability to predict or estimate the survivorship and ultimate recruitment of larvae into a fish population is important when analyzing the population dynamics of species, evaluating habitat value, and establishing fishery management regulations (Smith 1981). The extremely high temporal and spatial variability in larval production and survival, and the effects of this variability on the numbers of fishes that eventually recruit to the adult population, is perhaps the most difficult biological process to assess (Houde 1987).

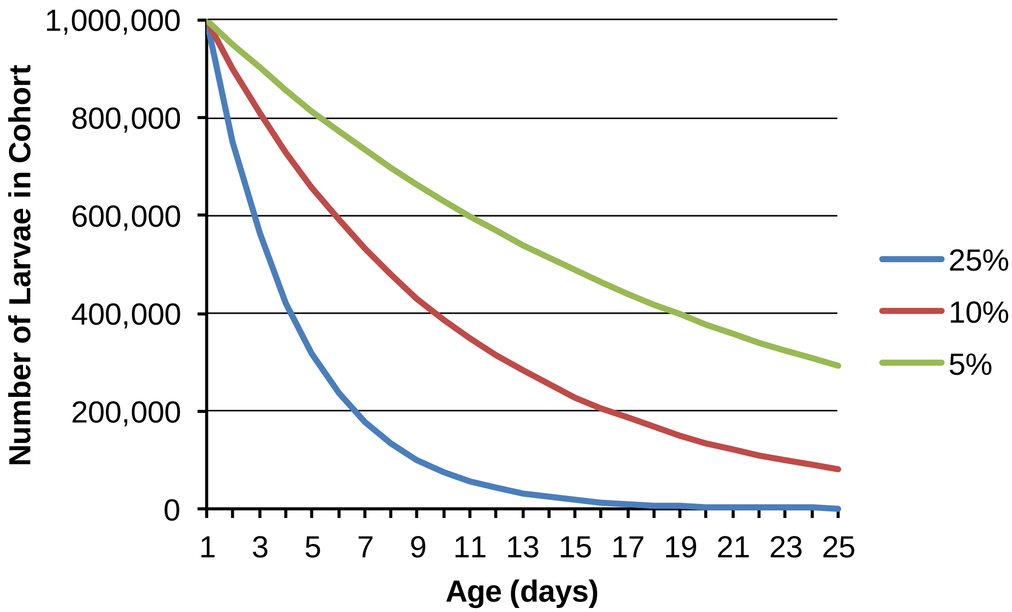



Figure 2. Survivorship curves for a cohort of 1,000,000 larval fish experiencing daily mortality rates of 25%, 10%, and 5%.

Several methods are used to attempt to extrapolate entrainment losses into adult losses. One approach is to generate the adult equivalent loss (i.e., how many adults would have been produced by the larvae that have been entrained) (Horst 1975; Goodyear 1978). Estimated adult loss can then be compared to measured standing stocks or other estimates of adult abundance to quantify impact. One drawback to this approach is that it requires detailed information on mortality of a species through all life stages. Another approach is to estimate the number of adults that produced the number of larvae entrained (Hunter and Goldberg 1980). If populations are at equilibrium, then the number of adults that produced the larvae should be approximately equal to the number of adults that would be lost. This approach, known as the Egg Production Model (EPM), was developed by the California Cooperative Fisheries Investigation to calculate abundance of adult northern anchovy (*Engraulis mordax*) and other fishes from egg surveys (Hunter and Goldberg 1980; Lasker 1985). The EPM has been widely used by many programs to estimate population biomass of a wide array of pelagic species (Alheit 1993; Zeldis and Francis 1998; Bunn et al. 2000; Armstrong et al. 2001; Melià et al. 2002; McBride et al. 2008; Dick 2009; Bernal et al. 2011).

The EPM is most effective when the reproductive parameters of the target species are well known. The method is most appropriate for large-scale, multiyear investigations into the population dynamics and management of fish species where each of the reproductive parameters has been individually studied. For example, such individual studies can determine the batch fecundity (number of eggs produced per female per spawning event), seasonal fecundity (total eggs produced per female per year), or lifetime fecundity (total eggs produced per female in her lifetime). These types of data are incorporated into models of egg production associated with size of the adults and the size/age distribution of fish in the adult population. Data are further used in population dynamic models that consider sampling frequency, egg/larval development rates at different temperatures and feeding conditions, and other biotic and abiotic conditions (Lasker 1985).

While the reproductive biology of the fishes of the St. Johns River has not been as intensely studied, the EPM approach can generate approximations of the number of adults that would have been lost to larval entrainment if the projected water withdrawal structures were in place during our survey. The sampling frequency employed during the 2008 and 2009 surveys resulted in high temporal and spatial resolution of larval abundance, providing excellent data on reproductive output. In addition, estimates of fecundity and development time are available for most of the species found in river (e.g., see fecundity estimates in Carlander 1969a; Carlander 1969b), providing a baseline for calculation of EPM-based entrainment losses. This is in contrast to a complete lack of information on the age-specific mortality rates for most of the species found in the St. Johns River, especially during their first year of life, which is necessary for calculating mortality-based estimates of adult equivalent loss.

# METHODS

## SAMPLING LOCATIONS

Ichthyoplankton were collected from 31 stations in four sampling regions (encompassing both river and lake habitats) in the Middle St. Johns River Basin from 11 February 2008 to 30 September 2009 (Figure 3). Each sampling region contained at least one potential surface water withdrawal location. Each region (with the exception of Lake Poinsett) had four to six ichthyoplankton sampling sites where approximately half were distributed both upstream and downstream of the potential intake sites. A GPS was used to mark the location of each sampling site.

The Lake Poinsett region (Figure 3) was the southernmost (upstream) region sampled and contained only one potential withdrawal site (located at the SR 520 bridge) (Figure 4). Four ichthyoplankton sample sites were located within the shallow lake proper (depths typically <1.5 m), and two stations were in the deeper (2.0–3.0 m deep) river outlet of the lake just upstream of the potential withdrawal location. Water levels during the dry season in Lake Poinsett were often too shallow to effectively sample.

The SR 50 sampling region (Figure 3) also contained only one potential withdrawal location, which was located just upstream of the bridge where SR 50 crosses the river east of Christmas, Florida (Figure 5). Habitat in this region consists of a narrow, meandering river channel within a wide, seasonally inundated floodplain. Water depths at the different ichthyoplankton sampling sites typically ranged from 1.0 to 2.5 m. Both the Lake Poinsett and SR 50 sampling regions are being considered as potential intake locations for only one withdrawal structure that will provide inflow to Taylor Creek Reservoir (Lowe et al. 2012b). If the potential intake structure location at SR 520 is chosen, then there will not be withdrawals at SR 50; if the SR 50 bridge location is chosen, then there will not be withdrawals at SR 520.

The Lake Monroe sampling region (Figure 3) contains three potential withdrawal locations. Potential withdrawal locations include SR 46 just upstream of the Lake Jesup outflow (Figure 6), Lake Monroe (Figure 7), and Yankee Lake (Figure 8). Water depths throughout the Lake Monroe region varied widely (1.2–7.6 m), with the greatest depths occurring at sites in the river channel downstream of Lake Monroe.

The sampling region at SR 44 is the farthest downstream site and is located where SR 44 crosses the St. Johns River near DeLand, Florida (Figure 9). Water depths at the five sites sampled in this region ranged from 3.0 to 7.6 m.

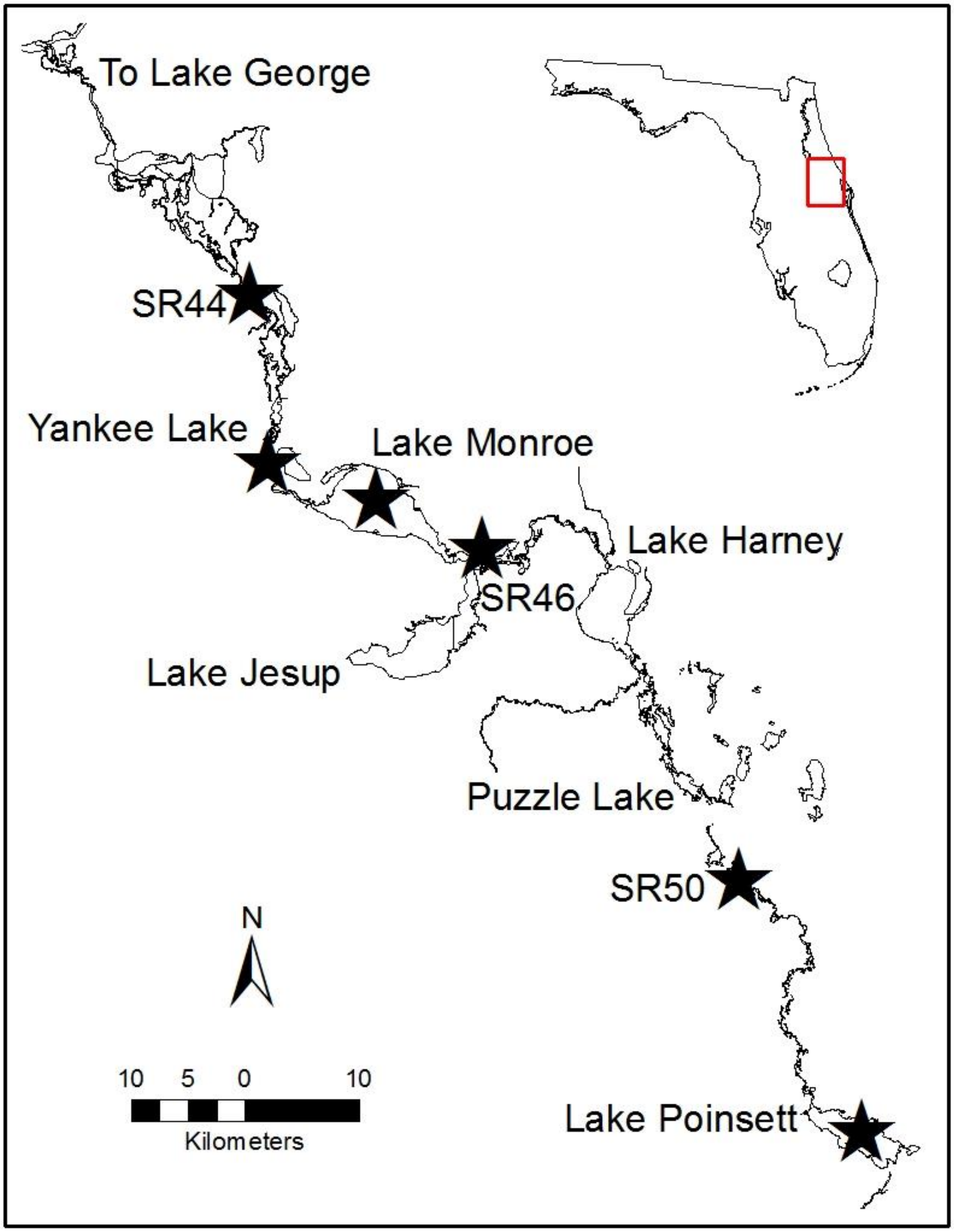



Figure 3. Ichthyoplankton sampling regions in the Middle St. Johns River Basin

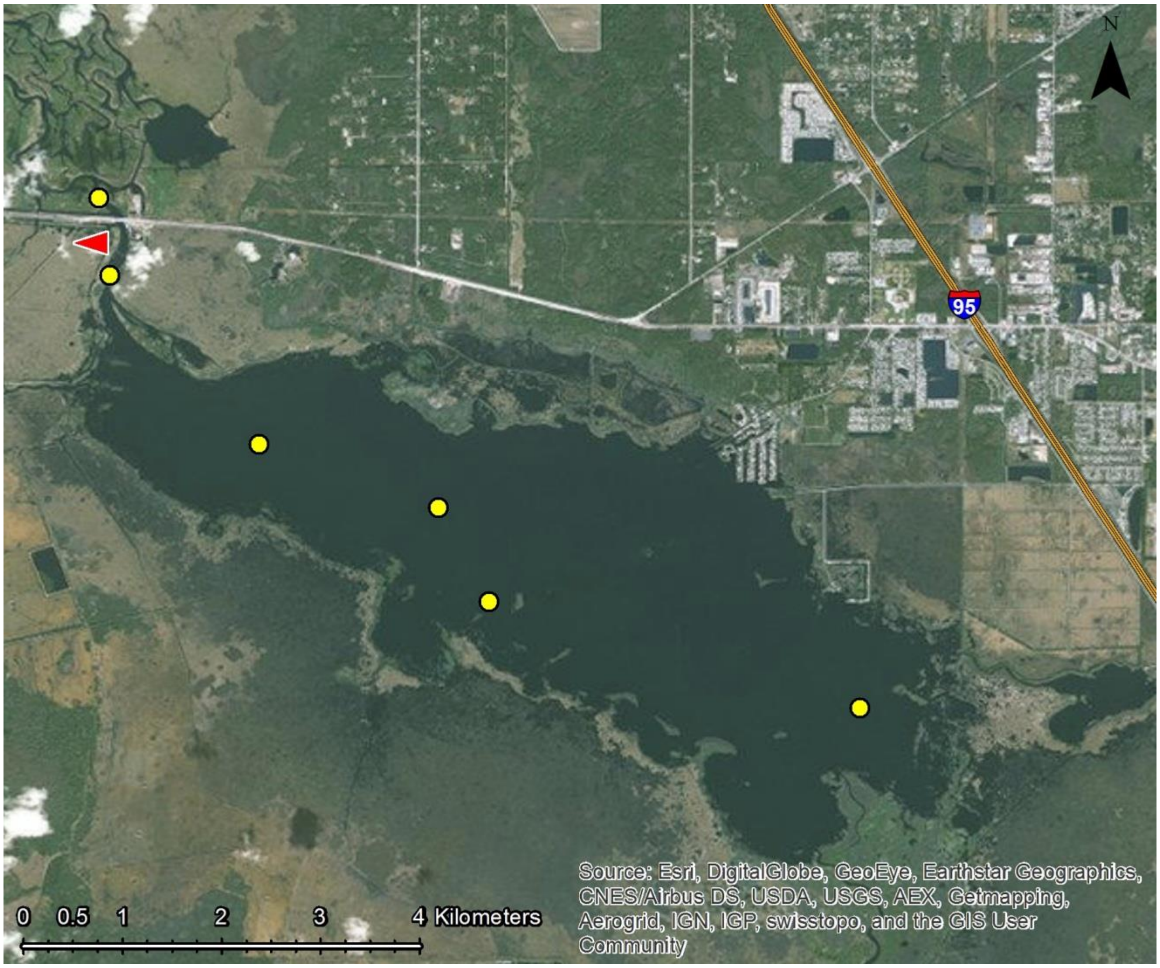



Figure 4. Ichthyoplankton sampling sites at Lake Poinsett

Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.

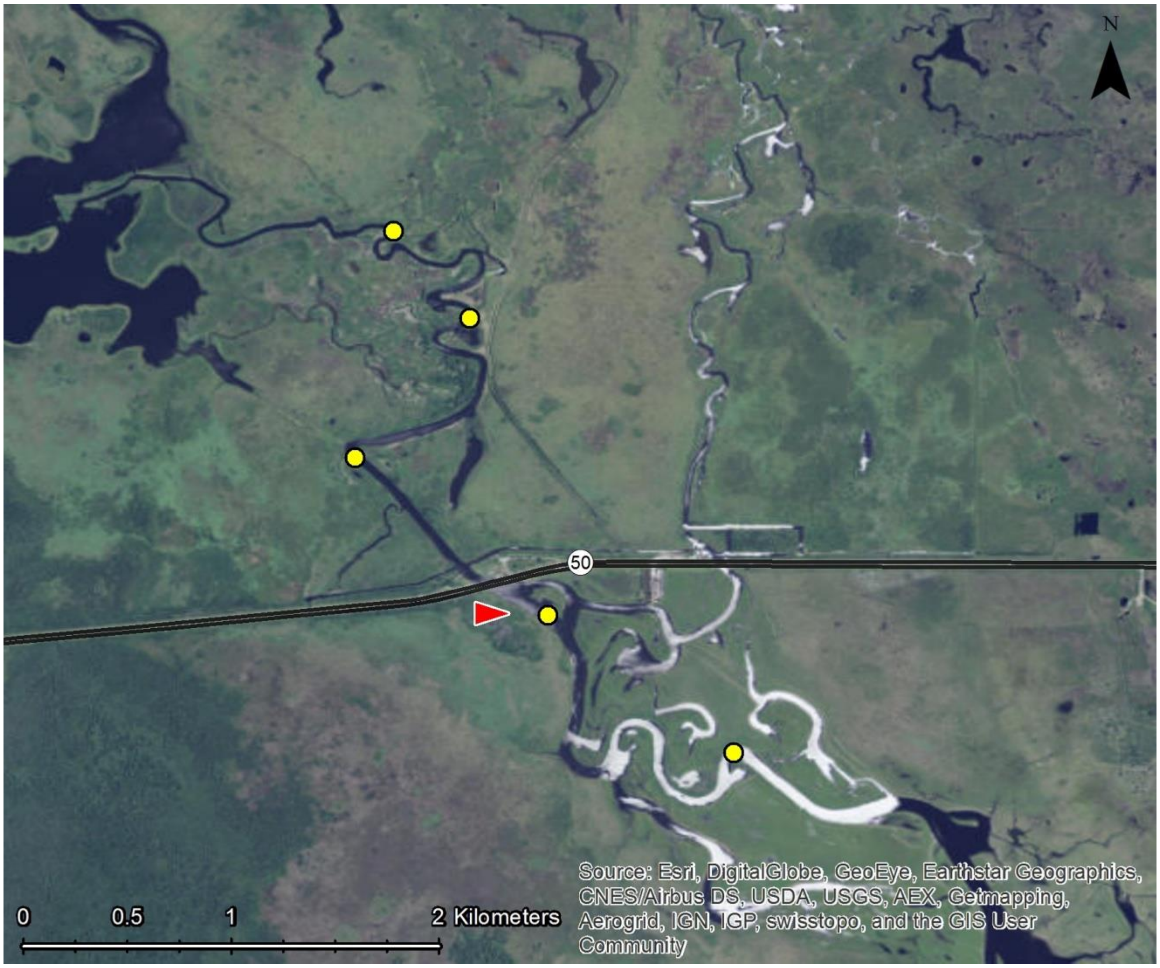



Figure 5. Ichthyoplankton sampling sites at State Road 50

Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.


Figure 6. Ichthyoplankton sampling sites near State Road 46

Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.

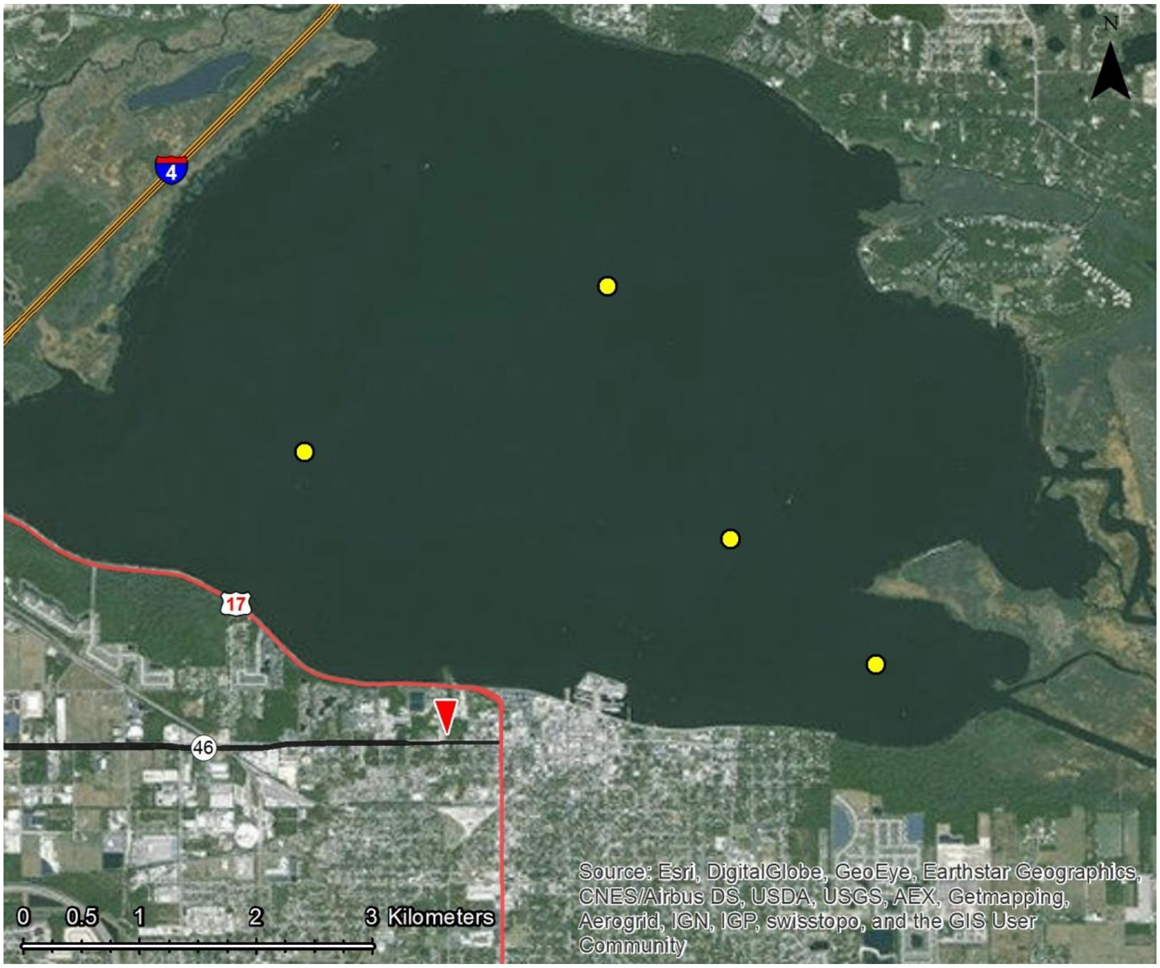



Figure 7. Ichthyoplankton sampling sites in Lake Monroe

Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.


Figure 8. Ichthyoplankton sampling sites in the St. Johns River at Yankee Lake just downstream of Lake Monroe

Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.

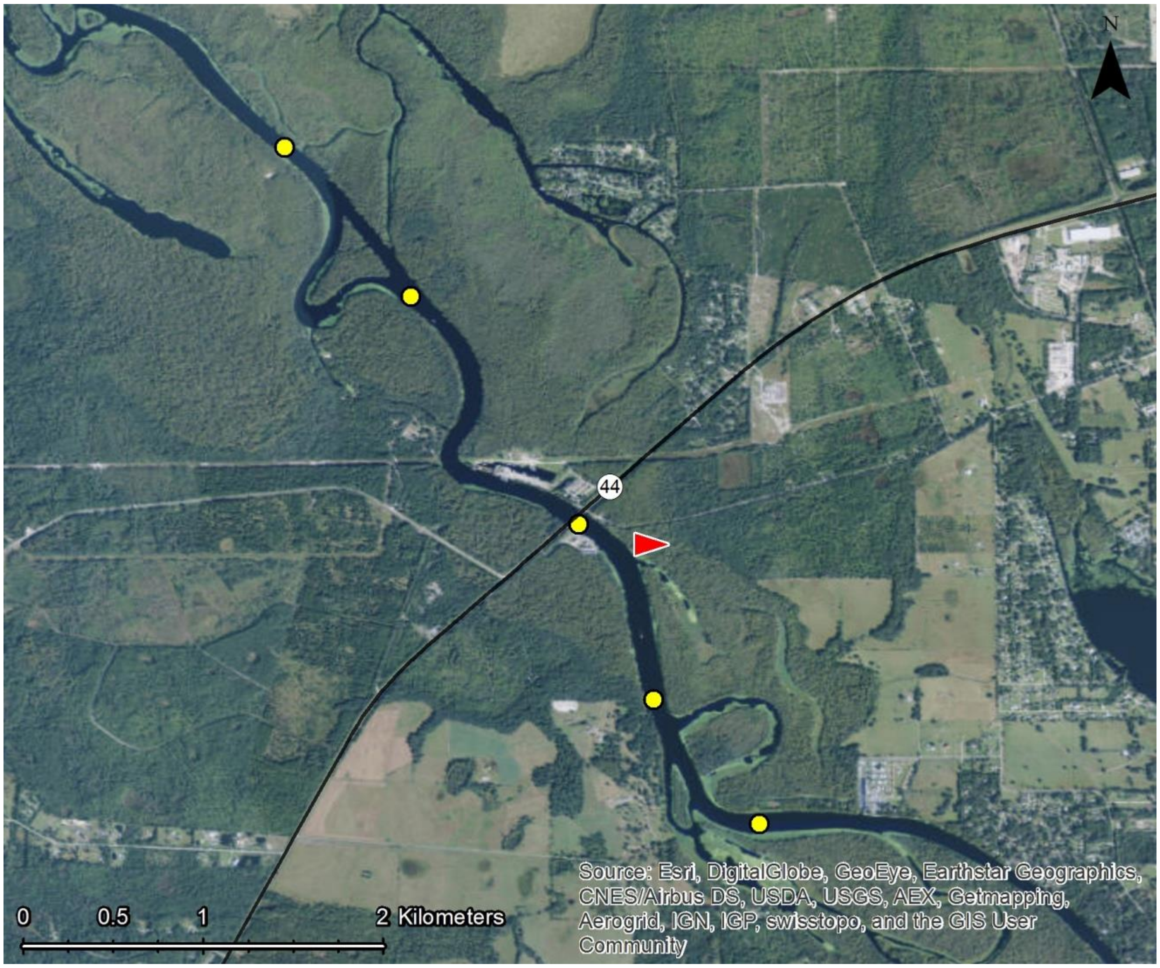



Figure 9. Ichthyoplankton sampling sites in the St. Johns River near State Road 44
Yellow shapes are sampling sites and the red arrow indicates the approximate withdrawal location.

## SAMPLING SCHEDULES

Each site in each sampling region was sampled twice weekly during the anticipated winter-spring peak spawning season and once weekly from summer through late fall. Sampling was initiated on 11 February 2008 and occurred twice weekly through the end of May. Weekly sampling occurred from June through December. In 2009, twice-weekly sampling occurred from January through May, and weekly sampling occurred from June through August. Because of very low ichthyoplankton catch rates during August of both 2008 and 2009, sampling during September 2009 was reduced to include only two sites near each of the proposed withdrawal locations and sampling frequency was reduced to once every two weeks. Sampling was terminated at the end of September 2009 due to low catch rates. Inclement weather (e.g., Tropical Storm Fay, periods of high winds, summer thunderstorms) and boat problems periodically required short-term modifications to the sampling schedule.

Most ichthyoplankton sampling was during daylight hours. However, to determine if species composition, abundance, spatial distribution, or size-structure of ichthyoplankton varied diurnally, day and night samples were collected during a single 24-hour period each month at all SR 46, Lake Monroe, and Yankee Lake sampling sites.

## SAMPLING METHODS

Ichthyoplankton samples were collected using paired 0.5-m diameter plankton nets that were 1.8 m long and constructed of 500 micrometer (µm) mesh mounted on sled frames (Snyder 1983) (Figure 10). Weighted sleds enabled sampling near the bottom without digging into the substrate. A ring welded to the upper end of the circular net frame allowed attachment of a towing cable and ensured the cable would not create a disturbance directly in front of the net. Calibrated Sea Gear® (in 2008) and General Oceanics® Model 2030 (in 2009) flowmeters were mounted in the mouth of each net to calculate the volume of water filtered.

Plankton nets were towed using outboard motor boats. Each boat was equipped with two davits located amidships, which allowed simultaneous towing of two sleds, one on each side of the boat. Davits were equipped with winches controlled by battery-operated electric drills. Specially machined sockets allowed drill attachment and replacement of manual winch handles.

At each site, paired sleds were towed upstream at approximately 1.5 m $s^{-1}$ for 10 minutes. Sleds were towed for 3-minute intervals along the bottom, at mid-depth, and at the surface to collect depth-integrated samples (Snyder 1983). After each tow, flowmeter readings were recorded from each net to calculate the volume of water filtered. Each net filtered approximately 50–100 $m^3$ of water. After retrieval, nets were rinsed from the outside, and the entire sample was washed into a cod end-collecting cup. Samples were placed in labeled jars, preserved in 10% formalin, and transported to the Sportfish Research Laboratory at the Florida Institute of Technology for processing. From January through May 2009, at least two samples collected within each sampling region were preserved in 70% ethanol to permit aging of larval American shad (*Alosa sapidissima*) from their otoliths.

At each site on each sampling date, a YSI® Model 556MPS multiparameter sonde was used to measure temperature (°C), conductivity (microohms per centimeter [µhos $cm^{-1}$]), pH, and dissolved oxygen (milligrams per Liter [mg $L^{-1}$]) within 0.6 m (2 ft) of the water's surface. A 30-cm diameter Secchi disk was lowered to measure water clarity during daytime sampling. Conductivity was later converted to salinity (in practical salinity units [psu]) using algorithms that account for temperature and atmospheric pressure at sea level (Fofonoff and Millard Jr. 1983).

Figure 10. Plankton net (0.5-m diameter, 500-µm mesh) mounted on a towing sled used to collect ichthyoplankton samples from the St. Johns River. A mounted flowmeter is visible in the bottom of the net opening

## SAMPLE PROCESSING

Samples remained in formalin for at least 4 days to ensure complete fixation. After fixation, each sample was water rinsed and then stored in a 70% ethanol/water mixture. Only one of the samples from the paired nets towed at each site was selected for analysis, while the second was archived.

A team of trained students at the Florida Institute of Technology processed the samples. To initiate sample processing, large larvae and juvenile fishes were first removed and stored in labeled vials. If large numbers of smaller larvae were present after the larger individuals were removed, or if there was a substantial amount of detritus present, samples were split into two equal subsamples using a Folsom plankton splitter (Snyder

1983). If large numbers of larvae were still present, additional subsampling further reduced the sample volume into halves, quarters, eighths, or sixteenths of the original sample. Subsampling occurred until at least two of the subsamples contained approximately 100–200 larvae. A stereo-zoom dissecting microscope was used to remove and count small larvae from each sample or subsample. All fish eggs and larvae were stored in labeled vials for identification and measurement. The processing date, name of the sample processor, subsampling factor, and total numbers of fish eggs, larvae, and juveniles were recorded on laboratory processing data sheets.

Identification of larval fishes is a difficult and slow process where minor differences in morphology and pigment among closely related species, as well as the changes that occur during ontogeny must be considered (Snyder 1983). Comprehensive identification manuals for the fish species that occur in the St. Johns River do not exist, so existing reports from other studies (e.g., Jones et al. 1978) were used as the basis for development of a photographic identification manual specific to this study. As specimens were identified, they were photographed and measured. Images were then added to a photographic atlas, along with the descriptive features (e.g., myomere counts and other morphometric features) needed to identify the various life stages of each species. Because eggs of many species in the St. Johns River are presently impossible to differentiate, they were counted, but not identified.

Larvae were identified, sorted, and counted from each labeled vial using a dissecting microscope. The lead taxonomist on the project, Matthew Scripter, reviewed and confirmed each species identified. Total length (TL), standard length (SL), and developmental stage were recorded for at least 25 individuals of each species present in a sample. Designated developmental stages were yolk-sac larvae (newly hatched and still subsisting on endogenous yolk reserves), preflexion larvae (early feeding larvae with a straight notochord, prior to development of the caudal fin supporting structures), postflexion larvae (older larvae with upward flexion of the posterior portion of the notochord and a caudal fin), and juveniles (with fully developed fins and pigmentation) (Snyder 1976; Snyder and Muth 2004).

All species data for each sample were archived on laboratory processing sheets, along with the date of final processing and the name of the sample processor.

Extensive Quality Assurance/Quality Control (QA/QC) procedures were developed for each step of laboratory processing and data entry. The QA/QC procedures focused on assessing the accuracy of initial sample counts, species identification, and larval measurements, as well as on the accuracy of data entered into a Microsoft Access database.

## DATA ANALYSIS

Densities of each species in a sample were calculated by multiplying subsample counts by the subsample factor and dividing by the volume of water filtered. Larval densities were then standardized to number of fish per100 $m^3$ of water. Mean (± Standard Deviation [SD]) densities for each taxon was determined for each sampling date for all sampling sites associated with each of the six proposed water withdrawal locations. Plots of combined larval density over time at each of these locations define the spatial distribution and densities of larvae within sampling regions.

Monthly plots of species catch describe the temporal occurrence of larvae in the vicinity of each of the proposed water withdrawal structures. Life stage and length-frequency data for selected species were also plotted to characterize the age composition of the larvae present in the river.

## ESTIMATION OF POTENTIAL ENTRAINMENT

Estimating potential egg and larval entrainment due to water withdrawals requires information on the morphometry of the water body, intake structure design, inflow rates, flow rates past the structure, and the distribution, abundance, and timing of occurrence of entrainable fish life stages (Boreman et al. 1981). Entrainment vulnerability of larvae and juvenile fishes is directly related to their maximum swimming speeds and inflow velocities in the immediate vicinity of the intake structure (Gowan et al. 1999). Without knowledge of the specific intake structure designs, entrainment estimates cannot be adjusted for age, size, or the swimming speed of different species and life stages of fishes.

Lacking design specifications for structures that might withdraw water from the St. Johns River, a functional definition of vulnerability to entrainment was used: any fish caught by the 0.5-m diameter plankton nets would be potentially subject to entrainment into a water withdrawal structure.

For each species, total larval transport past an intake location for each sampling interval was calculated by multiplying the mean density of larvae in the vicinity of that location on each sampling date by the total river discharge near the location for that interval. Daily discharge data were obtained from U.S. Geological Society (USGS) water gage stations located nearest the intake location. For Lake Monroe, where discharge across the lake was not measurable, entrainment estimates were calculated simply using daily density data and projected water withdrawal rates. For Lake Poinsett, entrainment estimates were generated using discharge data and larval densities from the two stations in the river just downstream of the lake (Figure 4).

Because sampling occurred at 3 to 14 day intervals, the mean daily larval density around each withdrawal location on each sampling date was used to span the duration between the midpoints of each sampling interval so the total numbers of larvae drifting past the location during that interval could be estimated. Proposed water withdrawal rates for each withdrawal location were then used to calculate the percentage of the total river discharge that would be subject to removal if the withdrawals had been occurring. Multiplying mean ichthyoplankton density by the volume of water removed generated the number, and percent, of the larvae in the river at each withdrawal location that could potentially be entrained into a withdrawal structure

Potential withdrawal rates used to estimate entrainment for each withdrawal location are presented in Table 1 (Cera et al. 2012). Two withdrawal rates (full and half) were evaluated for all stations except Lake Poinsett and SR 50, where the full withdrawal scenarios were considered. The full withdrawal rates projected for SR 50 varied with river discharge, ranging from 6.5 mgd (10 cubic feet per second [cfs]) at a river discharge of 200.4 mgd (310 cfs) to 84 mgd (130 cfs) at river discharges >439.5 mgd (>640 cfs; Table 1; Cera et al. 2012). Below a river discharge of 193.9 mgd (300 cfs) at SR 50, withdrawals would not occur (Cera et al. 2012). Both the Lake Poinsett and SR 50 locations are being considered as withdrawal sites for Taylor Creek Reservoir; only one of these withdrawal locations will be selected. Withdrawal rates modeled at SR 50, if implemented at Lake Poinsett, would cause a much larger percentage of the total flow at Lake Poinsett to be withdrawn, especially under lower flow conditions (Miller et al. 2012). Thus, withdrawal triggers and withdrawal rates at various discharges at SR 50 (Cera et al. 2012) were extrapolated to calculate corresponding discharges at Lake Poinsett using regressions relating discharge at the USGS gaging station at SR 50 to the USGS gaging station at SR 520 (Table 1). The USGS gaging station at SR 520 is located adjacent to the proposed Lake Poinsett withdrawal site. Extrapolated discharges from SR 50 to SR 520 indicate that the withdrawal cutoff at Lake Poinsett would be at a river discharge of 155.1 mgd (240 cfs) as opposed to the 193.9 mgd (300 cfs) river discharge cutoff at SR 50 (Table 1). Maximum withdrawal rates of 84 mgd (130 cfs) would occur when discharges out of Lake Poinsett exceed 341.3 mgd (528 cfs) as compared to maximum withdrawals occurring at a river discharge of 413.6 mgd (640 cfs) at SR 50 (Table 1).

Table 1. Projected water withdrawal rates and corresponding river discharges at each proposed withdrawal location along the St. Johns River used to estimate potential ichthyoplankton entrainment

| Station* | River Discharge Rate (mgd [cfs]) | Projected Water Withdrawal Rates (mgd [cfs]) |
|---|---|---|
| Lake Poinsett | <115.1 (<240) | 0 |
| | 160.9 (249) | 6.46 (10) |
| | 166.7 (258) | 12.92 (20) |
| | 172.6 (267) | 19.36 (30) |
| | 177.7 (275) | 25.85 (40) |
| | 183.6 (284) | 32.31 (50) |
| | 189.4 (293) | 38.78 (60) |
| | 194.5 (301) | 45.24 (70) |
| | 200.4 (310) | 51.70 (80) |
| | 206.2 (319) | 58.16 (90) |
| | 212–135.1 (328–509) | 64.63 (100) |
| | 329.6 (510) | 71.09 (110) |
| | 335.4 (519) | 77.55 (120) |
| | 341.3 (≥528) | 84.02 (130) |
| SR 50 | <193.9 (<300) | 0 |
| | 200.4 (310) | 6.46 (10) |
| | 206.8 (320) | 12.92 (20) |
| | 213.3 (330) | 19.36 (30) |
| | 219.7 (340) | 25.85 (40) |
| | 226.2 (350) | 32.31 (50) |
| | 232.7 (360) | 38.78 (60) |
| | 239.1 (370) | 45.24 (70) |
| | 245.6 (380) | 51.70 (80) |
| | 252.1–387.8 (390–600) | 58.16 (90) |
| | 394.3 (610) | 64.63 (100) |
| | 400.7 (620) | 71.09 (110) |
| | 407.2 (630) | 77.55 (120) |
| | 413.6 (640) | 84.02 (130) |
| SR 46 | Any level | 25 and 50 (38.7 and 77.4) |
| Lake Monroe | Any level | 10 and 20 (15.5 and 30.9) |
| Yankee Lake | Any level | 25 and 50 (38.7 and 77.4) |
| SR 44 | Any level | 25 and 50 (38.7 and 77.4) |

Source: (Cera et al. 2012)

*Stations are arranged in an upstream (south) to downstream (north) direction.

## CALCULATION OF EQUIVALENT ADULT ESTIMATES

For most freshwater fish species in the St. Johns River, age-specific mortality rates (e.g., survival from age 0 to age 1) are unknown. Thus a simplistic EPM approach was used to estimate adult equivalents first described by Horst (1975): populations are assumed to be at equilibrium, there are no compensatory mechanisms acting on the population, and the fecundity produced by a breeding pair adults over their lifetime results in the average survival of two breeding adults to the future population (Dey 2002). Thus, overall average survival across a generation can be described as:

$$S_{e\to a} = \frac{2}{f_a} \qquad \text{[Eq. 1]}$$

where, $S_{e\to a}$ is the overall survival from egg to adult, 2 is the average number of surviving adults, and $f_a$ is the lifetime fecundity of a breeding pair.

Mortality rates from eggs to adults ($M_{e\to a}$) can be described as:

$$M_{e\to a} = 1 - S_{e\to a} \qquad \text{[Eq. 2]}$$

If a known number of eggs are entrained ($NE_{eggs}$), the number of equivalent adults ($NA$) lost to the future populations can be described as:

$$NA = \frac{NE_{eggs}}{f_a} \times 2 \qquad \text{[Eq. 3]}$$

As an example, consider a hypothetical fish species that has an average lifetime fecundity ($f_a$) of 100,000 eggs. If 1,000,000 eggs of this species were entrained, 20 adults (10 females and 10 males) would be needed to produce the estimated 1,000,000 larvae:

$$20 = \frac{1{,}000{,}000}{100{,}000} \times 2$$

Thus the equivalent adults ($NA$) lost to the future population due to the entrainment of 1,000,000 eggs equals 20.

However, the number of entrained larvae actually reflects a higher number of eggs produced because mortality has reduced the population prior to entrainment. Thus, to calculate the number of equivalent adults represented by a given number of larvae ($NE_{larvae}$), the number of egg equivalents ($NE_{eggs}$) were back calculated using estimated daily instantaneous survival rates ($1 - M_{e\to l}$) and the estimated mean age of the larvae ($a$). This relationship can be described as:

$$NE_{eggs} = \frac{NE_{larvae\,(n)}}{(1 - M_{e \to l})^{a}} \quad \text{[Eq. 4]}$$

where, $NE_{larvae\,(n)}$ equals the number larvae alive on day $n$, $M_{e \to l}$ equals the daily instantaneous mortality rate, and $a$ equals the age of the larvae in days.

As an example, if 1,000,000 larvae were entrained, their average age from hatching was 5 days, and their daily instantaneous mortality rate was 10%, then the number of egg equivalents would be 1,693,509:

$$1{,}693{,}509 = \frac{1{,}000{,}000}{(1 - 0.1)^{5}}$$

If an $(f_a)$ of 100,000 eggs is assumed, this equates to an $NA$ loss of 34 individuals:

$$33.8 = \frac{1{,}693{,}509}{100{,}000} \times 2$$

Daily instantaneous mortality rates experienced during the early life history stages of St. Johns River fishes are undoubtedly highly variable both temporally and spatially. Therefore, the three hypothetical mean daily mortality rates (5%, 10%, and 25%) shown in Figure 2 were used to back calculate $NE_{eggs}$ from $NE_{larvae}$. These rates encompass the majority of the mortality rates for recently hatched larvae reported in the literature (Dahlberg 1976).

Average larval age-at-catch for each species was determined from modal size groups in the length-frequency distributions. Literature values on daily larval growth rates and the number of days between hatching and the initiation of exogenous feeding were used to estimate age from size.

Average fecundity estimates for individual species were determined from the literature. Since fecundity varies with size, a best estimate of the fecundity of an average mature female was used. Lower and upper fecundity boundaries were established based on spawning potential over several years. However, for species that only spawn during a single year (e.g., American shad and gobies) fecundity boundaries were based on potential spawning during a single year. It is noteworthy that the lower fecundity estimates produce higher estimates of adult loss. Therefore $NA$ calculated from the lower fecundity boundary likely represents a worst-case scenario that likely overestimates adult loss.

Finally, where possible, $NA$ for each species was compared to standing stock and sport harvest information reported for the river by the Florida Fish and Wildlife Conservation Commission (FWC) and to values reported for other lakes in Florida (Hoyer and Canfield

1994). This provided a more general population impact assessment of water withdrawal effects (Horst 1975).

# RESULTS

## ENVIRONMENTAL DATA

River discharges used to calculate entrainment at each of the six locations proposed as water withdrawal sites are presented in Figure 11. Discharges increased moving downstream from Lake Poinsett, and wind events along with tidal influence can be seen causing periodic reverse flows at Yankee Lake and SR 44. Discharges in 2008 and 2009 followed a tropical pattern with the lowest discharges occurring during late winter and spring dry season (January–June) and the highest discharges occurring during the August–October wet season (Figure 11).

High discharges that increased rapidly in August–September 2008 reflect the passing of Tropical Storm Fay, which dropped over 60 cm of rainfall on parts of the basin (Figure 11). Tropical Storm Fay caused extensive flooding throughout the middle basin, and discharges did not return to normal levels until December.

Average monthly chemical-physical data collected near each of the proposed water withdrawal locations are presented in Figure 12. Water temperatures ranged between 15 degrees Celsius (°C; 59 degrees Fahrenheit [°F]) and 32°C (90°F) and only varied by a few degrees among locations during any individual month. Interestingly, in 2008, the average monthly water temperatures from February through May were lowest at Lake Poinsett, which is our southernmost sampling location (Figure 12). Average monthly salinities at all locations ranged between 0.2 and 1.0 psu (Figure 12). Salinity was generally lowest at Lake Poinsett and highest at SR 46 and SR 50. Salinity was lowest during wet season periods of higher discharge and lowest during the dry season. Dissolved oxygen (DO), pH, and water depth also followed similar seasonal patterns (Figure 12). Water clarity, as measured by Secchi depth, ranged between 0.3 and 1.2 m and was highly variable, exhibiting only weak seasonal patterns (Figure 12).

Concomitant with the passing of Tropical Storm Fay in August 2008 and the significant increase in discharge, there were substantial drops in DO, salinity, and pH at all locations (Figure 12).

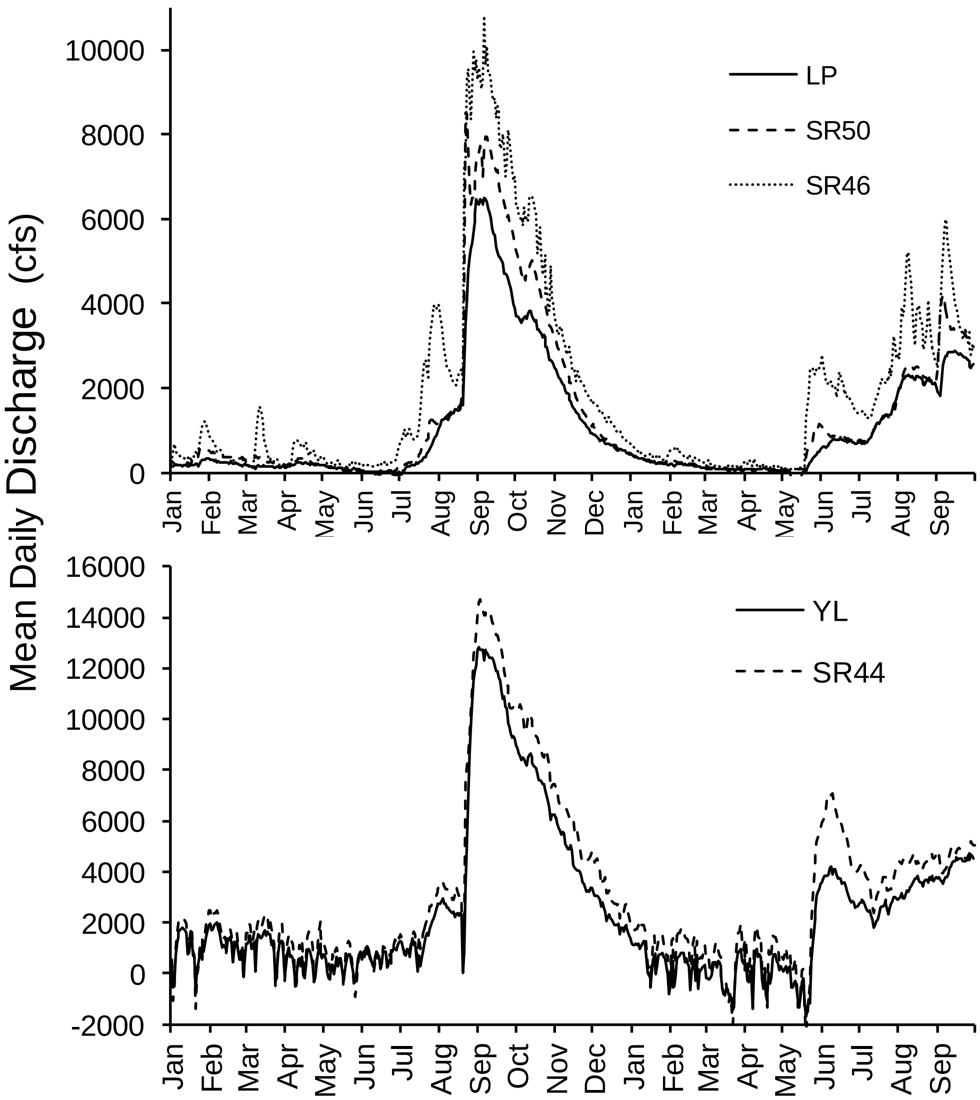



Figure 11. Mean daily discharge (cfs) at Lake Poinsett (LP; USGS 02232400), SR 50 (USGS 02232500), SR 46 (USGS 02234440), Yankee Lake (YL; USGS 02234500), and SR 44 (USGS 02236000) gaging stations from 1 January 2008 through 30 September 2009

Lowest average monthly DO levels (<1.0 mg $L^{-1}$) in August 2008 were recorded at SR 50 and at SR 44 (Figure 12). Although average monthly DO in August 2008 ranged between 2.0 mg $L^{-1}$ and 6.0 mg $L^{-1}$ at all the other locations, there were several days between 29 August and 3 September when surface DO at all these locations fell to <0.8 mg $L^{-1}$. The sudden intense drops in DO to <1.0 mg $L^{-1}$ throughout the basin following Tropical

Storm Fay resulted in numerous fish kills along the river (B. Eisenhauer, FWC, pers. comm. 2009).

## ICHTHYOPLANKTON CATCH

During the 18 months from February 2008 through September 2009, 6,204 ichthyoplankton samples were taken over 266 sampling dates. Of the 6,204 samples, 2,991 were processed. This sampling intensity is far greater than has been generally accomplished in other freshwater larval fish surveys, providing a detailed description of the larval distributions in the St. Johns River that is unparalleled by other studies.

Processed samples yielded a total of 8,071 eggs, 709,603 larvae and 19,086 juvenile/adult fishes representing 47 identifiable taxa (Table 2). Gizzard shad and threadfin shad numerically dominated the catch at all stations, comprising 35.4% and 34.2% of the total larvae collected, respectively. Gobies (primarily clown goby and naked goby) were the next most abundant larval fishes, comprising 16.4% of the total catch. Dominant species of commercial and recreational importance were black crappie *(*2.9% of the larval catch), bluegill sunfish (2.2%), and American shad (1.9%).

Total larval abundance varied widely among seasons and years around each withdrawal location (Figure 13). Densities of larval fishes at all locations exhibited the same general abundance patterns with the highest catch occurring from February through June. Very few larval fishes at any location were collected from August through November.

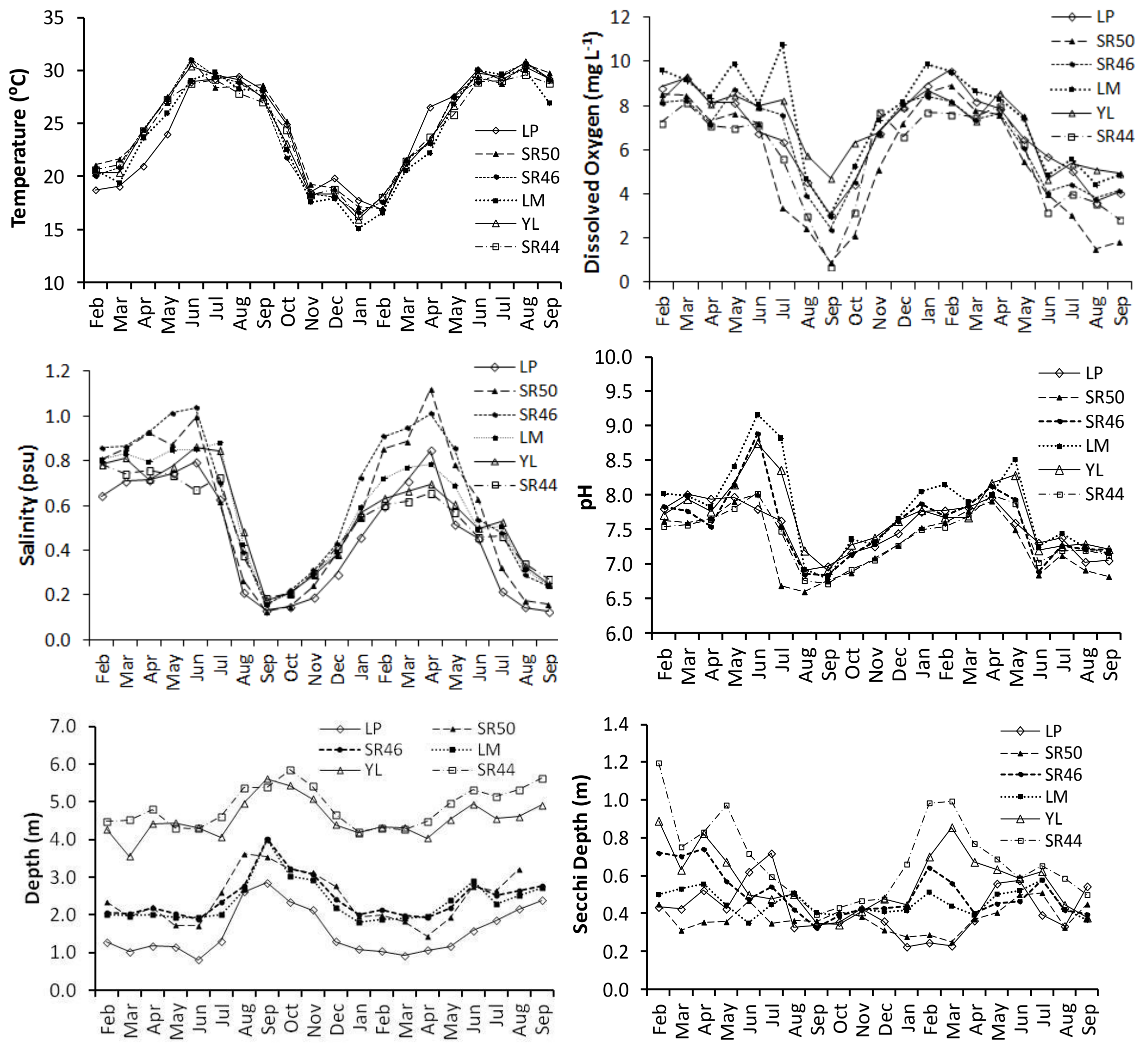



Figure 12. Average monthly water temperature, dissolved oxygen (DO), salinity, pH, water depth, and Secchi depth collected within sampling regions associated with potential water withdrawal locations (February 2008–September 2009)

Table 2. Total catch of ichthyoplankton and juvenile and adult fishes in the St. Johns River from February 2008 through September 2009

| Scientific Name | Common Name | Total Larvae | Total Juv./Adult | Total |
|---|---|---|---|---|
| *Dorosoma cepedianum* | gizzard shad | 251,367 | 868 | 252,235 |
| *Dorosoma petenense* | threadfin shad | 243,099 | 1,617 | 244,716 |
| *Microgobius gulosus* | clown goby | 50,187 | 547 | 50,734 |
| *Gobiosoma bosc* | naked goby | 49,995 | 437 | 50,432 |
| *Pomoxis nigromaculatus* | black crappie | 20,666 | 100 | 20,766 |
| *Dorosoma* spp. | shad spp. | 20,748 | 0 | 20,748 |
| *Menidia beryllina* | tidewater silverside | 15,069 | 1,531 | 16,600 |
| Gobiidae | goby spp. | 16,326 | 0 | 16,326 |
| *Lepomis macrochirus* | bluegill sunfish | 15,648 | 227 | 15,875 |
| *Alosa sapidissima* | American shad | 13,729 | 24 | 13,753 |
| *Gambusia holbrooki* | mosquitofish | 0 | 11,355 | 11,355 |
| *Lepomis microlophus* | redear sunfish | 5,016 | 152 | 5,168 |
| *Etheostoma fusiforme* | swamp darter | 4,250 | 7 | 4,257 |
| *Ameiurus catus* | white catfish | 652 | 901 | 1,553 |
| *Notropis maculatus* | taillight shiner | 261 | 241 | 502 |
| *Membras martinica* | rough silverside | 468 | 24 | 492 |
| *Pterygoplichthys multiradiatus* | sailfin catfish | 69 | 358 | 427 |
| *Hoplosternum littorale* | brown hoplo catfish | 363 | 58 | 421 |
| *Ictalurus punctatus* | channel catfish | 236 | 128 | 364 |
| *Alosa aestivalis* | blueback herring | 297 | 0 | 297 |
| *Opsopoeodus emiliae* | pugnose minnow | 104 | 121 | 225 |
| *Alosa mediocris* | hickory shad | 214 | 3 | 217 |
| *Fundulus seminolis* | Seminole killifish | 141 | 27 | 168 |
| *Strongylura marina* | Atlantic needlefish | 97 | 61 | 158 |
| *Lepomis gulosus* | warmouth | 135 | 9 | 144 |
| *Labidesthes sicculus* | brook silverside | 115 | 2 | 117 |
| *Lepomis* sp. | Sunfish spp. | 86 | 0 | 86 |
| *Lucania goodei* | bluefin killifish | 34 | 49 | 83 |
| *Heterandria formosa* | least killifish | 13 | 62 | 75 |
| *Notemigonus crysoleucas* | golden shiner | 54 | 5 | 59 |
| *Poecilia latipinna* | sailfin molly | 13 | 45 | 58 |
| *Jordanella floridae* | American flagfish | 7 | 50 | 57 |
| *Syngnathus scovelli* | gulf pipefish | 25 | 21 | 46 |

**Table 2—*Continued***

| Scientific Name | Common Name | Total Larvae | Total Juv./Adult | Total |
|---|---|---|---|---|
| *Oreochromis aureus* | blue tilapia | 19 | 24 | 43 |
| *Micropterus salmoides* | largemouth bass | 18 | 19 | 37 |
| *Lepomis auritus* | redbreast sunfish | 21 | 0 | 21 |
| *Fundulus cingulatus* | banded topminnow | 16 | 0 | 16 |
| unidentifiable | | 16 | 0 | 16 |
| *Anchoa mitchilli* | bay anchovy | 8 | 1 | 9 |
| *Ameiurus nebulosus* | brown bullhead | 2 | 5 | 7 |
| *Enneacanthus gloriosus* | bluespotted sunfish | 7 | 0 | 7 |
| *Elassoma* sp. | pygmy sunfish | 6 | 0 | 6 |
| *Aphredoderus sayanus* | pirate perch | 0 | 4 | 4 |
| *Lepisosteus platyrhincus* | Florida gar | 0 | 3 | 3 |
| *Ameiurus natalis* | yellow bullhead | 2 | 0 | 2 |
| *Trinectes maculatus* | hogchoker | 2 | 0 | 2 |
| *Erimyzon sucetta* | lake chubsucker | 1 | 0 | 1 |
| *Etheostoma edwini* | brown darter | 1 | 0 | 1 |
| | | | | |
| Eggs | | | – | 8,071 |
| Total Larvae | | 709.603 | – | 709,603 |
| Total Juvenile/Adults | | | 19,086 | 19,086 |
| TOTAL | | 709,603 | 19,086 | 736,760 |

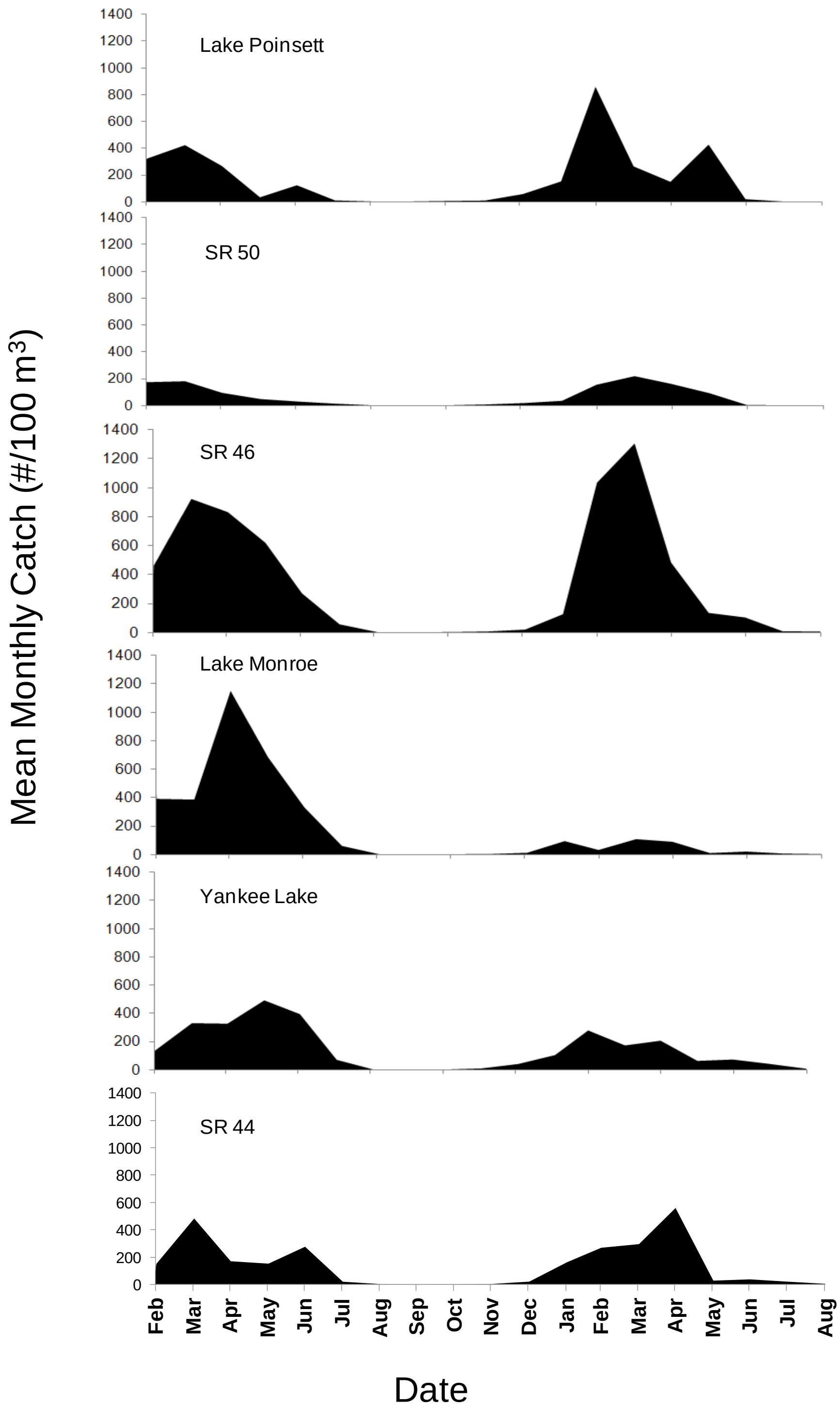



Figure 13. Monthly mean larval fish density (number 100 $m^{-3}$) at six St. Johns River potential withdrawal locations from February 2008 through August 2009.

Larval fish abundance was highest at the SR 46 and Lake Monroe locations and lowest at SR 50 (Table 3). In fact, average larval fish catch per transect at the SR 46 and Lake Monroe locations was at least twice the average catch per transect at any other site. Species composition of the larval fish catch also varied among sampling locations (Table 3). For example, goby larvae were most abundant near Lake Monroe but were rare in Lake Poinsett and at SR 50. In contrast, swamp darter were most abundant at Lake Poinsett and SR 50. Although total larval fish catch and average catch per transect was lowest at SR 50, the 11,883 American shad larvae collected at this location were more than 10-fold the number collected anywhere else (Table 3). In addition, hickory shad and blueback herring, two other important anadromous river herring species that use the freshwater reaches of the St. Johns River for spawning, were both most abundant at SR 46.

Table 3. Species composition of larval fishes collected at six St. Johns River potential water withdrawal locations from February 2008 through September 2009

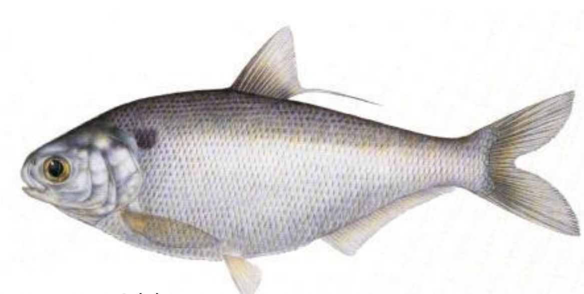
| **Scientific Name** | **Common Name** | **Lake Poinsett** | **SR 50** | **SR 46** | **Lake Monroe** | **Yankee Lake** | **SR 44** |
|---|---|---|---|---|---|---|---|
| *Dorosoma cepedianum* | gizzard shad | 38,848 | 18,237 | 102,593 | 29,884 | 28,319 | 33,049 |
| *Dorosoma petenense* | threadfin shad | 6,590 | 2,350 | 103,513 | 89,009 | 24,594 | 16,435 |
| *Dorosoma* spp. | unidentified shad | 44 | 7 | 11,296 | 831 | 13 | 4,862 |
| *Microgobius gulosus* | clown goby | 7 | 25 | 17,377 | 11,658 | 15,733 | 5,387 |
| *Gobiosoma bosc* | naked goby | 2 | 33 | 14,296 | 16,867 | 13,851 | 4,786 |
| | unidentified goby | 0 | 33 | 9,236 | 931 | 3,067 | 2,913 |
| *Pomoxis nigromaculatus* | black crappie | 3,575 | 377 | 1,939 | 1,379 | 3,596 | 9,800 |
| *Lepomis macrochirus* | bluegill | 3,095 | 519 | 5,277 | 1,656 | 1,738 | 3,337 |
| *Alosa sapidissima* | American shad | 109 | 11,883 | 674 | 493 | 313 | 295 |
| *Lepomis microlophus* | redear sunfish | 950 | 370 | 1,837 | 346 | 910 | 531 |
| *Etheostoma fusiforme* | swamp darter | 2,712 | 908 | 287 | 36 | 80 | 139 |
| *Alosa aestivalis* | blueback herring | 0 | 52 | 145 | 70 | 19 | 11 |
| *Alosa mediocris* | hickory shad | 3 | 18 | 100 | 53 | 26 | 14 |
| *Ictalurus punctatus* | channel catfish | 0 | 3 | 227 | 1 | 4 | 1 |
| *Ameiurus catus* | white catfish | 3 | 27 | 565 | 8 | 31 | 18 |
| | Others | 6,063 | 1,608 | 3,683 | 2,875 | 5,639 | 979 |
| | Totals | 61,968 | 36,450 | 273,045 | 156,097 | 97,933 | 82,557 |
| | Number of transects | 6 | 5 | 6 | 4 | 5 | 5 |
| | Average catch per transect | 10,328 | 7,290 | 45,508 | 39,024 | 19,587 | 16,511 |

Although the total catch of larval fishes was second highest at Lake Monroe, there was a substantial decline in mean monthly catch rates in the lake between 2008 and 2009 (Figure 13). This decline is attributable primarily to a large reduction in the abundance of goby larvae in the lake between the 2 years. One possible explanation for the observed reduction of gobies is that they declined following the large salinity declines in 2008 associated with the passage of Tropical Storm Fay (Figure 12). Another possible explanation is that low DO (<1.0 mg $L^{-1}$) for several consecutive days following the passage of Tropical Storm Fay resulted in high mortality (Breitburg 1992; Miller et al. 2014).

## PREDICTED ENTRAINMENT LOSSES AND ADULT EQUIVALENTS

Predicted larval entrainment losses and their adult equivalents were calculated for gizzard shad, threadfin shad, clown goby, naked goby, black crappie, bluegill, and American shad. Together these seven species comprised >95% of the total larval catch (Table 2). Black crappie and bluegill are important sport fishes in the river whereas American shad is an important migratory herring whose harvest is closely regulated by the Atlantic States Marine Fisheries Commission (ASMFC) because of recent dramatic population declines along the entire east coast of the United States.

### Gizzard Shad (*Dorosoma cepedianum*)



Adult
[image obtained from MDC 2016]

Postflexion larvae (9.7 mm TL)
Photo by Matt Scripter

Gizzard shad are a native herring of the family Clupeidae and are common in fresh and brackish waters throughout North America (Carlander 1969a). They are an important prey item for a number of recreationally important species (e.g., largemouth bass [*Micropterus salmoides*]) but because of their high reproductive potential and rapid growth they can quickly overpopulate many water bodies. Gizzard shad can attain lengths >500 millimeters (mm; 19 inches [in.]) and weigh more than 1.9 kilograms (kg; 4.0 pounds [lbs]; Carlander 1969a). Gizzard shad are found throughout the St. Johns River drainage basin and are one of the most abundant species collected in the lakes (McLane 1955).

### Temporal and Spatial Patterns of Distribution

Gizzard shad were the most abundant and ubiquitous fish collected throughout our surveys, with 251,367 larvae and 868 juveniles enumerated from processed samples. Gizzard shad spawning began in January as temperatures approached 16°C (60.8°F) and continued through June (Figure 14); peak spawning generally occurred from March through May. In 2009, spawning began slightly earlier in Lake Monroe than at any other station with peak spawning occurring in February (Figure 14). Gizzard shad larvae reached a peak density of >2,600 larvae 100 $m^{-3}$ in the river immediately downstream of Lake Poinsett during an intense plankton bloom observed in March 2009. Frequent sampling indicated that gizzard shad spawning was not continuous throughout the spawning season but occurred in pulses that may reflect fluctuations in water temperature (Buynak and Mitchell 1994). No consistent differences in total density of gizzard shad larvae (<15 mm TL) were observed between day and night samples although larger juveniles (>15 mm TL) were more often collected at night.

### Size Distributions

Length-frequency and stage-frequency data indicate that  70% of the gizzard shad larvae collected were 4–6 mm TL and 92%  were in the yolk-sac or preflexion development stage (Figure 15). Early yolk-sac and preflexion gizzard shad larvae have limited swimming capability and are most vulnerable to entrainment. The observed decline in larval abundance with increasing size larvae (Figure 15) reflects both larval mortality and the increasing ability of larger individuals to avoid capture.

Gizzard shad larvae 3–5 mm TL have a slender body form that affects retention by the sampling gear. Tomljanovich and Heuer (1986) reported that approximately 50% of 4-day old gizzard shad larvae passed through a 500-µm mesh plankton net towed at a speed of 1.0 m $s^{-1}$. Thus, density estimates likely underestimate the actual abundance of small larvae. To account for extrusion losses, a 50% correction factor was applied to the number of 3–5 mm gizzard shad larvae collected, which increased overall estimated larval gizzard shad densities by approximately 32%.

### Entrainment Losses and Adult Equivalents

Estimated numbers of gizzard shad larvae potentially entrained during 2008–2009 under full withdrawal scenarios ranged from approximately 14 million at Lake Monroe to 167 million at SR 46 (Table 4). Under full withdrawals, an estimated 6–12% of the total gizzard shad larval transported by flow past any of the individual proposed riverine intake locations would have been entrained (Table 4).

Fecundity estimates for gizzard shad are not available for St. Johns River populations so literature values were used to develop equivalent adult estimates. Kilambi and Baglin

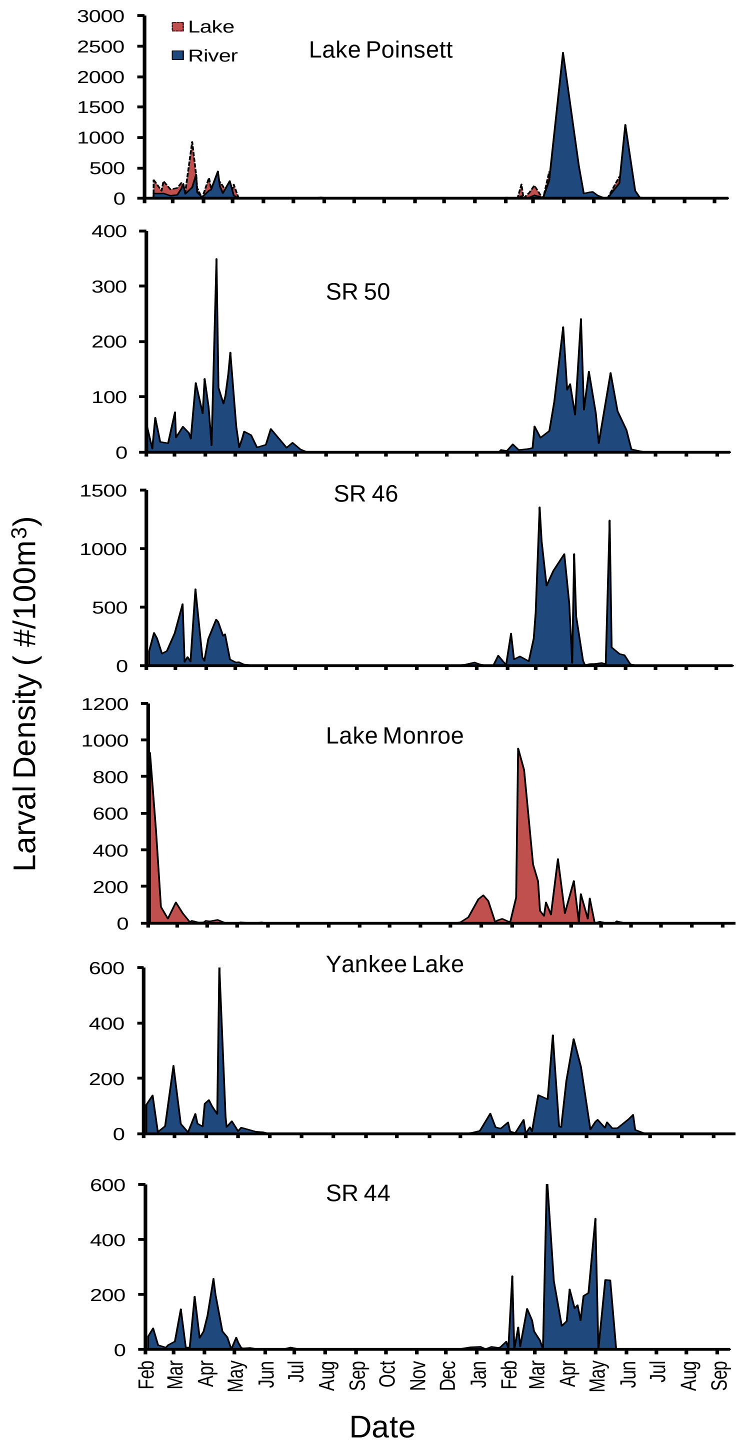



Figure 14. Mean density (number 100 m$^{-3}$) of gizzard shad larvae collected on each sampling date from February 2008 through September 2009.

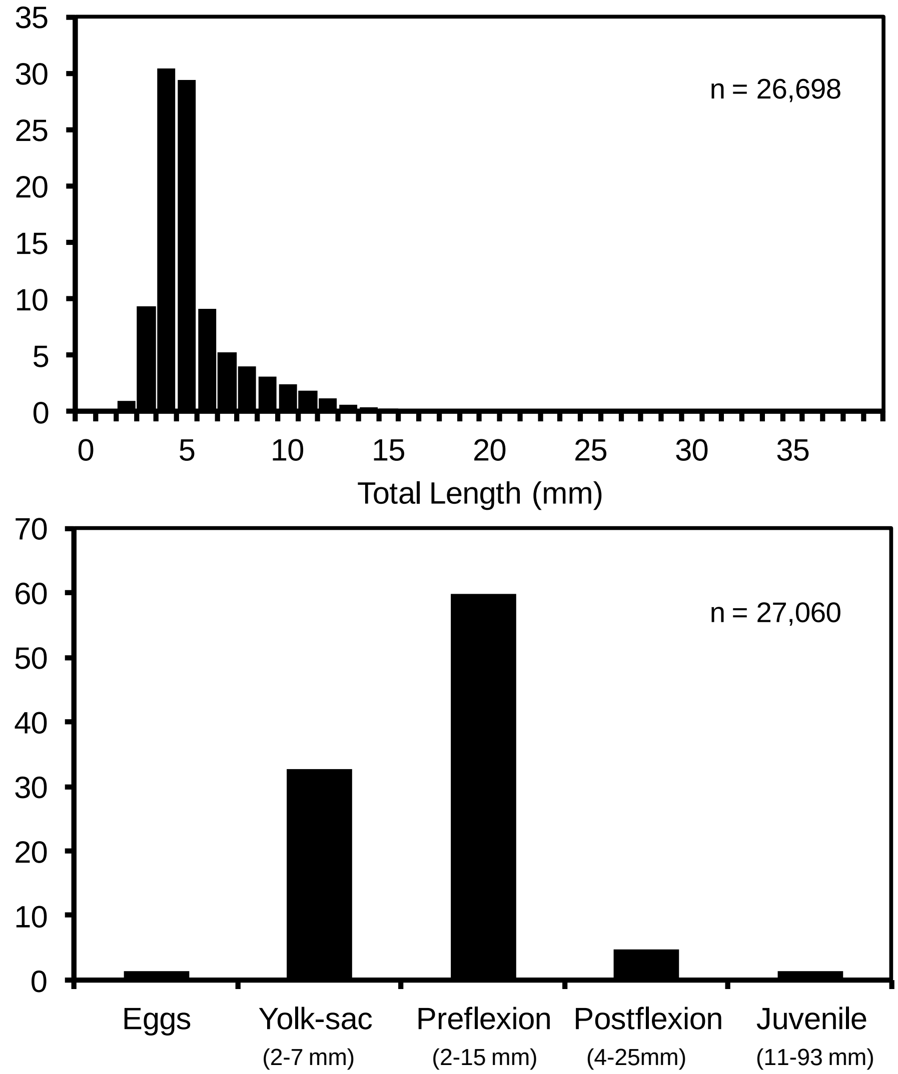



Figure 15. Length-frequency and life stage data on gizzard shad collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

Table 4. Estimates of the total number of larval gizzard shad that would have been entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimate of Total Number of Larvae Entrained (1,000's) | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | 51,429 | 12.05 |
| SR 50 | Variable | 13,038 | 9.26 |
| SR 46 | 25 mgd | 83,524 | 5.59 |
| | 50 mgd | 167,049 | 11.18 |
| Lake Monroe | 10 mgd | 14,971 | NC |
| | 20 mgd | 29,944 | NC |
| Yankee Lake | 25 mgd | 27,971 | 6.00 |
| | 50 mgd | 55,942 | 12.01 |
| SR 44 | 25 mgd | 38,995 | 3.09 |
| | 50 mgd | 77,990 | 6.18 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

(1969a) found that fecundity of gizzard shad in Arkansas ranged from 50,000 eggs in 100 gram (g) shad to 150,000 eggs in 500 g fish. Fecundity of gizzard shad in Lake Erie ranged from an average of 50,000 eggs in age 1 gizzard shad to over 500,000 in age 3 fish (Bodola 1965). As a conservative estimate, an individual average adult gizzard shad female was assumed to produce 100,000 eggs annually during each spawning episode.

Gizzard shad typically live for up to 5 years (Bodola 1965; Schramm and Pugh 1997; Fontenot 2006). Females usually do not mature until age 2, but a few may spawn at age 1 (Carlander 1969a). If an average female can spawn just once annually from age 2 to age 5, she would produce approximately 300,000 eggs in her lifetime. If the fish are batch spawners (releases multiple batches of eggs per breeding season) and spawn monthly during the 3-month main spawning season, then the female would produce approximately 900,000 eggs in her lifetime.

The modal size of gizzard shad larva collected in the St. Johns River ranged from 4 to 6 mm TL. Based on an estimated hatching size of 3.5 mm (Shelton and Stephens 1980) and a projected growth rate of about 0.6 mm $d^{-1}$ (Davis et al. 1985), modal length larvae were approximately 3 days old. Therefore, 3 days was used as the age to back calculate the number of egg equivalents ($NE_{eggs}$) that produced the observed larval catch.

Table 5. Predicted adult equivalent (*NA*) loss of gizzard shad due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | | | | | |
|---|---|---|---|---|---|---|---|
| | | Fecundity =300,000 eggs/female | | | Fecundity =900,000 eggs/female | | |
| | | Larval Mortality Rate (per day) | | | Larval Mortality Rate (per day) | | |
| | | 5% | 10% | 25% | 5% | 10% | 25% |
| Lake Poinsett | Variable | 400 | 470 | 813 | 133 | 157 | 271 |
| SR 50 | Variable | 101 | 119 | 206 | 34 | 40 | 69 |
| SR 46 | 25 mgd | 649 | 764 | 1,320 | 216 | 255 | 440 |
| | 50 mgd | 1,299 | 1,528 | 2,640 | 433 | 509 | 880 |
| Lake Monroe | 10 mgd | 116 | 137 | 237 | 39 | 46 | 79 |
| | 20 mgd | 233 | 274 | 473 | 78 | 91 | 158 |
| Yankee Lake | 25 mgd | 217 | 256 | 442 | 72 | 85 | 147 |
| | 50 mgd | 435 | 512 | 884 | 145 | 171 | 295 |
| SR 44 | 25 mgd | 303 | 357 | 616 | 101 | 119 | 205 |
| | 50 mgd | 606 | 713 | 1,232 | 202 | 238 | 411 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

The potential adult equivalent *NA* losses calculated for two fecundity scenarios (300,000 and 900,000 eggs/lifetime) at the three daily mortality rates (5%, 10%, and 25%) are presented in Table 5. The maximum predicted *NA* loss over the 2-year period was 2,640 gizzard shad at the SR 46 withdrawal location under a 50 mgd withdrawal scenario with the assumptions that an average female produced only 300,000 eggs in her lifetime and the daily larval mortality rate was 25% (Table 5). *NA* increases with lower fecundity estimates and higher larval mortality rates. The assumption that an average female gizzard shad produces only 300,000 eggs in her lifetime is likely very conservative and, therefore, provides a worst-case scenario of actual *NA* loss at all withdrawal locations.

Based on gill net data of an unexploited gizzard shad population in Lake Apopka, Florida, the average length of an adult gizzard shad (both males and females combined) in the St. Johns River was assumed to be approximately 308 mm TL with a corresponding average weight of approximately 406 g (Schramm and Pugh 1997). Multiplying *NA* loss under the worst-case scenario (fecundity = 300,000 eggs/female; daily larval mortality rate = 25%) by the average weight of an adult (406 g) provides an estimate of total biomass of adult gizzard shad potentially removed from the population by larval entrainment from February 2008 through September 2009.

Gizzard shad are abundant in the St. Johns River lakes, often comprising greater than 60% of the total fish biomass collected in pelagic haul seine samples (Cox et al. 1978; Cox et al. 1979). Abundance of gizzard shad is lowest in Lake Washington and increases moving downstream in response to increasing primary productivity. To estimate potential 2008–2009 entrainment loss effects on adult gizzard shad standing stock, the calculated biomass of adults lost, standardized by lake size (kg hectare [ha]$^{-1}$), was compared to past standing stock estimates derived from block nets. Because block net data were not available for river channel habitats, potential losses were compared to estimated standing stocks of the nearest lake(s). Estimates of gizzard shad standing stocks were also derived for each lake using mean chlorophyll values measured between 1995 and 2005 (Coveney et al. 2012) using a relationship between gizzard shad standing stock and chlorophyll derived for 22 Florida lakes (Allen et al. 2000; Table 6).

Based on littoral block net data collected between 1972 and 1996, the average standing stock of gizzard shad in Lake Poinsett was 6.1 kg ha$^{-1}$ (Table 6). This compares favorably to a standing stock estimate based on mean chlorophyll (13 µg L$^{-1}$) of 4.0 kg ha$^{-1}$. Under the worst-case scenario, 813 adult equivalent gizzard shad representing a biomass of 0.2 kg ha$^{-1}$ would have been lost to entrainment in Lake Poinsett over 2008–2009 (Table 6). Assuming a standing stock of 4.0 kg ha$^{-1}$, this represents only a 5% reduction over 2 years, or 2.5% annually. Water withdrawal at SR 50 would have resulted in entrainment losses over 2008 and 2009 that equate to only 1.3% of the estimated standing stock of Lake Poinsett (Table 6). Withdrawals at SR 46 caused the greatest *NA* loss (n = 2,640) but this loss constitutes only 3.8% of the estimated Lake Monroe annual standing stock alone and only 0.9% of the Lake Jesup standing stock estimate with the 2 years combined (Table 6). *NA* losses at SR 44 would have only resulted in a reduction of 0.2% in the gizzard shad standing stock in Lake George (Table 6).

Finally, if 50 mgd withdrawals at Lake Poinsett, 50 mgd withdrawals at SR 46, 50 mgd withdrawals at Yankee Lake, and 50 mgd withdrawals at SR 44 occurring simultaneously (200 mgd total) are assumed, coupled with the most conservative entrainment scenario (lowest fecundity and highest larval mortality), the total *NA* loss of gizzard shad due to entrainment during 2008 and 2009 would have been 5,569 individuals. This would equate to a reduction over the 2 years of only 1.9% of the estimated standing stock (based on chlorophyll) of gizzard shad in Lake Jesup or a 1.1% reduction of the estimated standing stock in Lake George. However, this worst-case estimate of adult loss based on a withdrawal rate of 200 mgd total is an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

Table 6. Mean reduction (%) of adult gizzard shad estimated standing stocks (kg ha$^{-1}$) resulting from calculated larval entrainment in 2008 and 2009 combined

| Withdrawal Location | Year | Estimated Standing Stock* (kg ha$^{-1}$) | Standing Stock Removed by 2008–2009 Entrainment (kg ha$^{-1}$) | Mean Reduction of Standing Stock Based on 2008–2009 Entrainment | Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | 1972 | 5.6 | 0.20 | 3.5% | (Cox et al. 1976) |
| | 1972 | 8.0 | 0.20 | 2.5% | (Cox et al. 1976) |
| | 1979 | 14.1 | 0.20 | 1.4% | (Cox et al. 1979) |
| | 1991 | 1.5 | 0.20 | 13.1% | (Eisenhauer et al. 1993) |
| | 1992 | 3.8 | 0.20 | 5.2% | (Eisenhauer et al. 1993) |
| | 1995 | 5.0 | 0.20 | 3.9% | (Cox et al. 1996) |
| | 1996 | 4.9 | 0.20 | 4.0% | (Cox et al. 1996) |
| | Avg. | 6.1 | 0.20 | 3.2% | |
| Lake Poinsett Chlorophyll† | Avg. | 4.0 | 0.20 | 5.0% | (Allen et al. 2000) |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Avg. | 6.1 | 0.05 | 0.8% | |
| Lake Poinsett Chlorophyll† | Avg. | 4.0 | 0.05 | 1.3% | (Allen et al. 2000) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | 1990 | 50.9 | 0.06 | 0.1% | (McDaniel and Cox 1993) |
| | 1991 | 31.3 | 0.06 | 0.2% | (McDaniel and Cox 1993) |
| | Avg. | 41.1 | 0.06 | 0.1% | |
| Lake Monroe Chlorophyll† | Avg. | 8.4 | 0.06 | 0.7% | (Hoyer and Canfield 1994) |
| **SR 46** | | | | | |
| Lake Jesup Chlorophyll† | Avg. | 28.0 | 0.31 | 0.9% | (Allen et al. 2000) |
| Lake Monroe Chlorophyll† | Avg. | 8.4 | 0.31 | 3.7% | (Allen et al. 2000) |
| **Yankee Lake** | | | | | |
| Lake Monroe Chlorophyll† | Avg. | 8.4 | 0.10 | 1.2% | (Allen et al. 2000) |
| **SR 44** | | | | | |
| Lake Monroe Chlorophyll† | Avg. | 8.4 | 0.14 | 1.7% | (Allen et al. 2000) |
| Lake George Chlorophyll† | Avg. | 10.8 | 0.03 | 0.2% | (Allen et al. 2000) |

*****Standing stocks were estimated from block net data and from a regression relating standing stock to mean chlorophyll concentrations (Allen et al. 2000). Adult loss was calculated assuming the most conservative lifetime fecundity scenario of 300,000 eggs per female and a daily larval mortality rate of 25%.

†Calculated from regression relating chlorophyll (µg L$^{-1}$) to gizzard shad standing stock (kg ha$^{-1}$) for 22 Florida lakes presented in Allen et al. (2000) where $\log_{10}(\text{biomass}) = 1.10\ (\log_{10}\text{chlorophyll}) - 0.65$ ($r^2$=0.39). Mean chlorophyll values for Lake Poinsett (13 µg L$^{-1}$), Lake Jesup (81 µg L$^{-1}$), Lake Monroe (27 µg L$^{-1}$), and Lake George (34 µg L$^{-1}$) were derived from data presented in Coveney et al. (2012).

The ecological impact of *NA* loss depends on the role of the species in the ecosystem. Gizzard shad are considered a nuisance species in the St. Johns River that can potentially have a significant negative impact on water quality and sport fish food webs, even though at smaller sizes they are important prey for a number of other fishes (Noble 1981). As a lake restoration strategy, the District currently employs commercial fishers to remove adult gizzard shad from Lake Apopka and Lake George. Given the potential negative impacts of gizzard shad, their high reproductive potential, their large standing stocks, and the relatively small numbers of adult equivalents removed, the predicted loss of gizzard shad larvae due to entrainment by water withdrawals is believed to be insignificant and not detrimental to the St. Johns River ecosystem even under the highest proposed withdrawal rates.

## Threadfin Shad (*Dorosoma petenense*)

Adult photo by Chad Thomas, Texas State University-San Marcos [image obtained from Hendrickson and Dean 2015]

Postflexion larvae (7 mm TL)
Photo by Matt Scripter

Threadfin shad are another small member of the family Clupeidae that are common in freshwater and brackish habitats throughout the United States. They are also an important forage fish for a number of recreationally important species because of their abundance and because they rarely attain a size >175 mm TL (6.9 in.; Carlander 1969a). Threadfin shad are found throughout the St. Johns River Basin and vary widely in both temporal and spatial abundance (McLane 1955). Large schools of several hundred to several thousand individuals are commonly seen swimming at the surface in tidally influenced reaches of the river in the vicinity of Lake George (McLane 1955).

Temporal and Spatial Patterns of Distribution

Threadfin shad were the second most abundant fish collected in our surveys; 243,099 larvae and 1,617 juveniles were enumerated in the processed samples. Threadfin shad larvae were collected from mid-March through early August at all stations in both years and were most abundant in May and June (Figure 16). The highest mean densities of larval threadfin shad occurred at the SR 46 and Lake Monroe stations whereas the lowest densities occurred at SR 50 (Figure 16).

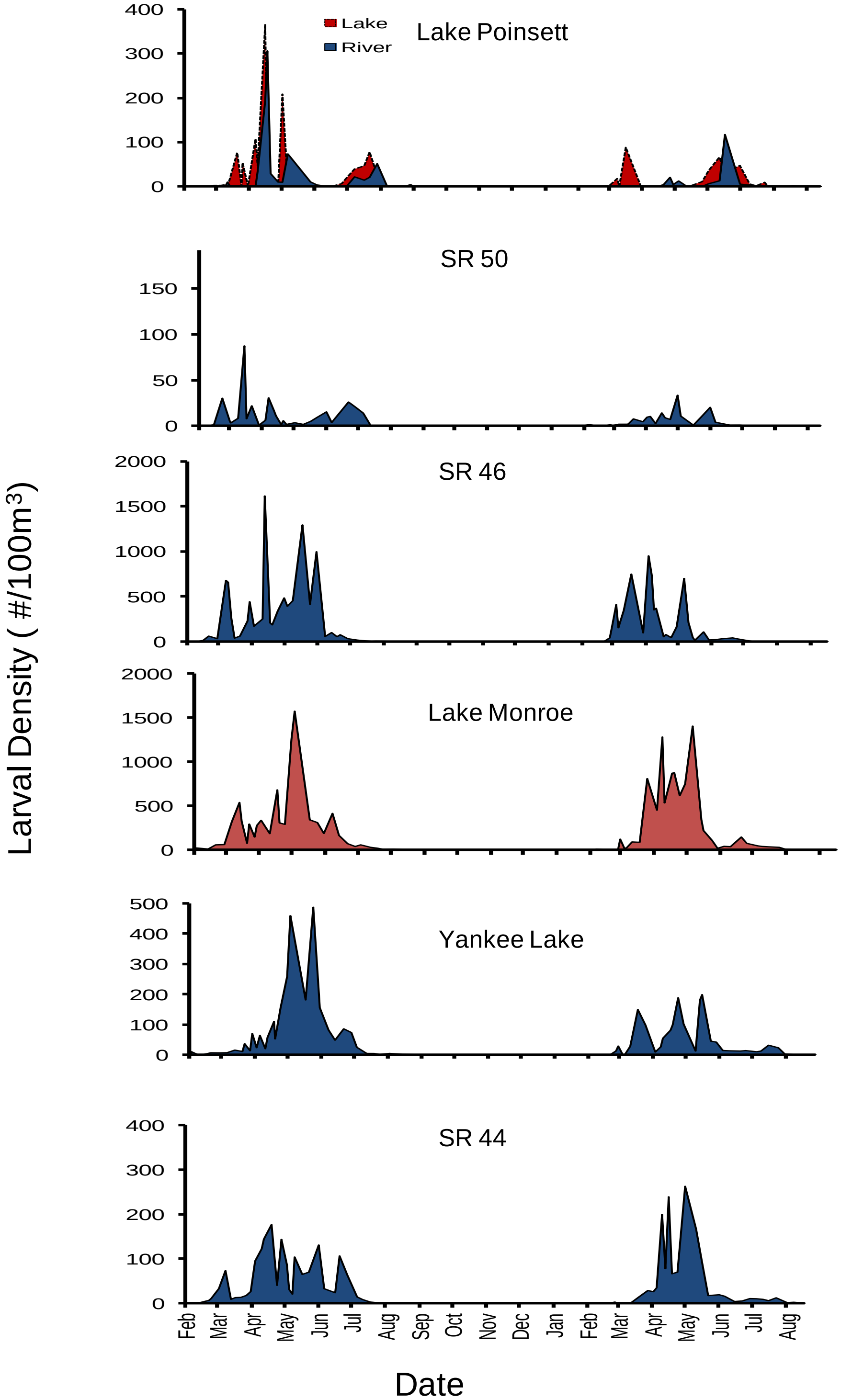



Figure 16. Mean density (number 100 $m^{-3}$) of threadfin shad larvae collected on each sampling date from February 2008 through August 2009.

## Size Distributions

Unlike gizzard shad, which had a large proportion of yolk-sac larvae, 77% of the threadfin shad larvae were in the preflexion developmental stage (4–6 mm TL; Figure 17). Yolk –sac and postflexion larvae >13 mm TL were relatively rare. As with gizzard shad, there were no consistent differences in larval abundance between daytime and nighttime samples, although more large larvae and juveniles were taken at night.

Only about 23% of the threadfin shad larvae collected were small enough (3–5 mm TL) to be subjected to exclusion through the sampling net (Tomljanovich and Heuer 1986). To account for this exclusion, overall threadfin shad densities were multiplied by a factor of 1.23 to calculate total larval numbers subject to potential entrainment.

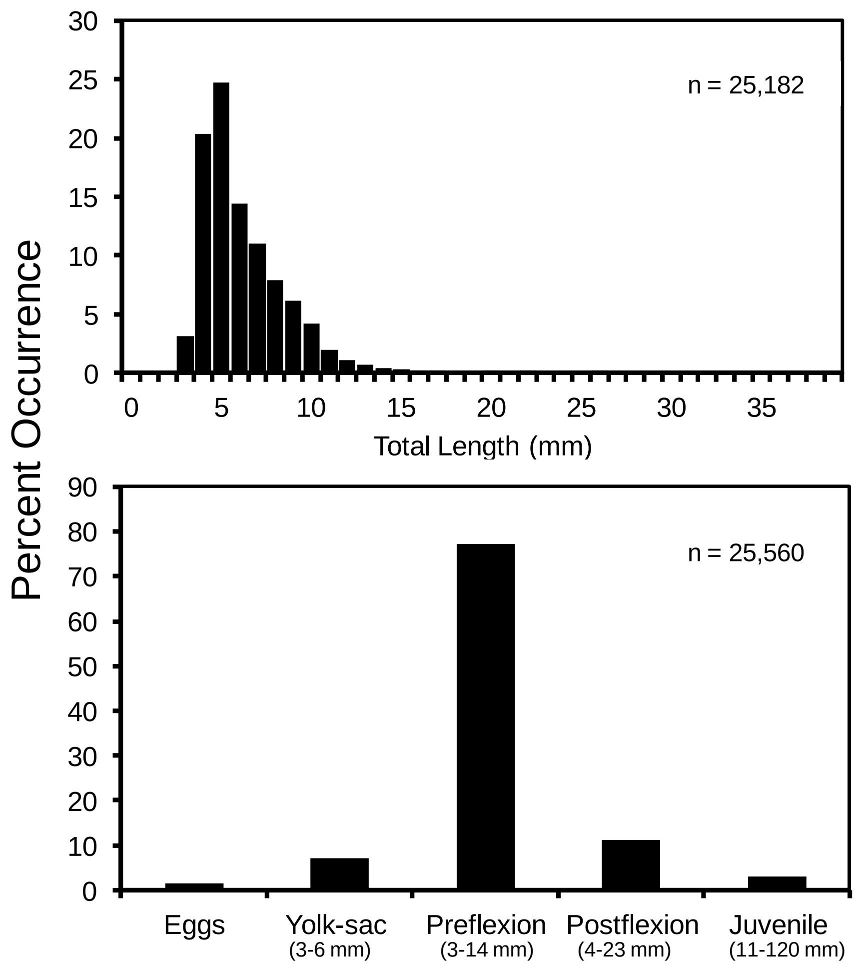



Figure 17. Length-frequency and life stage of threadfin shad collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

Entrainment Losses and Adult Equivalents

Estimated numbers of threadfin shad larvae potentially entrained during 2008–2009 under full withdrawal scenarios ranged from 1.4 million at SR 50 to 191 million at SR 46 (Table 7). At SR 46, full water withdrawals would have removed approximately 15% of the total threadfin shad larvae transported past this intake site (Table 7). At the outflow of Lake Poinsett >11% of the larval transport would have been removed. At the other locations, entrainment losses would have been between 6% and 9% of the larval transport (Table 7).

Fecundity data for threadfin shad in the St. Johns River are lacking so values derived from the literature were used for estimating *NA*. In two Arkansas reservoirs, threadfin shad fecundity ranged from 1,000 to 25,000 eggs per female with the number of eggs increasing with fish size (Kilambi and Baglin 1969b). In Arizona reservoirs, maximum fecundity was estimated to be around 8,500 eggs per female (Johnson 1971). In California reservoirs, 100 mm TL females contained between 6,700 and 12,400 eggs (Burns 1966). Individual threadfin shad females are thought to spawn only once a year although the possibility of multiple spawning events for each fish cannot be discounted given the extended spawning season. Threadfin shad generally become mature at age 2 although some individuals have been documented to spawn during their first summer of life (Heidinger and Imboden 1974; Kuklinski 2006). Threadfin shad may live up to 4 years but most do not live past age 2 (Johnson 1970).

For analysis of adult-loss equivalent scenarios, a single reproductive season during the life of each fish for two spawning scenarios (one and three spawns per year) and two total female fecundity estimates (10,000 and 30,000 eggs per female per year) was assumed.

Based on a size at hatching of 2–3 mm TL (Shelton and Stephens 1980) and a projected daily growth rate of 0.6 mm (Davis et al. 1985), captured threadfin larvae at the modal size of 5 mm TL were approximately 3–4 days old. Therefore, 3 days was used as the age to back calculate the number of egg equivalents ($NE_{eggs}$) that produced the observed larval catch.

Potential *NA* losses calculated for two fecundity scenarios (10,000 and 30,000 eggs/lifetime) at the three daily mortality rates (5%, 10%, and 25%) are presented in Table 8. The maximum predicted *NA* loss over the 2-year period was 10,048 threadfin shad at the SR 46 withdrawal location under a 50 mgd withdrawal scenario with the assumptions that an average female produced only 10,000 eggs in her lifetime and the daily larval mortality rate was 25% (Table 8). *NA* losses of threadfin shad at SR 50 were less than 100 individuals. *NA* increases with lower fecundity estimates and higher larval mortality rates. As with gizzard shad, the assumption that an average female threadfin shad produces only 10,000 eggs in her lifetime is likely conservative and therefore it is believed that the estimate at SR 46 provides a worst-case scenario of actual *NA* loss at all withdrawal locations.

Table 7. Estimates of the total number of larval threadfin shad that would have been entrained during 2008–2009 under various water withdrawal scenarios

| **Withdrawal Location** | **Withdrawal Scenario*** | **Estimate of Total Number of Larvae Entrained (thousands)** | **% of Total Larval Transport Entrained†** |
|---|---|---|---|
| Lake Poinsett | Variable | 8,537 | 11.54 |
| SR 50 | Variable | 1,488 | 7.23 |
| SR 46 | 25 mgd | 95,475 | 7.51 |
| | 50 mgd | 190,950 | 15.02 |
| Lake Monroe | 10 mgd | 41,498 | NC |
| | 20 mgd | 82,995 | NC |
| Yankee Lake | 25 mgd | 26,751 | 4.27 |
| | 50 mgd | 53,501 | 8.53 |
| SR 44 | 25 mgd | 22,762 | 3.05 |
| | 50 mgd | 45,526 | 6.09 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Table 8. Predicted adult equivalent (*NA*) loss of threadfin shad due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | | | | | |
|---|---|---|---|---|---|---|---|
| | | Fecundity =10,000 eggs/female | | | Fecundity =30,000 eggs/female | | |
| | | Larval Mortality Rate (per day) | | | Larval Mortality Rate ( per day) | | |
| | | 5% | 10% | 25% | 5% | 10% | 25% |
| Lake Poinsett | Variable | 221 | 260 | 449 | 66 | 78 | 135 |
| SR 50 | Variable | 39 | 45 | 78 | 12 | 14 | 24 |
| SR 46 | 25 mgd | 2,472 | 2,907 | 5,024 | 742 | 873 | 1,509 |
| | 50 mgd | 4.944 | 5,815 | 10,048 | 1,485 | 1,746 | 3,017 |
| Lake Monroe | 10 mgd | 1,075 | 1,264 | 2,184 | 323 | 379 | 656 |
| | 20 mgd | 2,149 | 2,527 | 4,367 | 645 | 759 | 1,312 |
| Yankee Lake | 25 mgd | 693 | 815 | 1,408 | 208 | 245 | 423 |
| | 50 mgd | 1,385 | 1,629 | 2,815 | 416 | 489 | 845 |
| SR 44 | 25 mgd | 589 | 693 | 1,198 | 177 | 208 | 360 |
| | 50 mgd | 1,179 | 1,386 | 2,396 | 354 | 416 | 719 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

Threadfin shad are extremely abundant in the St. Johns River and undoubtedly fill a significant role in the trophic structure of the ecosystem (McLane 1955). Adults of this species are often observed forming large, dense schools of hundreds to thousands of individuals that are visible as they move along the surface of the river (McLane 1955) . Because of their small adult size (<175 mm TL) threadfin shad are an important prey species that has been stocked extensively in reservoirs throughout the southern United States to supplement the sport fish forage base (Noble 1981). Threadfin shad are planktivorous and may affect recruitment of other fishes through interspecific competition for zooplankton (DeVries et al. 1991); although, potential interactions with other species may vary widely among systems (Allen et al. 2000).

Threadfin shad in the St. Johns River appear to rarely exceed 100 mm TL. McLane (1955) plotted the length-frequency of over 1,200 threadfin shad and the largest individual observed was 75 mm TL. Thus, the average size of a reproductive threadfin shad from the St. Johns River was assumed to be approximately 70 mm TL with a corresponding weight of approximately 5.0 g (Carlander 1969a).

Because they are a pelagic schooling fish, block net estimates of threadfin shad standing stock likely underestimate true abundance (Siler 1986). Although block net data provide underestimates of standing stock, they provide a sense of scale regarding the potential magnitude of *NA* loss effects (Table 9). Block net data collected from the St. Johns River indicate standing stock of threadfin shad ranged from a low of 0.06 kg $ha^{-1}$ in Lake Poinsett to 63.9 kg $ha^{-1}$ in Lake George (Table 9). Even with a low standing stock estimate of 0.06 kg $ha^{-1}$, *NA* losses attributable to entrainment over 2 years with maximum withdrawal would only account for a 2.2% reduction of the estimated standing stock in Lake Poinsett. For Lake Monroe and the other river withdrawal sites that could only be compared to adjacent or nearby lakes, potential *NA* losses over the 2 years were extremely small (<0.5%; Table 9).

Allen et al. (2000), using the data from Hoyer and Canfield (1994), reported a positive relationship between chlorophyll and the standing stock of threadfin shad in 22 Florida lakes. However, a significant inverse relationship between threadfin standing stock and zooplankton density is reported. Because zooplankton density estimates are not available for St. Johns River lakes, their predictive relationship could not be used to predict biomass of threadfin shad in this study. An average standing stock of 22 kg $ha^{-1}$ is reported for other lakes, however, and, if this value is assumed in St. Johns River lakes, then *NA* losses due to entrainment over the 2 years of this study would account for <0.1% of the estimated standing stock of threadfin shad in any individual lake (Table 9).

These data suggest that *NA* losses of threadfin shad larvae due to entrainment would have a minimal impact on overall adult populations in the river. For additional perspective, if withdrawals of 50 mgd at Lake Poinsett, SR 46, Yankee Lake, and SR 44 are assumed to occur simultaneously (200 mgd total), the total *NA* loss of threadfin shad due to entrainment during 2008 and 2009 under a worst-case scenario would have been 15,708 individuals. This equates to a reduction over the 2 years of only 0.4% of the average estimated standing stock of threadfin shad in Lake Monroe alone. Because potential adult losses of threadfin shad comprise such a small percentage of estimated standing stocks, water withdrawals under the rates considered in this study will have minimal impact of threadfin shad in the St. Johns River due to larval entrainment. In addition, this worst-case estimate of adult loss based on a total surface water withdrawal rate of 200 mgd is an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

Table 9. Mean reduction of estimated standing stocks of adult threadfin shad resulting from calculated larval entrainment in 2008 and 2009 combined

| Withdrawal Location | Year | Estimated Standing Stock* (kg $ha^{-1}$) | Standing Stock Removed by 2008–2009 Entrainment (kg $ha^{-1}$) | Mean Reduction of Standing Stock Based on 2008–2009 Entrainment | Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | 1972† | 0.06 | 0.001 | 2.2% | (Cox et al. 1976) |
| | 1976‡ | 0.13 | 0.001 | 1.0% | (Cox et al. 1976) |
| | Avg. | 0.10 | 0.001 | 1.4% | |
| Florida Lakes | Avg. | 22.0 | 0.001 | 0.01% | (Hoyer and Canfield 1994; Allen et al. 2000) |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Avg. | 0.10 | 0.0002 | 0.2% | |
| Florida Lakes | Avg. | 22.0 | 0.0002 | 0.001% | (Hoyer and Canfield 1994; Allen et al. 2000) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | 1990† | 1.5 | 0.006 | 0.42% | (McDaniel and Cox 1993) |
| | 1990‡ | 7.4 | 0.006 | 0.08% | (McDaniel and Cox 1993) |
| | 1991‡ | 6.3 | 0.006 | 0.10% | (McDaniel and Cox 1993) |
| | Avg. | 5.1 | 0.006 | 0.12% | |
| Florida Lakes | Avg. | 22.0 | 0.006 | 0.03% | (Hoyer and Canfield 1994; Allen et al. 2000) |
| **SR 46** | | | | | |
| Compared to Lake Monroe | Avg. | 5.1 | 0.014 | 0.28% | (McDaniel and Cox 1993) |
| Florida Lakes | Avg. | 22.0 | 0.014 | 0.07% | (Hoyer and Canfield 1994; Allen et al. 2000) |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | Avg. | 5.1 | 0.001 | 0.02% | (McDaniel and Cox 1993) |
| Florida Lakes | Avg. | 22.0 | 0.001 | 0.004% | (Hoyer and Canfield 1994; Allen et al. 2000) |
| **SR 44** | | | | | |
| Compared to Lake George (18,616 ha) | 1984 | 0.2 | 0.0003 | 0.16% | (Cheek et al. 1984) |
| | 1990‡ | 63.9 | 0.0003 | 0.0005% | (Cross et al. 1993) |
| | 1991‡ | 1.0 | 0.0003 | 0.03% | (Cross et al. 1993) |
| | Avg. | 21.7 | 0.0003 | 0.002 | |
| Florida Lakes | Avg. | 22.0 | 0.0003 | 0.002 | (Hoyer and Canfield 1994; Allen et al. 2000) |

*Standing stocks were estimated from block net data. Adult loss was calculated assuming the most conservative lifetime fecundity scenario of 10,000 eggs per female and a daily larval mortality rate of 25%.

†Littoral samples

‡Limnetic samples

**Clown Goby (*Microgobius gulosus*)**

Postflexion larvae (7 mm TL)

Adult (>30 mm TL)

Photos by Matt Scripter

Clown gobies are small (maximum TL <60 mm) bottom-dwelling euryhaline fish that are common in estuarine habitats throughout the Southeast and along the Gulf Coast of the United States (Birdsong 1981). In the St. Johns River, they have been collected from Doctor’s Lake near Jacksonville south to Lake Harney (McLane 1955; Tagatz 1968). Clown gobies are rare in the higher salinity waters downstream of Doctor’s Lake (MacDonald et al. 2009). Clown gobies occupy both vegetated and unvegetated habitats that have bottom sediments consisting of sand, mud, or a mixture of sand, mud, and shell (McLane 1955). Clown gobies excavate elaborate burrows for spawning and as protection from predation (McLane 1955; Birdsong 1981). McLane (1955) reported that all of the St. Johns River lakes from Lake George to Lake Harney had enormous populations of clown gobies although they were rare in later FWC block net samples. The rarity of clown goby in block net samples probably reflects more the inefficiency of the sampling technique to collect this benthic species rather than a decline in abundance from when McLane sampled the river with other gear types.

Temporal and Spatial Patterns of Distribution

In the 2008–2009 surveys, clown goby larvae were major components of the ichthyoplankton at all sampling sites from SR 44 to SR 46 but were rare in samples collected farther south (Figure 18). A total of 50,187 clown goby larvae and 547 juveniles/adults were enumerated in processed samples. Another 16,180 goby larvae could be identified only to family and could not be differentiated from larval naked goby (*Gobiosoma bosc*), which were also abundant in the samples. Based on the ratio of identified clown goby larvae to naked goby larvae in the samples, 50.2% of these unidentified larvae were estimated to be clown gobies and 49.8% were naked gobies. Therefore, to more accurately estimate total potential clown goby entrainment 50.2% of the unidentified goby larvae were added to the total clown goby density entrainment estimate.

Clown goby larvae appeared in samples from mid-March through mid-August with peak abundance occurring in May, June, and July (Figure 18). Larval abundance in 2008 was

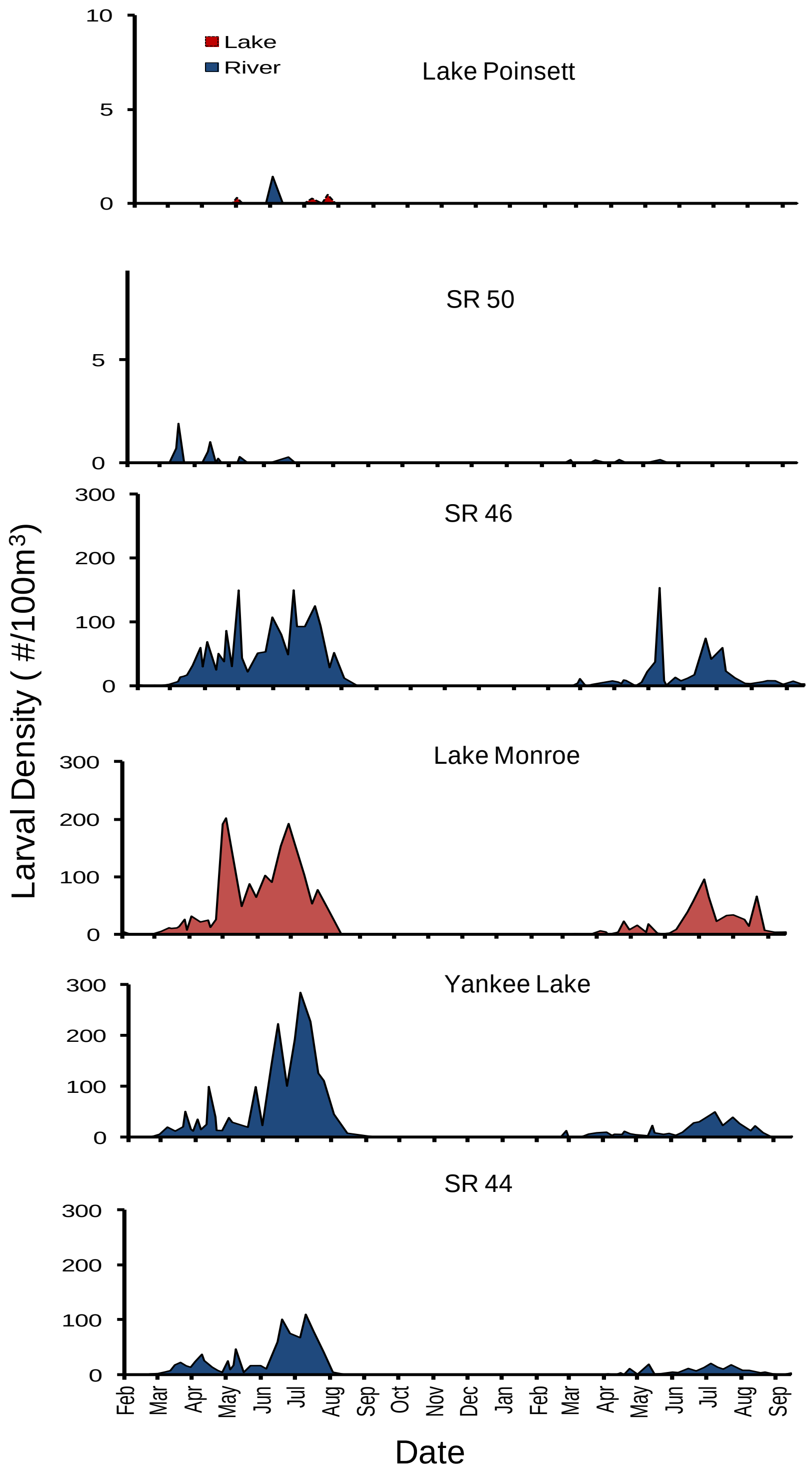



Figure 18. Mean density (number 100 m$^{-3}$) of clown goby larvae collected on each sampling date in 2008 and 2009

substantially higher than in 2009 at all sites (Figure 18). Major flooding of the entire basin following the passage of Tropical Storm Fay in August 2008 caused significant declines in river salinity throughout the fall (Figure 12). This large influx of freshwater, along with the prolonged drop in salinity, may have contributed to the decline in larval abundance observed in 2009, possibly reflecting reduced adult recruitment in 2008, increased mortality, downstream displacement, or a combination of these factors. Increased mortality due to low DO immediately following the storm event may also have been a factor (Breitburg 1992; Miller et al. 2014).

### Size Distributions

Clown gobies ranged in size from 2 mm TL yolk-sac larvae to 65 mm TL adults (Figure 19). Most of the individuals were 4–5 mm TL and were in the postflexion developmental stage. Larger individuals were typically more abundant in nighttime samples but the abundance of smaller larvae did not differ appreciably between day and night samples.

### Entrainment Losses and Adult Equivalents

Total estimated numbers of clown goby larvae potentially entrained during 2008–2009 under full withdrawal scenarios ranged from 29.2 million at Yankee Lake to 26.7 million at SR 46 (Table 10). However, at SR 46, full water withdrawals would have removed only 7.5% of the total clown goby larvae transported by this intake site (Table 10). At the other river intake sites, <5% of the estimated larval transport would have been entrained.

Provancha and Hall (1991) reported that fecundity of clown goby collected from the Banana River ranged between 300 and 500 eggs per female. McLane (1955) reported five females (21–33 mm TL) collected from the St. Johns River contained an average of 310 ova. Clown goby nests inspected by Gainser (2005) contained from 340 to 442 eggs or developing embryos. Clown goby rarely survive past age 2 and most mature adults have only a single year of reproductive success (Provancha and Hall 1991). Gobies maintained in culture systems at the Florida Institute of Technology were capable of reproducing at approximate monthly intervals (J. Shenker, pers. obs.). Therefore, to span the possible range of total annual egg production, two spawning scenarios were computed: 800 eggs (two nests in the one reproductive year) and 2,000 eggs (five nests in the reproductive year). These scenarios assume an average estimated fecundity of 400 eggs per female per spawning attempt.

Clown goby males construct burrows in soft sediment, often at the base of emergent vegetation (Gainser 2005). During spawning, females deposit their eggs in the burrow, and both parents guard the eggs throughout embryonic development (Gainser 2005). Because yolk-sac larvae remain in the burrows until the egg yolk is absorbed, they are assumed to be vulnerable to plankton nets immediately after leaving the nest. Because the most abundant larvae captured by the plankton nets were small postflexion larvae that

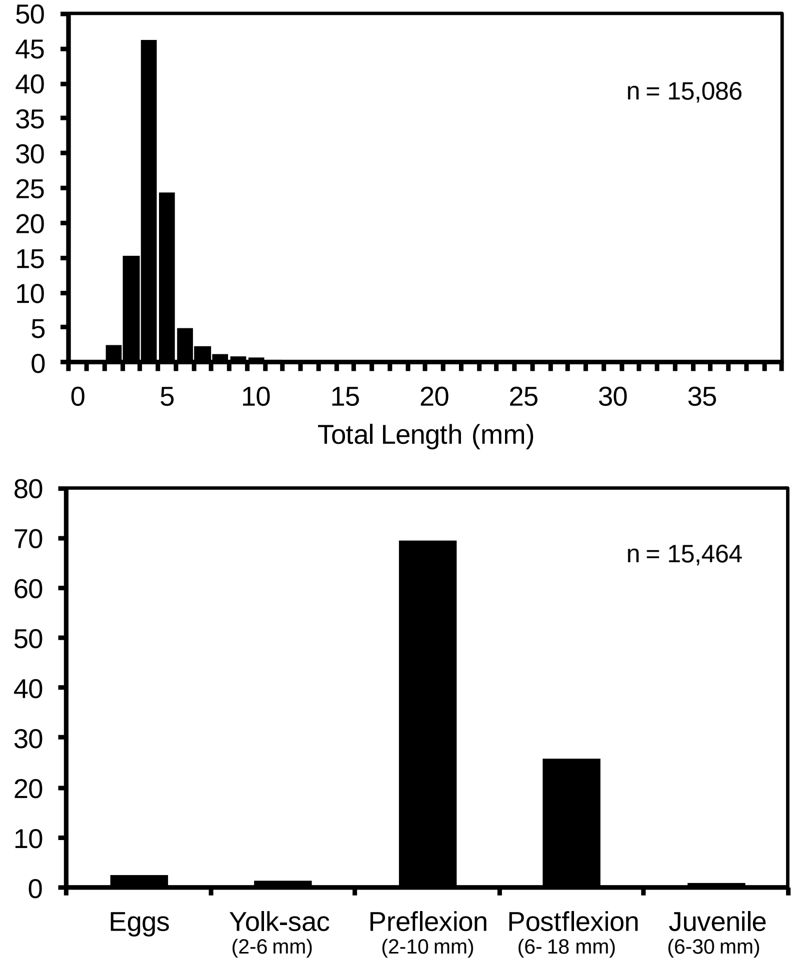



Figure 19. Length-frequency and life stage of clown goby collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

had recently absorbed their yolk sac, a precollection pelagic mortality rate was not included in the estimate of adult equivalent mortality.

The potential *NA* losses calculated for the two reproductive scenarios (excluding mortality from egg deposition until swim-up) are presented in Table 11. The maximum predicted *NA* loss of clown goby adults was 84,686 individuals at the Yankee Lake withdrawal site under a 50 mgd withdrawal scenario with the assumption that an average female produced 800 eggs in her lifetime.

Little is known about the ecological role of clown goby in the freshwater habitats of the St. Johns River ecosystem. Clown goby are opportunistic benthic feeders. In the St. Johns River they consume primarily chironomids, copepods, larval fishes, polychaetes, and ostracods (McLane 1955). Although Gainser (2005) found clown goby burrows associated primarily with emergent vegetation in shallow water habitats, McLane (1955) reported clown goby in the St. Johns River were equally abundant between both shallow and deep water habitats.

Because they are small benthic fishes that burrow and are difficult to quantitatively sample, estimates of clown goby densities in the Middle St. Johns River are lacking. McLane (1955) only provided a qualitative description of abundance while clown goby were rare in FWC block net samples. In the Banana River, Provancha and Hall (1991) reported annual mean clown goby densities ranged from 2.2 to 8.7 individuals $m^{-2}$. These are within the range of densities reported for other gobioids occupying similar habitats (Nero 1976; Crabtree and Dean 1982). MacDonald et al. (2009), however, reported an average density of clown gobies in the St. Johns River estuary collected monthly with a 21.3-m seine over a 9-year period was 0.009 fish $m^{-2}$ (90 fish $ha^{-1}$). Maximum density observed was 1.0 fish $m^{-2}$ (10,000 fish $ha^{-1}$). The average density value reported by MacDonald et al. (2009; 90 fish $ha^{-1}$) and the maximum density value (10,000 fish $ha^{-1}$) were used to give perspective to potential entrainment effects on clown goby populations in the river. However, our predicted effects may be slightly overestimated because 30% of the smaller gobies (primarily either clown goby or naked goby) collected by MacDonald et al. (2009) could not be identified to species in the field and thus were not included in his density estimates.

Table 10. Estimates of the total number of larval clown goby that would have been entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Total Number of Larvae Entrained (thousands) | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | 0 | NC |
| SR 50 | Variable | 0 | NC |
| SR 46 | 25 mgd | 13,328 | 3.73 |
| | 50 mgd | 26,656 | 7.46 |
| Lake Monroe | 10 mgd | 6,881 | NC |
| | 20 mgd | 13,762 | NC |
| Yankee Lake | 25 mgd | 14,601 | 2.48 |
| | 50 mgd | 29,202 | 4.96 |
| SR 44 | 25 mgd | 6,191 | 1.85 |
| | 50 mgd | 12,382 | 3.70 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Table 11. Predicted adult equivalent (*NA*) loss of clown goby due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*)† | |
|---|---|---|---|
| | | Fecundity =800 eggs/female | Fecundity =2,000 eggs/female |
| Lake Poinsett | Variable | NC | NC |
| SR 50 | Variable | NC | NC |
| SR 46 | 25 mgd<br>50 mgd | 38,651<br>77,302 | 15,460<br>30,921 |
| Lake Monroe | 10 mgd<br>20 mgd | 19.955<br>39,910 | 7,982<br>15,964 |
| Yankee Lake | 25 mgd<br>50 mgd | 42,343<br>84,686 | 16,937<br>33,874 |
| SR 44 | 25 mgd<br>50 mgd | 17.954<br>35,908 | 7,182<br>14,363 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Potential *NA* loss at each withdrawal site was compared to the estimated abundance of clown gobies in the area encompassed by Lake Monroe alone. Lake Monroe was used because the lake encompasses a known area as compared to river sites, which are open-ended. Having a known area allows extrapolation of *NA* to a fish $ha^{-1}$ basis for evaluating withdrawal effects simply by dividing *NA* by 3,500 (Table 12).

A predicted *NA* loss of clown goby under a worst-case, 50 mgd withdrawal scenario at Yankee Lake over 2 years (84,686 fish) represents 26.9% of the estimated abundance of clown goby in Lake Monroe alone in an individual year if average densities in the lake were 90 fish $ha^{-1}$ (Table 12). However, since the *NA* loss represents 2 years of entrainment, the average annual percent of the Lake Monroe population potentially lost to entrainment should be halved (13.5%). If clown goby densities are closer to 10,000 fish $ha^{-1}$, *NA* losses would be <0.2% of the adult population in Lake Monroe. Similar *NA* losses are calculated to occur at the SR 46 withdrawal site under full withdrawals (Table 12).

Table 12. Estimated percent reduction in mean density of adult clown goby within Lake Monroe alone due to larval entrainment at various withdrawal sites in 2008 and 2009

| Withdrawal Location | Estimated Density in Lake Monroe (No. ha$^{-1}$) | | Adults (*NA*)* Removed by 2008–2009 Larval Entrainment (No. ha$^{-1}$) | % Reduction in Density† of Adults in Lake Monroe Based on 2008–2009 Entrainment | |
|---|---|---|---|---|---|
| | **Mean** | **Max.** | | **Mean** | **Max.** |
| Lake Monroe (3,500 ha) | 90 | 10,000 | 11.4 | 12.7% | 0.1% |
| **SR 46** | | | | | |
| Compared to Lake Monroe | 90 | 10,000 | 22.1 | 24.6% | 0.2% |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | 90 | 10,000 | 24.2 | 26.9% | 0.2% |
| **SR 44** | | | | | |
| Compared to Lake Monroe | 90 | 10,000 | 10.3 | 11.4% | 0.1% |

*Adult loss represents losses for 2008 and 2009 combined. Adult loss was calculated assuming the most conservative lifetime fecundity scenario of 800 eggs per female in conjunction with maximum water withdrawal rates at each location.

†Clown goby mean and maximum density estimates are from MacDonald et al. (2009).

If a worst-case, 50 mgd withdrawal at SR 46, along with a 20 mgd withdrawal at Lake Monroe, a 50 mgd withdrawal at Yankee Lake, and a 50 mgd withdrawal at SR 44 were all occurring simultaneously (170 mgd total), the total estimated *NA* loss of clown goby due to entrainment during 2008–2009 combined would be 237,806 adult individuals. This equates over the 2 years to 75% (67.9 fish ha$^{-1}$) of the estimated density of clown goby in Lake Monroe alone in a single year if average adult densities in the lake were 90 fish ha$^{-1}$. However, considering effects on an annual basis, this loss would only be 38% (34 fish ha$^{-1}$). Although this might be considered a significant reduction if withdrawal effects were all centered in Lake Monroe, effects will be spread among all the withdrawal sites. In addition, this worst-case estimate of adult loss based on a total surface water withdrawal rate of 170 mgd is an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

Data suggests that *NA* losses due to larval entrainment under the worst-case full withdrawal scenarios could have localized impacts on clown goby populations if population levels are similar to those observed in the lower basin. Future studies should attempt to quantify actual adult clown goby abundance in the middle basin near each of the withdrawal locations to provide a better estimation of potential entrainment effects. In addition, fecundity studies and studies on spawning frequency would also provide useful information for these calculations.

Based on the data presented here, potential entrainment effects on clown gobies under the worst-case withdrawal scenarios are minor. A minor impact conclusion is supported by the relatively small percentage (<8%) of larvae passing by any of the intake sites that would have been entrained (Table 10). In addition, data suggest *NA* losses could account for up to 24.2 fish $ha^{-1}$ of the adult population (Yankee Lake) but only on a localized basis. Impacts may be localized near withdrawal sites because planktonic clown goby larvae become demersal when then reach a size of 6–11 mm TL after which they settle to the benthos and metamorphose into juveniles.

Abundance of clown goby in the St. Johns River sites varied widely between years and may reflect freshwater discharge effects on either salinity or DO. Clown goby in the St. Johns River are most abundant at salinities between 1 and 9 psu (McLane 1955; MacDonald et al. 2009). From February through June 2008, salinities in Lake Monroe ranged from 0.8 to 1.0 psu (1500–2300 µmhos conductivity). After the passage of Tropical Storm Fay, salinities dropped to <0.2 psu (<410 µmhos conductivity) in less than a month. Salinities remained low until January 2009 and did not recover to 2008 levels until March. Clown goby near Lake Monroe are likely approaching their physiological limit with regards to the species' minimum salinity tolerance. For marine invaders that are near their physiological extremes, sudden salinity drops may cause significant stress or mortality through effects on their ability to osmoregulate (Odum 1953). Johnson and Snelson (1996) implicated low conductivities (<700 µmhos $cm^{-1}$; salinity <0.4 psu) as the cause for reproductive failures and mortality of Atlantic stingrays (*Dasyatis sabina*) in Lake Monroe. Following passage of Tropical Storm Fay, surface DO levels in Lake Monroe also dropped to below 1.0 mg $L^{-1}$ for nearly a month. Although DO tolerances of clown goby are lacking in the literature, levels below 1.0 mg $L^{-1}$ likely caused direct mortality of naked goby (Breitburg et al. 1994; Miller et al. 2014). Survival of clown goby may also have been reduced because of reduced feeding under the anoxic conditions (Breitburg et al. 1994).

Wide interannual variations in the abundance of clown goby larvae observed in this study may serve to override or mask entrainment effects that might occur due to water withdrawals. Further information on the ecological role of clown goby in the St. Johns River and on species abundance and its relationship to freshwater discharge, salinity, and DO is needed if potential entrainment effects are to be fully understood.

**Naked Goby (*Gobiosoma bosc*)**

Postflexion larvae (6 mm TL)

Photos by Matt Scripter

Adult (29 mm TL)

Naked gobies are also small (maximum TL <50 mm), common bottom-dwelling euryhaline fish that are commonly found in estuarine habitats along the Atlantic and Gulf coasts from Cape Cod to northern Mexico (Lee et al. 1980). In the St. Johns River, naked gobies have been collected from the mouth of the river south to the Puzzle Lakes area between Lake Harney and Lake Poinsett (McLane 1955; Tagatz 1968; MacDonald et al. 2009). Similar to clown gobies, naked gobies occupy both vegetated and unvegetated habitats that have bottom sediments consisting of sand, mud, or a mixture of sand, mud, and shell (McLane 1955). McLane (1955) reported naked gobies in the St. Johns were frequently encountered among thick deposits of snail shells and in woody debris. An association with thick deposits of snail shells and woody debris may reflect use of these habitats for spawning and refugia. In other estuaries, naked gobies are often found in close association with oyster reefs (Dahlberg and Conyers 1973; Harding and Mann 2000; Lehnert and Allen 2002). Females reportedly lay adhesive eggs in clean oyster shells whereas the complex habitat also provides the adults refugia from predation (Dahlberg and Conyers 1973). In more freshwater habitats, other bivalve shells may be used as spawning substrate (Lehnert and Allen 2002). Naked gobies, unlike clown gobies, do not excavate elaborate burrows for spawning but they may burrow into the substrates during the coldest part of the winter (Dahlberg and Conyers 1973). McLane (1955) reported naked gobies were abundant in the St. Johns River from Doctor's Lake south to Lake Harney. Their greatest abundance occurred in soft bottom lakes with large numbers of snails and mussels. Unfortunately, no quantitative estimates of abundance were presented. Naked gobies were rarely reported from FWC block net samples.

Temporal and Spatial Patterns of Distribution

A total of 49,995 naked goby larvae and 437 juveniles/adults were enumerated in processed samples. Like the clown goby, naked goby larvae were major components of the ichthyoplankton at the four northernmost sampling regions but were rare in samples from SR 50 and Lake Poinsett (Figure 20). Of the 16,180 goby larvae that could not be identified to species, 8,058 (49.8%) were added to the naked goby entrainment estimates.

In 2008, naked goby larvae first appeared in our samples in late March and were present through mid-August but were absent throughout the rest of the year following the passage of Tropical Storm Fay (Figure 20). Peak abundance occurred in May and June. In 2009, naked goby larvae first appeared in late March or early to mid-April and were present at some sites into September (Figure 20). Timing of peak abundance also varied among sites in 2009. For example, peak abundance at SR 46 occurred in April while in Lake Monroe larvae were most abundant in July (Figure 20). In 2009, naked goby were collected at all sites through August. Densities of naked goby were substantially less in 2009 than in 2008 (Figure 20).

Size Distributions

Naked gobies ranged in size from 2 mm TL yolk-sac larvae up to 63 mm TL adults. Almost all of the naked goby larvae collected were in the preflexion and postflexion life stage with >75% of the larvae ranging between 3 and 5 mm TL (Figure 21). Larger adults were typically collected at night. However, during those months when both day and night samples were collected, ANOVA also indicated that larvae were significantly more abundant at night ($P < 0.05$). Differences were most pronounced during the peak spawning months of May and June in 2008. Day versus night differences were less pronounced in 2009 than in 2008.

Entrainment Losses and Adult Equivalents

Total estimated number of naked goby larvae potentially entrained during 2008–2009 under full withdrawal scenarios ranged from 20.1 million at Yankee Lake to 9,000 at SR 50 (Table 13). At SR 46, full withdrawals would have removed approximately 13.6% of the total naked goby larvae transported by this intake site (Table 13). At the other river intake sites, <7% of the estimated larval transport would have been entrained. Almost all of the potential larval entrainment would have occurred during 2008, as larval densities were low in 2009.

Naked goby eggs hatch in 1–2 weeks and larvae are capable of feeding within a few hours after hatching (Nero 1976; Harding and Mann 2000). Following a larval duration of 18–20 days, and often using tidal flows to achieve an upriver migration, small juveniles settle into complex benthic habitats (Shenker et al. 1983; Breitburg 1989; Breitburg 1991). Naked goby mature at age 2 and typically survive for a single spawning year (Nero 1976). Fecundity estimates range from 700 to 1,400 eggs per female (Dahlberg and Conyers 1973). Naked gobies are polygamous in that several females may deposit eggs in a nest guarded by a single male (Dahlberg and Conyers 1973).

For naked goby, two possible fecundity scenarios were used to calculate *NA* loss. The first scenario assumes average adult fecundity is 1,300 eggs and females nest once during the reproductive season. The second scenario assumes that females nest five times during

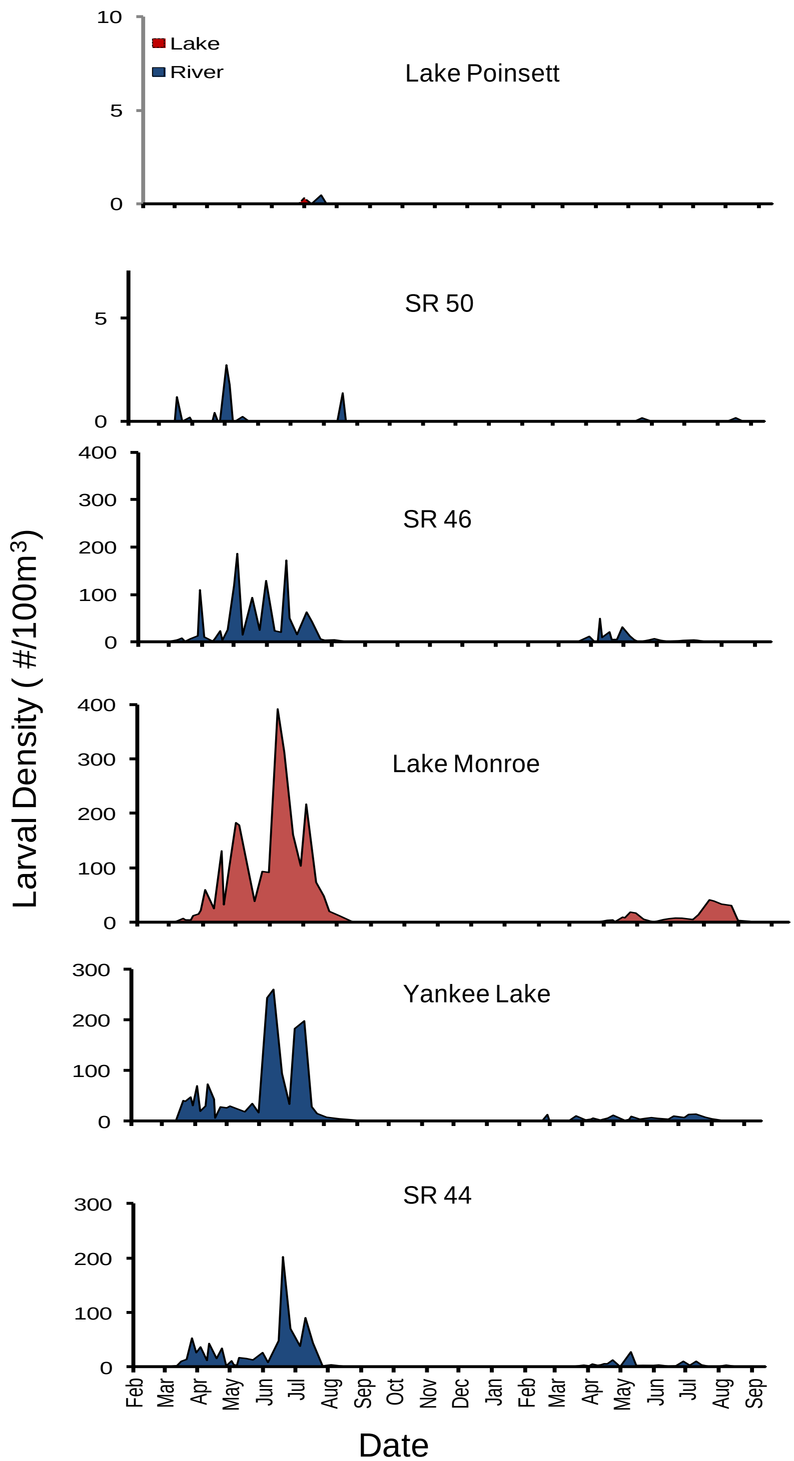



Figure 20. Mean density (number 100 $m^{-3}$) of naked goby larvae collected on each sampling date in 2008 and 2009

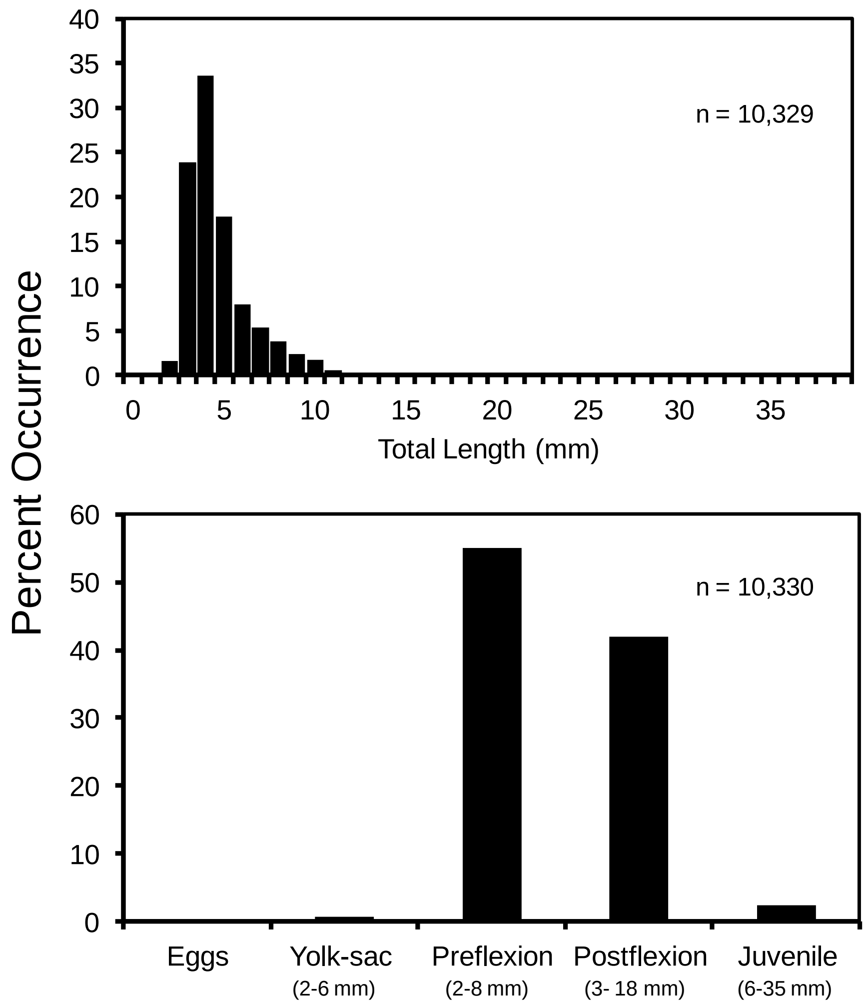



Figure 21. Length-frequency and life stage data on naked goby collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

a single reproductive season (6,500 eggs). Since newly hatched, yolk-sac naked gobies remain in the nest until they reach the preflexion stage, mortality during the first few days of life is likely low. Once they reach the preflexion stage they become pelagic and were vulnerable to sampling gear. Therefore, precollection mortality rates were not incorporated into *NA* calculations. The maximum predicted *NA* loss of naked gobies due to larval entrainment under a worst-case withdrawal scenario was 17,533 individuals at Yankee Lake (Table 14).

Table 13. Estimates of the total number of larval naked goby that would have been entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Total Number of Larvae Entrained (1,000's)† | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | NC | NC |
| SR 50 | Variable | 9 | 5.21 |
| SR 46 | 25 mgd | 6,060 | 6.79 |
| | 50 mgd | 12,120 | 13.58 |
| Lake Monroe | 10 mgd | 5,625 | NC |
| | 20 mgd | 11,250 | NC |
| Yankee Lake | 25 mgd | 10,070 | 3.48 |
| | 50 mgd | 20,141 | 6.96 |
| SR 44 | 25 mgd | 5,271 | 2.59 |
| | 50 mgd | 10,542 | 5.19 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Table 14. Predicted adult equivalent (*NA*) loss of naked goby due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | |
|---|---|---|---|
| | | Fecundity =2,600 eggs/female | Fecundity =6,500 eggs/female |
| Lake Poinsett | Variable | NC | NC |
| SR 50 | Variable | 8 | 3 |
| SR 46 | 25 mgd<br>50 mgd | 5,281<br>10,563 | 2,113<br>4,225 |
| Lake Monroe | 10 mgd<br>20 mgd | 4,902<br>9,804 | 1,961<br>3,992 |
| Yankee Lake | 25 mgd<br>50 mgd | 8,776<br>17,553 | 3,510<br>7,021 |
| SR 44 | 25 mgd<br>50 mgd | 4,594<br>9,817 | 1,837<br>3,675 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Like the clown goby, little is known of the ecological role of naked gobies in the St. Johns River ecosystem. Naked gobies are intermediate in the trophic structure. In estuaries, adults feed on infaunal and epibenthic invertebrates (Dahlberg and Conyers 1973) and are preyed upon by important piscivorous predators including striped bass (*Morone saxatilis*), bluefish (*Pomatomus saltatrix*), and weakfish (*Cynoscion regalis*; Harding and Mann 2000). Naked goby likely serve a similar intermediate ecological role in the St. Johns River.

Because they are small benthic fishes that are difficult to quantitatively sample, estimates of naked goby densities in the freshwater reaches of the St. Johns River are also lacking. Similar to clown gobies, McLane (1955) only provided qualitative descriptions of abundance and naked goby were rare in FWC block net samples. Crabtree and Dean (1982) reported naked goby densities ranged between 7.7 and 25.2 fish $m^{-2}$ in two South Carolina tidal pools. Mean densities of naked goby in the Chesapeake Bay were 8.0 fish $m^{-2}$ (Nero 1976). Hoese (1962) reported naked goby densities of 4.9 fish $m^{-2}$ along the eastern Chesapeake Bay shore in Virginia. Since these studies were conducted in habitats that contain oyster reefs, reported densities are likely higher than those found in the St. Johns River around Lake Monroe. MacDonald et al. (2009) reported the average density of naked gobies in the St. Johns River estuary collected monthly with a 21.3-m seine over a 9-year period was 0.004 fish $m^{-2}$ (40 fish $ha^{-1}$). Maximum densities encountered were 0.7 fish $m^{-2}$ (7,000 fish $ha^{-1}$). The values reported by MacDonald et al. (2009) were used to give perspective to potential entrainment effects on naked goby populations in the river (Table 15). As with clown gobies, potential *NA* loss of naked gobies at each withdrawal site were compared to estimated abundance within the area encompassed by Lake Monroe alone.

A predicted *NA* loss of naked goby under a worst-case, 50 mgd withdrawal scenario at Yankee Lake over 2 years (17,533 fish) represents a potential loss of 12.5% of the estimated total numbers of naked goby adults in Lake Monroe alone in an individual year if average densities in the lake are 40 fish $ha^{-1}$ (Table 15). However, since the *NA* loss represents 2 years of entrainment, the average annual adult loss should be halved (6.3%). If naked goby densities are closer to 7,000 fish $ha^{-1}$, *NA* losses would be <0.1% of the adult population. Similar *NA* losses of around 7% (3.5% annually) of the standing stock in Lake Monroe alone are predicted for the worst-case scenarios at the other sites (Table 15).

If a worst-case, 50 mgd withdrawal at SR 46, along with a 20 mgd withdrawal at Lake Monroe, a 50 mgd withdrawal at Yankee Lake, and a 50 mgd withdrawal at SR 44 were all occurring simultaneously (170 mgd total), the total estimated *NA* loss of naked goby due to entrainment during 2008–2009 combined would be 47,737 individual adults (13.6 fish $ha^{-1}$ in Lake Monroe). This equates to a reduction over the 2 years of 34.0% of the estimated density of naked goby in Lake Monroe alone in a single year if average adult densities in the lake were 40 fish $ha^{-1}$. However, considering losses on an annual basis, the loss would only be 17%. Based on the data presented here, potential entrainment effects on naked gobies under the worst-case withdrawal scenarios are minor because effects would be distributed among all withdrawal sites.

As discussed relative to clown gobies, abundance of naked goby in the St. Johns River study sites varied widely between years and likely reflect freshwater discharge effects on DO (Miller et al. 2014). It is possible that these wide interannual variations in abundance may serve to override or mask entrainment effects that might occur due to water withdrawals. Further information on the ecological role of naked goby in the St. Johns River and on species abundance and its relationship to freshwater discharge, salinity, and DO is needed if potential entrainment effects are to be fully understood.

Table 15. Percent reduction in estimated mean density of adult naked goby in Lake Monroe alone due to larval entrainment at various withdrawal sites in 2008 and 2009

| Withdrawal Location | Estimated Density in Lake Monroe (No. ha$^{-1}$) | | Adults (*NA*)* Removed by 2008–2009 Larval Entrainment (No. ha$^{-1}$) | % Reduction in Density† of Adults in Lake Monroe Based on 2008–2009 Entrainment | |
|---|---|---|---|---|---|
| | Mean | Max. | | Mean | Max. |
| Lake Monroe (3,500 ha) | 40 | 7,000 | 3.0 | 7.5% | <0.1% |
| **SR 46** | | | | | |
| Compared to Lake Monroe | 40 | 7,000 | 2.8 | 7.0% | <0.1% |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | 40 | 7,000 | 5.0 | 12.5% | <0.1% |
| **SR 44** | | | | | |
| Compared to Lake Monroe | 40 | 7,000 | 2.8 | 7.0% | <0.1% |

**NA* represents losses for 2008 and 2009 combined. *NA* was calculated for the worst-case scenario: a lifetime fecundity of 2,600 eggs per female and maximum water withdrawal rates at each location.

†Density estimates were for Lake Monroe were derived from MacDonald et al. (2009).

## Black Crappie (*Pomoxis nigromaculatus*)

Adult

Yolk-sac larvae (4 mm TL)

Photo by Matt Scripter

Black crappie is a sunfish species found in reservoirs, lakes, and rivers throughout the central and eastern United States (Lee et al. 1980). In Florida, black crappies are often referred to as speckled perch or specks, and, in terms of angler effort, they are one of the most popular freshwater sport fishes in the state. Daily catch limits for black crappies in Florida are 25 fish per person. Black crappie can attain a maximum weight of 1.7 kg (3.8

lbs), although fish greater than 0.9 kg (2 lbs) are relatively rare in most water bodies (Hoyer and Canfield 1994).

In the St. Johns River, black crappie are widely distributed throughout the drainage system from Doctor's Lake south (McLane 1955; Tagatz 1968). Black crappies occur primarily in larger lakes and in the main river channel where they spend most of the year in deeper water (McLane 1955). Black crappies move into shallower water during the winter and early spring where they build nests and spawn, often in association with aquatic vegetation (Carlander 1969a; Pope and Willis 1997; Phelps et al. 2009). Most sport fishing effort for this species occurs during the spawning season (Cox et al. 1981). The FWC routinely collects angler harvest data for black crappies, and they are vulnerable to a variety of sampling gears. In terms of biomass, black crappies typically comprise <6% of the total fish community biomass (Moody 1954; Cox et al. 1978).

Temporal and Spatial Patterns of Distribution

A total of 20,666 black crappie larvae and 100 juveniles were enumerated from processed samples. Larvae were present at all potential withdrawal locations but were most abundant at SR 44 and Lake Poinsett (Figure 22). Larvae were least abundant in the river channel at SR 50 and at SR 46. At all sites larval densities rarely exceeded 100 fish $m^{-2}$.

Black crappie larvae began appearing in samples in December at water temperatures around 15.6°C (60°F) and were present generally through April (Figure 22). Based on larval abundance, peak spawning occurred in February, March, and April.

Size Distributions

Black crappies ranged in size from 3 mm TL yolk-sac larvae to 93 mm TL juveniles (Figure 23). Nearly 80% of the individuals sampled were in the 4–5 mm TL size range and were in the preflexion development stage (Figure 23). Black crappie juveniles were more abundant in nighttime than in daytime samples. Abundance of larvae, however, varied little between daytime and nighttime sampling events.

Entrainment Losses and Adult Equivalents

The estimated total number of larvae potentially entrained during 2008–2009 under the worst-case, full withdrawal scenarios ranged from 97,000 at SR 50 to 17.8 million at SR 44 (Table 16). Under the worst-case scenarios, 5%–15% of the total black crappie larvae transported past each potential withdrawal site would have been subject to entrainment (Table 16).

Black crappies are sexually mature by age 2 and may live up to age 9; however, relatively few fish live beyond age 6 (Carlander 1969a; Miller et al. 1990). Fecundity increases with size, ranging from approximately 20,000 eggs per female for smaller, recently matured individuals (230 g) up to 158,000 eggs per female for larger adults (>400 g; Carlander 1969a; Warren Jr. 2009). For calculation of *NA*, an average black crappie female with an average life span of 4 years that was reproductively active for at least 3 years was assumed. Given the range of fecundities reported in the literature, a reasonable estimate of lifetime fecundity would be between 100,000 and 300,000 eggs per female (Warren Jr. 2009).

Spawning black crappie place eggs in nests excavated in the substrate. Males guard the nest through hatching and until the young completely use their yolk reserves and begin exogenous feeding (swim-up; Warren 2009) . At hatching, larvae are approximately 2.3 mm TL and reach swim-up at approximately 4.0 mm TL (Chatry and Conner 1980). After swim-up, larvae disperse into the water column and grow at an approximate daily rate of 0.7 mm (Travnichek et al. 1996). Since 4–6 mm TL post-swim-up larvae were the most abundant size group collected in our samples (Figure 23), a precapture mortality rate was not used in calculation of potential *NA* loss.

Potential *NA* losses calculated for the two reproductive scenarios are presented in Table 17. The maximum predicted *NA* loss of black crappies was 356 individuals at the SR 44 withdrawal site under a worst-case, 50 mgd withdrawal scenario with the assumption that the average female produced 100,000 eggs in her lifetime. At all other sites under the worst-case withdrawals, *NA* was predicted to be <80 fish (Table 17).

Black crappie standing stock estimates calculated from block net data collected from Lakes Poinsett, Harney, Jesup, Monroe, and George ranged from 0.4 to 22.1 kg $ha^{-1}$ (Table 18). To estimate predicted *NA* losses on standing stocks, the average size of an adult harvestable-size black crappie was assumed to be 280 mm TL (11 in.) with a corresponding weight of 220 g (0.49 lbs; Hoyer and Canfield 1994). Multiplying *NA* loss using the lowest fecundity (fecundity = 100,000 eggs/female) by the average weight (220 g) and dividing by lake area indicates *NA* losses within the individual lakes under the most conservative reproductive scenario would range from <0.001 to 0.008 kg $ha^{-1}$ (Table 18). Compared to estimated standing stocks on a number of dates, *NA* losses for 2008 and 2009 combined account for <2% decline in the standing stock of black crappies in any one lake (<1% annually). For most standing stock estimates, *NA* losses for the 2 years combined account for <0.5% of standing stock (Table 18). *NA* losses over 2008 and 2009 would also account for ≤0.2% of average standing stock of black crappies estimated for a number of other Florida lakes (Hoyer and Canfield 1994).

To add additional perspective, if 50 mgd withdrawals at Lake Poinsett, SR 46, Yankee Lake, and SR 44 were to occur simultaneously (200 mgd total), the total *NA* loss of black crappies due to entrainment during 2008 and 2009 would be 537 individuals. Based on an average adult size of 220 g, this would equate to a reduction over the 2 years of only 2.6% (1.3% annually) of the estimated standing stock of black crappie in Lake Monroe

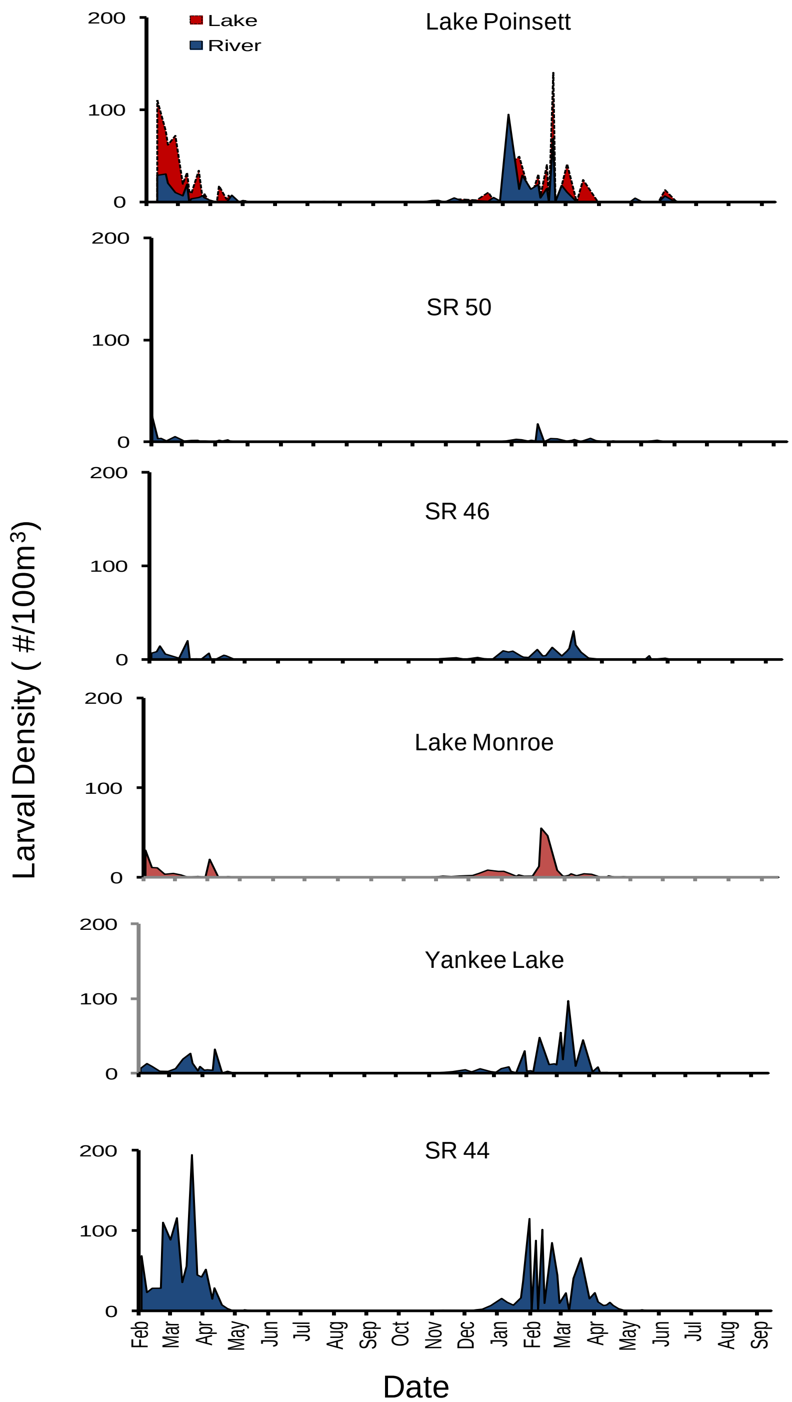
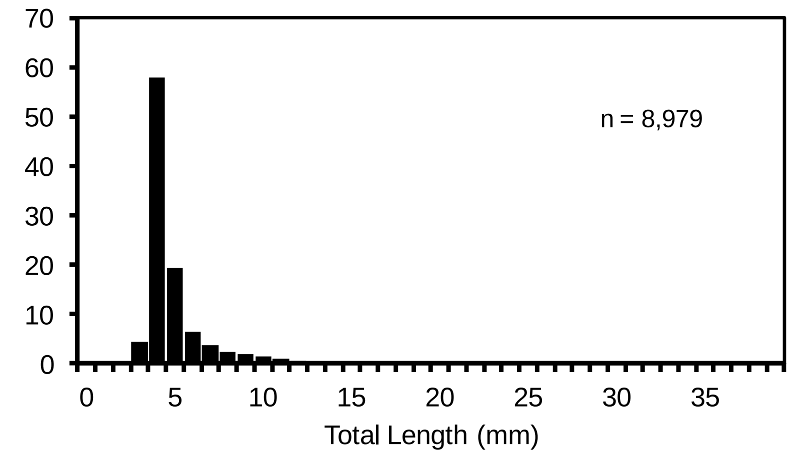



Figure 22. Mean density (number 100 $m^{-3}$) of black crappie larvae collected on each sampling date in 2008 and 2009



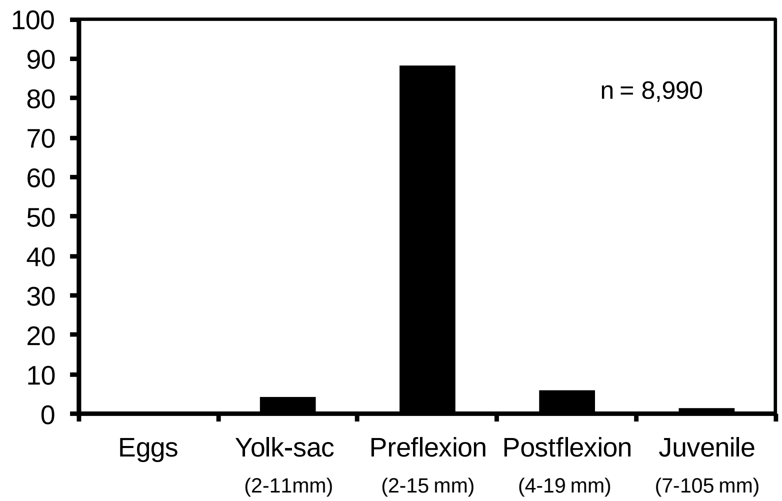




Figure 23. Length-frequency and life stage data of black crappies collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

Table 16. Estimated total number of black crappie larvae potentially entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Total Number of Larvae Entrained (1,000's) | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | 3,001 | 12.86 |
| SR 50 | Variable | 97 | 4.59 |
| SR 46 | 25 mgd | 1,090 | 7.02 |
| | 50 mgd | 2,180 | 14.05 |
| Lake Monroe | 10 mgd | 535 | NC |
| | 20 mgd | 1,070 | NC |
| Yankee Lake | 25 mgd | 1,923 | 4.29 |
| | 50 mgd | 3,846 | 8.59 |
| SR 44 | 25 mgd | 8,880 | 3.31 |
| | 50 mgd | 17,761 | 6.62 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

Table 17. Predicted adult equivalent (*NA*) loss of black crappies due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | |
|---|---|---|---|
| | | Fecundity =100,000 eggs/female | Fecundity =300,000 eggs/female |
| Lake Poinsett | Variable | 60 | 20 |
| SR 50 | Variable | 2 | 1 |
| SR 46 | 25 mgd | 22 | 7 |
| | 50 mgd | 44 | 15 |
| Lake Monroe | 10 mgd | 11 | 4 |
| | 20 mgd | 21 | 7 |
| Yankee Lake | 25 mgd | 38 | 13 |
| | 50 mgd | 77 | 26 |
| SR 44 | 25 mgd | 178 | 59 |
| | 50 mgd | 356 | 18 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

alone using the lowest estimated standing stock of 1.3 kg $ha^{-1}$ for the lake. In addition, this worst-case estimate of adult loss based on a withdrawal rate of 200 mgd total is an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

Finally, FWC creel survey data were summarized to compare potential *NA* loss to the number of fish removed by recreational harvest (Table 19). At SR 46, predicted *NA* loss over 2 years was only 1.1% of the black crappie harvested from the river by anglers from December 1997 through April 1998 (Table 19). *NA* loss at SR 46, Lake Monroe, and Yankee Lake were also just a small percentage of black crappie harvested annually from Lakes Jesup and Monroe alone.

The greatest effects of entrainment on black crappies may occur at SR 44 where an estimated 356 adults (178 annually) would have been be lost over 2 years (Table 17). Although the *NA* loss makes up only a small percentage of the overall estimated standing stocks of Lakes Monroe and George (Table 18), this loss could result in an increase in total estimated harvest in either lake alone of 2.7% to 3.9% (1.4%–2.0% annually; Table 19). However, if recreational harvest for the lakes were combined, 2 years of *NA* loss at SR 44 would only account for 1.5% of the recreational harvest. This loss is considered to be minor and not likely detectable in future recreational harvest statistics.

In conclusion, data suggest that potential entrainment effects on black crappies in the St. Johns River under the worst-case scenarios would only have minimal population effects. Standing stock estimates for the lakes alone would not be reduced measurably and overall harvest of adults would only slightly increase. Despite minimal impact, intake designs should be considered that minimize entrainment of black crappie larvae to the greatest extent possible, especially in the area around SR 44.

Table 18. Mean reduction of adult black crappie estimated standing stocks resulting from calculated larval entrainment in 2008 and 2009 combined

| Withdrawal Location | Date | Estimated Standing Stock* (kg ha$^{-1}$) | Standing Stock Removed by 2008–2009 Entrainment (kg ha$^{-1}$) | Mean Reduction of Standing Stock Based on 2008–2009 Entrainment | Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | Jun 1972 | 0.4 | 0.008 | 2.0% | (Cox et al. 1976) |
| | Mar 1973 | 2.9 | 0.008 | 0.3% | (Cox et al. 1976) |
| | May 1975 | 5.5 | 0.008 | 0.1% | (Cox et al. 1976) |
| | May 1976 | 2.5 | 0.008 | 0.3% | (Cox et al. 1976) |
| | May 1979 | 7.6 | 0.008 | 0.1% | (Cox et al. 1979) |
| | Apr 1991 | 9.9 | 0.008 | <0.1% | (Eisenhauer et al. 1993) |
| | May 1992 | 2.0 | 0.008 | 0.4% | (Eisenhauer et al. 1993) |
| | Apr 1993 | 0.4 | 0.008 | 2.0% | (Eisenhauer et al. 1993) |
| | Spr 1995 | 5.2 | 0.008 | 0.2% | (Cox et al. 1996) |
| | Spr 1996 | 0.6 | 0.008 | 1.3% | (Cox et al. 1996) |
| | Avg. | 3.7 | 0.008 | 0.2% | |
| Florida Lakes | Avg. | 5.1 | 0.008 | 0.2% | (Hoyer and Canfield 1994) |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Avg. | 3.7 | <0.001 | <0.1% | |
| Compared to Lake Harney (2,300 ha) | Fall 1991 | 22.2 | <0.001 | <0.1% | (McDaniel and Cox 1993) |
| | Fall 1991 | 6.5 | <0.001 | <0.1% | (McDaniel and Cox 1993) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | Fall 1990 | 1.3 | 0.001 | 0.1% | (McDaniel and Cox 1993) |
| | Fall 1991† | 2.8 | 0.001 | 0.4% | (McDaniel and Cox 1993) |
| | Fall 1991‡ | 19.0 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| **SR 46** | | | | | |
| Compared to Lake Jesup (4,300 ha) | Fall 1990 | 5.3 | 0.002 | <0.1% | (McDaniel and Cox 1993) |
| Compared to Lake Monroe | Fall 1990 | 1.3 | 0.003 | 0.2% | (McDaniel and Cox 1993) |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | Fall 1990 | 1.3 | 0.005 | 0.4% | (McDaniel and Cox 1993) |
| **SR 44** | | | | | |
| Compared to Lake Monroe | Fall 1990 | 1.3 | 0.022 | 1.7% | (McDaniel and Cox 1993) |
| Compared to Lake George (18,615 ha) | Mar 1984† | 1.8 | 0.004 | 0.2% | (Cheek et al. 1984) |
| | Fall 1990† | 16.2 | 0.004 | <0.1% | (Cross et al. 1993) |
| | Fall 1991† | 8.0 | 0.004 | <0.1% | (Cross et al. 1993) |

*Standing stocks were estimated from block net data. Adult loss was calculated for worst-case scenarios maximum withdrawal assuming a lifetime fecundity of 100,000 eggs per female.

†Littoral samples

‡Limnetic samples

Table 19. Comparison of potential *NA* losses from total annual entrainment in 2008 and 2009 to recreational black crappie harvest statistics

| Withdrawal Location | Date | Number Harvested by Anglers | Adult Equivalents (*NA*) Lost to Entrainment | *NA* Losses as a Percent of Sport Harvest | Harvest Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | Mar–May79 | 1,840 | 60 | 3.3% | (Cox et al. 1981) |
| | Nov 79–Mar 80 | 2,199 | 60 | 2.7% | (Cox et al. 1981) |
| | Jan–Apr 2000* | 30,061 | 60 | 0.2% | (Eisenhauer et al. 2002) |
| | Jan–Apr 2002* | 14,773 | 60 | 0.4% | (Eisenhauer et al. 2002) |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Mar 79–May 79 | 1,840 | 2 | 0.1% | (Cox et al. 1981) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | Avg. Oct–May 1992–1997 | 55,263 | 21 | <0.1% | (Holder et al. 2005) |
| | 1998–2004† | 4,295 | 21 | 0.5% | (Holder et al. 2005) |
| | 2005–2007‡ | 8,908 | 21 | 0.2% | (Holder et al. 2007) |
| | Jan–May 2010‡ | 10,550 | 21 | 0.2% | (Holder et al. 2010) |
| **SR 46** | | | | | |
| River (near Mullet Lake) | Dec 97–Apr 98 | 3,932 | 44 | 1.1% | (Jenkins et al. 1999) |
| Compared to Lake Jesup | Jan–May 2006 | 2,319 | 44 | 1.9% | (Holder et al. 2007) |
| | Jan–May 2007 | 5,258 | 44 | 0.8% | (Holder et al. 2007) |
| Compared to Lake Monroe | 1998–2004 | 4,295 | 44 | 1.0% | (Holder et al. 2005) |
| | 2005–2007 | 8,908 | 44 | 0.5% | (Holder et al. 2007) |
| | 2010 | 10,550 | 44 | 0.4% | (Holder et al. 2010) |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | 1998–2004 | 4,295 | 77 | 1.8% | (Holder et al. 2005) |
| | 2005–2007 | 8,908 | 77 | 0.9% | (Holder et al. 2007) |
| **SR 44** | | | | | |
| Lake Beresford Area | Jan–Dec 2000 | 9,141 | 356 | 3.9% | (Jenkins et al. 2002) |
| Compared to Lake George | 1985–1990 Avg. | 13,147 | 356 | 2.7% | (Snyder et al. 1990) |

* Creel data from Lake Poinsett and Lake Harney combined.

† 254 mm (10 in.) length limit instituted in fall 1998.

‡ Includes bank harvest from seawall fisherman.

**Bluegill Sunfish (*Lepomis macrochirus*)**

Adult

Postflexion larvae (7.2 mm TL)

Photo by Matt Scripter

Bluegill is a common sunfish species found in warm-water reservoirs, lakes, and rivers throughout the United States although the species was originally native only from the Mississippi River drainage east (Lee et al. 1980; Warren 2009). In Florida, bluegills are one of the most common fishes in the state. Daily catch limits of bluegills are 50 fish per person. Bluegills can attain a maximum weight of 0.9 kg (2.0 lbs), although fish greater than 0.5 kg (1 lb) are rare (Hoyer and Canfield 1994).

In the St. Johns River, bluegills are described as being the "most ubiquitous and abundant species of fish in the entire drainage system" (McLane 1955). Bluegills are common in virtually all habitat types. Bluegills are probably most abundant in vegetated littoral zones but they may also be abundant in deeper offshore waters. For example, bluegills are usually one of the most abundant fish collected by offshore haul seines (Moody 1954; Cox et al. 1978).

Bluegills spawn in shallow water in the summer, generally away from cover in dense colonies of up to several hundred nests; although, solitary spawning also occurs (Spotte 2007; Warren 2009). Sport fishing for bluegills occurs year-round but harvest statistics for this species are generally only collected during the winter months (Cox et al. 1981). Bluegill are grouped with redear sunfish (*Lepomis microlophus*) and other sunfishes in creel surveys into a general sunfish category called bream.

Temporal and Spatial Patterns of Distribution

A total of 15,648 bluegill larvae and 227 juveniles were enumerated in the processed samples. Bluegill larvae were common at all sampling locations but were most abundant at Lake Poinsett, Lake Monroe, and SR 46 (Figure 24). In 2008 bluegill larvae first appeared in late April and were present until early August (Figure 24). Spawning at all stations ceased after the passage of Tropical Storm Fay. In 2009, bluegill larvae again began appearing in April, but were present through the end of August and even into early

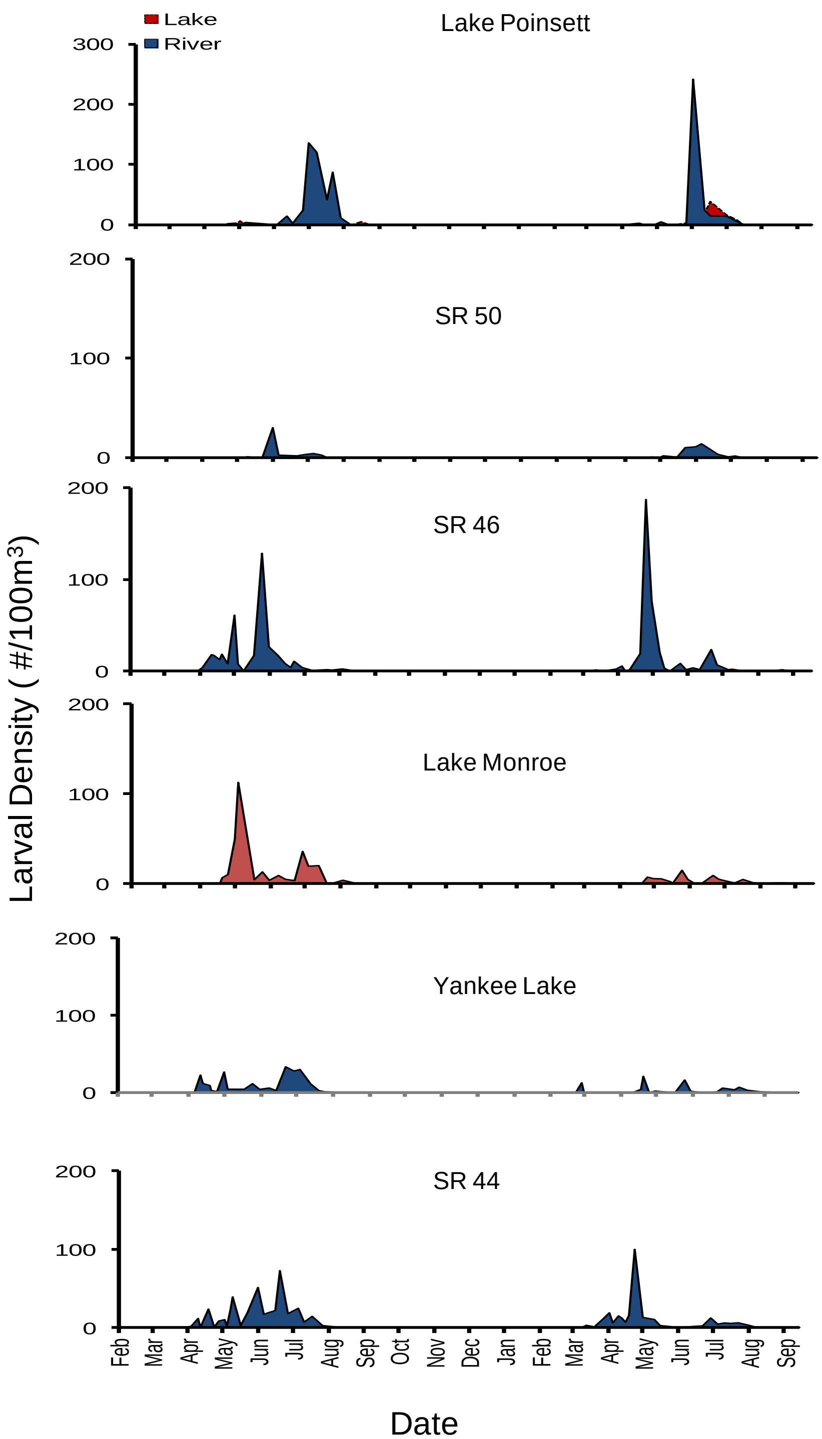



Figure 24. Mean density (number 100 $m^{-3}$) of bluegill larvae collected on each sampling date in 2008 and 2009

September. Bluegill larvae were less abundant in Lake Monroe in 2009 than in 2008 but between-year differences in abundance were not evident at the other stations (Figure 24).

Size Distributions

Bluegills ranged in size from 2 mm TL yolk-sac larvae to 38 mm TL juveniles (Figure 25). Approximately 80% of the larvae captured were in the 4–6 mm TL size range and were in the preflexion development stage (Figure 25). Bluegill juveniles were more abundant in nighttime than in daytime samples. Larval abundance, however, varied little between daytime and nighttime sampling events.

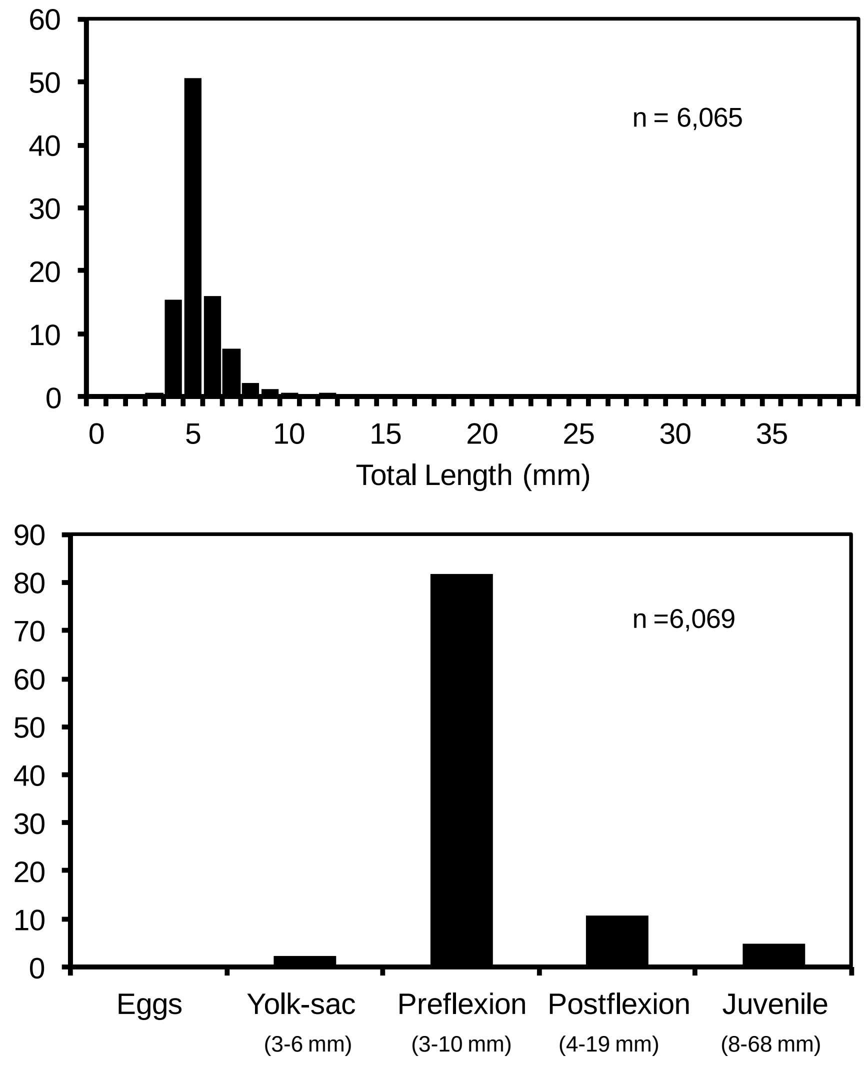



Figure 25. Length-frequency and life stage data of bluegills collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

Entrainment Losses and Adult Equivalents

The estimated total number of bluegill larvae potentially entrained during 2008–2009 under the worst-case full withdrawal scenarios ranged from 1.2 million at SR 50 to 9.8 million at Lake Poinsett (Table 20). Under the worst-case scenarios, 3%–18% of the total bluegill larvae transported by each potential withdrawal site would have been subject to entrainment (Table 20). Bluegills can become sexually mature by age 1 and may live up to age 9; however, relatively few fish live beyond age 5 (Carlander 1969a). Fecundity increases with size and average from approximately 17,000 eggs per female for smaller, recently matured individuals (165 mm TL; 6.5 in.) up to 50,000 eggs per female for larger adults (>216 mm TL; 8.5 in.; Carlander 1969a; Warren Jr. 2009). Females presumably are capable of spawning multiple times in a season (Warren 2009). For calculation of *NA*, an average bluegill female was assumed to have an average life span of 4 years, be reproductively active for at least 3 years, and spawn at least twice in a season. Assuming average fecundities of 17,000 to 50,000 eggs per female, this results in lifetime fecundity estimates of 100,000 and 300,000 eggs for use in calculating *NA* estimates

Male bluegills excavate the nest and guard the eggs and newly hatched young through swim-up (Spotte 2007; Warren 2009). At hatching, bluegill larvae are approximately 2–4 mm TL and swim-up in 3–4 days when they are 4–5 mm TL (Spotte 2007; Warren 2009).

Table 20. Estimated total number of bluegill larvae potentially entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Total Number of Larvae Entrained (1,000's) | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | 9,784 | 18.34 |
| SR 50 | Variable | 1,247 | 14.12 |
| SR 46 | 25 mgd | 3,732 | 6.75 |
| | 50 mgd | 7,463 | 13.51 |
| Lake Monroe | 10 mgd | 722 | NC |
| | 20 mgd | 1,444 | NC |
| Yankee Lake | 25 mgd | 1,621 | 3.17 |
| | 50 mgd | 3,242 | 6.34 |
| SR 44 | 25 mgd | 3,537 | 3.09 |
| | 50 mgd | 7,075 | 6.19 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

After swim-up, larvae disperse into the water column (Werner 1967). At sizes between 11 and 25 mm TL, bluegill fry apparently return to shoreline habitats (Storck et al. 1978; Conrow et al. 1990).

Since newly hatched preflexion larvae were most abundant (Figure 25), a precapture mortality rate was not included in the calculation of potential *NA* loss.

The 100,000 egg/female fecundity scenario yielded the highest potential *NA* loss (196 adult bluegill) at Lake Poinsett under the worst-case withdrawal scenario over the 2-year sampling period (Table 21). Lowest predicted *NA* loss under the worst-case scenario was at SR 50 (25 individuals). The maximum potential *NA* loss at Lake Poinsett only equates to a 4-day maximum, 50 fish per day per angler harvest.

Bluegill standing stock estimates calculated from block net data collected from Lakes Poinsett, Harney, Jesup, Monroe, and George ranged from 0.5 to 144.7 kg ha$^{-1}$ (Table 22). To compare predicted *NA* losses to standing stocks the average size of an adult harvestable-size bluegill was assumed to be 180 mm TL (7 in.) with a corresponding weight of 65 g (0.15 lbs; Hoyer and Canfield 1994). Multiplying *NA* loss using the lowest fecundity (fecundity = 100,000 eggs /female) by the average weight (65 g) and dividing by lake area indicates *NA* losses within the individual lakes under the most conservative reproductive scenario would range from <0.001 to 0.008 kg ha$^{-1}$ (Table 22). Compared to estimated standing stocks on a number of dates, *NA* losses for 2008 and 2009 combined accounts for ≤1.6% of the standing stock of bluegills in any one lake (<0.9% annually) . For the vast majority of most standing stock estimates, *NA* losses for the 2 years combined accounts for <0.1% of the estimated standing stock (Table 22).

To add additional perspective, if maximum withdrawals at Lake Poinsett and 50 mgd withdrawals at SR 46, Yankee Lake, and SR 44 were to occur simultaneously (200 mgd total), the total *NA* loss of bluegills due to entrainment during 2008 and 2009 would be 552 individuals. Based on an average adult size of 65 g, this would equate to a reduction over the 2 years of 0.1% (0.05% annually) of the estimated standing stock of bluegills in Lake Poinsett alone using the average estimated standing stock of 14.7 kg ha$^{-1}$ for the lake. However, this worst-case estimate of adult loss based on a withdrawal rate of 200 mgd total is an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

Finally, FWC creel survey data were summarized to compare potential *NA* loss to the number of bream removed by recreational harvest (Table 23). Predicted *NA* losses for Lake Poinsett (196 fish over 2 years) represents only 1.0%–7.9% (0.5%–4.0% annually) of the recreational harvest of bream from the lake (Table 23). *NA* loss in the river near station SR 46 only represents 1.9%–6.5% (1%–3.3% annually) of the harvest depending on the site considered. At all other sites, potential *NA* loss only constitutes ≤ 1.1% of the recreational harvest in nearby lakes (Table 23). Harvest estimates from all the areas are

Table 21. Predicted adult equivalent (*NA*) loss of bluegills due to larval entrainment during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | |
|---|---|---|---|
| | | Fecundity =100,000 eggs/female | Fecundity =300,000 eggs/female |
| Lake Poinsett | Variable | 196 | 65 |
| SR 50 | Variable | 25 | 8 |
| SR 46 | 25 mgd<br>50 mgd | 75<br>149 | 25<br>50 |
| Lake Monroe | 10 mgd<br>20 mgd | 14<br>29 | 5<br>10 |
| Yankee Lake | 25 mgd<br>50 mgd | 32<br>65 | 11<br>22 |
| SR 44 | 25 mgd<br>50 mgd | 71<br>142 | 24<br>48 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

conservative, however, as bream fisherman tend to fish year-round whereas creel statistics usually only represent winter months. Conversely, the bream category also contains other sunfish (e.g., redear sunfish), which may cause actual harvest estimates of bluegill to be higher.

In conclusion, comparisons based on block net data suggest that potential entrainment effects on bluegill populations in the St. Johns River under the worst-case scenarios would have negligible effects. *NA* loss of adult bream only constitutes a very small percentage of the adult recreational harvest at any one site. Given the wide-spread distribution of bluegill throughout the drainage basin and their overall abundance, entrainment will likely have an undetectable effect on bluegill populations in the St. Johns River.

Table 22. Mean reduction of adult bluegill estimated standing stocks resulting from calculated larval entrainment in 2008 and 2009 combined.

| Withdrawal Location | Date | Estimated Standing Stock* (kg $ha^{-1}$) | Standing Stock Removed by 2008–2009 Entrainment (kg $ha^{-1}$) | Mean Reduction of Standing Stock Based on 2008–2009 Entrainment | Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | Jun 1972 | 11.5 | 0.008 | <0.1% | (Cox et al. 1976) |
| | Dec 1972 | 0.5 | 0.008 | 1.6% | (Cox et al. 1976) |
| | Mar 1973 | 25.4 | 0.008 | <0.1% | (Cox et al. 1976) |
| | May 1975 | 1.7 | 0.008 | 0.5% | (Cox et al. 1976) |
| | May 1976 | 7.8 | 0.008 | 0.1% | (Cox et al. 1976 |
| | May 1977 | 0.5 | 0.008 | 1.6% | (Cox et al. 1977) |
| | May 1979 | 8.5 | 0.008 | <0.1% | (Cox et al. 1979) |
| | May 1980 | 27.5 | 0.008 | <0.1% | (Cox et al 1980) |
| | Apr 1991 | 17.7 | 0.008 | <0.1% | (Eisenhauer et al. 1993) |
| | May 1992 | 19.1 | 0.008 | <0.1% | (Eisenhauer et al. 1993) |
| | Apr 1993 | 16.2 | 0.008 | <0.1% | (Eisenhauer et al. 1993) |
| | Spr 1994 | 32.2 | 0.008 | <0.1% | (Cox et al. 1996) |
| | Spr 1995 | 13.9 | 0.008 | <0.1% | (Cox et al. 1996) |
| | Spr 1996 | 22.7 | 0.008 | <0.1% | (Cox et al. 1996) |
| | Avg. | 14.7 | 0.008 | <0.1% | |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Avg. | 12.9 | <0.001 | <0.1% | |
| Compared to Lake Harney (2,300 ha) | Fall 1991 | 37.8 | <0.001 | <0.1% | (McDaniel and Cox 1993) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | Fall 1990[†] | 9.8 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| | Fall 1990[‡] | 14.6 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| | Fall 1991[†] | 18.3 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| | Fall 1991[‡] | 11.6 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| **SR 46** | | | | | |
| Lake Jesup (4,300 ha) | Fall 1990 | 8.0 | 0.002 | <0.1% | (McDaniel and Cox 1993) |
| Compared to Lake Monroe | Fall 1990 | 9.8 | 0.003 | <0.1% | (McDaniel and Cox 1993) |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | Fall 1990 | 9.8 | 0.001 | <0.1% | (McDaniel and Cox 1993) |
| **SR 44** | | | | | |
| Compared to Lake Monroe | Fall 1990 | 9.8 | 0.003 | <0.1% | (McDaniel and Cox 1993) |
| Compared to Lake George (18,615 ha) | Mar 1984[†] | 15.4 | 0.0004 | <0.1% | (Cheek et al. 1984) |
| | Fall 1990[†] | 144.7 | 0.0004 | <0.1% | (Cross et al. 1993) |
| | Fall 1991[†] | 130.2 | 0.0004 | <0.1% | (Cross et al. 1993) |

*Adult loss was calculated for worst-case withdrawal scenarios assuming a lifetime fecundity of 100,000 eggs per female.

[†]Littoral samples

‡Limnetic samples

Table 23. Comparison of potential adult equivalent losses of bluegills due to total larval entrainment in 2008 and 2009 to recreational bream harvest statistics

| Withdrawal Location | Date | Number Harvested by Anglers | Adult Equivalents (*NA*) Lost to Entrainment | *NA* losses as a Percent of Sport Harvest | Harvest Reference |
|---|---|---|---|---|---|
| **Lake Poinsett (1,680 ha)** | | | | | |
| | Mar 79–Sep 79 | 2,481 | 196 | 7.9% | (Cox et al. 1981) |
| | Nov 79–Mar 80 | 5,289 | 196 | 3.7% | (Cox et al. 1981) |
| | Jan–Apr 2000* | 6,304 | 196 | 3.1% | (Eisenhauer et al. 2002) |
| | Jan–Apr 2002* | 19,023 | 196 | 1.0% | (Eisenhauer et al. 2002) |
| **SR 50** | | | | | |
| Compared to Lake Poinsett | Mar 79–Sep 79 | 2,841 | 25 | <0.9% | (Cox et al. 1981) |
| **Lake Monroe (3,500 ha)** | | | | | |
| | Avg. Oct–May 2002–2007† | 6,696 | 29 | 0.4% | (Holder et al 2005; 2007) |
| | Jan–May 2010[2] | 7,651 | 29 | 0.4% | (Holder et al. 2010) |
| **SR 46** | | | | | |
| River (near Mullet Lake) | Dec 97–Apr 98 | 2,273 | 149 | 6.5% | (Jenkins et al. 1999) |
| Compared to Lake Jesup | Jan–May 2006 | 3,429 | 149 | 4.3% | (Holder et al. 2007) |
| | Jan–May 2007 | 3,456 | 149 | 4.3% | (Holder et al. 2007) |
| Compared to Lake Monroe | Avg 2002–2007 | 6,696 | 149 | 3.7% | (Holder et al. 2005; 2007) |
| | Jan–May 2010 | 7,651 | 149 | 1.9% | (Holder et al. 2010) |
| **Yankee Lake** | | | | | |
| Compared to Lake Monroe | Avg 2002–2007 | 6,696 | 65 | 1.0% | (Holder et al. 2005; 2007) |
| | Jan–May 2010 | 7,651 | 65 | 1.0% | (Holder et al. 2010) |
| **SR 44** | | | | | |
| Lake Beresford Area | Jan–Dec 2000 | 20,160 | 142 | 0.7% | (Jenkins et al. 2002) |
| Compared to Lake George | Avg 1985–1990 | 12,841 | 142 | 1.1% | (Snyder et al. 1990) |

*Data from Lake Poinsett and Lake Harney combined (Cox et al. 1981).

†Includes bank harvest from seawall fisherman.

**American Shad (*Alosa sapidissima*)**

Adult

Yolk-sac larvae (7.2 mm TL)

Photo by Matt Scripter

American shad are anadromous herrings that spend most of their life at sea but return to natal freshwater rivers to spawn (Melvin et al. 1986; Weiss-Glanz et al. 1986). Genetically distinct populations of American shad, each specific to its natal river, extend from the St. Lawrence River in Canada to the St. Johns River in Florida (Limburg et al. 2003). American shad have supported important commercial and recreational fisheries along the east coast of the United States since the 1700s (McPhee 2002). However, recent data suggest that population abundances of American shad along the entire east coast, except for a few drainage basins, are at all-time historic lows (Limburg 2007).

American shad are long distance coastal migrants (Neves and Depres 1979; Dadswell et al. 1987). During the spring and early-summer, shad from Florida to Maine leave their natal rivers and over-wintering habitat and migrate en masse to summer feeding grounds in the Bay of Fundy. With the onset of fall and winter, they migrate from the Bay of Fundy down the east coast of the United States and form large, somewhat discrete aggregations off the coast of Florida and in the middle Atlantic Bight (Dadswell et al. 1987). During the spring, individuals from these aggregations, using both olfactory and visual cues, move into their natal rivers to spawn (Dodson and Leggett 1973; Melvin et al. 1986). For American shad populations south of the Carolinas, water temperatures are too high and migration distances too long for them to spawn more than once in a lifetime (McBride 2000). Thus, American shad south of the Carolinas, including the St. Johns River, spawn only once and die; whereas, American shad from rivers north of the Carolinas may spawn repeatedly over multiple years (Talbot and Sykes 1958).

Historically, American shad were abundant in the St. Johns River and supported valuable commercial and recreational fisheries (Walburg 1960a; McBride and Holder 2008). Currently, abundance of American shad in the St. Johns River is near historic lows but the population is considered stable (Trippel et al. 2007; McBride and Holder 2008; Holder et al. 2012). In the St. Johns River, American shad spawning occurs in the river channel primarily between Lake Monroe and Lake Poinsett (Williams and Bruger 1972; Williams et al. 1972). Areas of peak American shad spawning may shift spatially between years in response to interannual variability in river discharge (Williams and Bruger 1972).

### Temporal and Spatial Patterns of Distribution

A total of 13,729 larvae and 24 juvenile American shad were enumerated in processed samples. American shad larvae were collected at all sampling locations but were most abundant at SR 50 where mean larval densities in both 2008 and 2009 periodically exceeded 100 fish 100 $m^{-3}$ (Figure 26). At all the other stations, mean densities of American shad larvae rarely exceeded 5 fish 100 $m^{-3}$. American shad larvae began appearing in samples in late December and were present until mid to late May (Figure 26). Peak spawning occurred from early March through early April in both years.

### Size Distributions

American shad ranged in size from 6 mm TL yolk-sac larvae to 93 mm TL juveniles (Figure 27). Approximately 80% of the larvae captured were in the 6–8 mm TL size range and were in the yolk-sac or preflexion development stage (Figure 27). Abundance of American shad larvae varied little between daytime and nighttime samples. As determined by otolith analysis, the modal age of American shad larvae collected in 2008 and 2009 at SR 50 was 3–4 days (Boucher 2010).

### Entrainment Losses and Adult Equivalents

The estimated total number of American shad larvae potentially entrained during 2008–2009 under the worst-case full withdrawal scenarios ranged from 226,000 at Lake Monroe to 4.5 million at SR 50 (Table 24). At SR 50, approximately 7% of the larval transport by the site would have been subject to entrainment. At SR 46, potential entrainment was >13% of the larval transport. Estimated entrainment at SR 46 under the worst-case, 50 mgd withdrawal scenario was second highest of all the stations at 484,000 larvae (Table 24).

Studies on the fecundity of American shad have not been able to develop precise estimates of the numbers of eggs produced by females (Olney and McBride 2003). A study by Leggett and Carscadden (1978) assumed that females produced a single batch of eggs each year, which in the case of the semelparous St. Johns River stock of shad, is equivalent to the total lifetime production. However, Olney and McBride (2003) demonstrated that in the St. Johns River, American shad produce multiple batches of eggs during their single spawning season. Since batch spawning frequency and spawning duration are unknown, it is not possible to determine lifetime fecundity. Mean batch fecundity of St. Johns River American shad was approximately 50,000 eggs (Olney and McBride 2003). Lacking information on the number of times a female may actually spawn, 50,000 eggs was used as a lower estimate of total lifetime fecundity. For an upper limit, it was assumed a female may spawn up to five times, generating a lifetime fecundity estimate of 250,000 eggs per female.

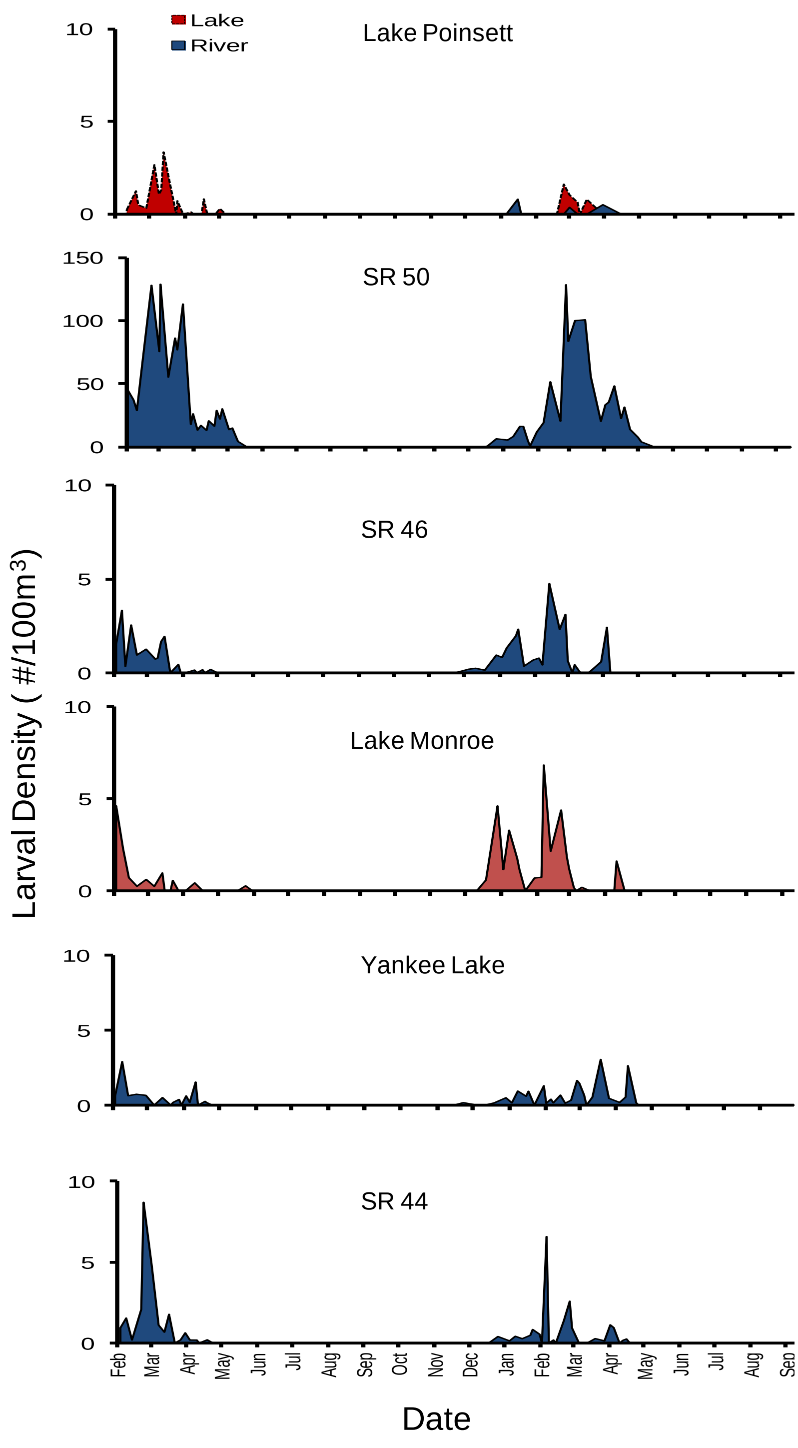



Figure 26. Mean density (number 100 $m^{-3}$) of American shad larvae collected on each sampling date in 2008 and 2009

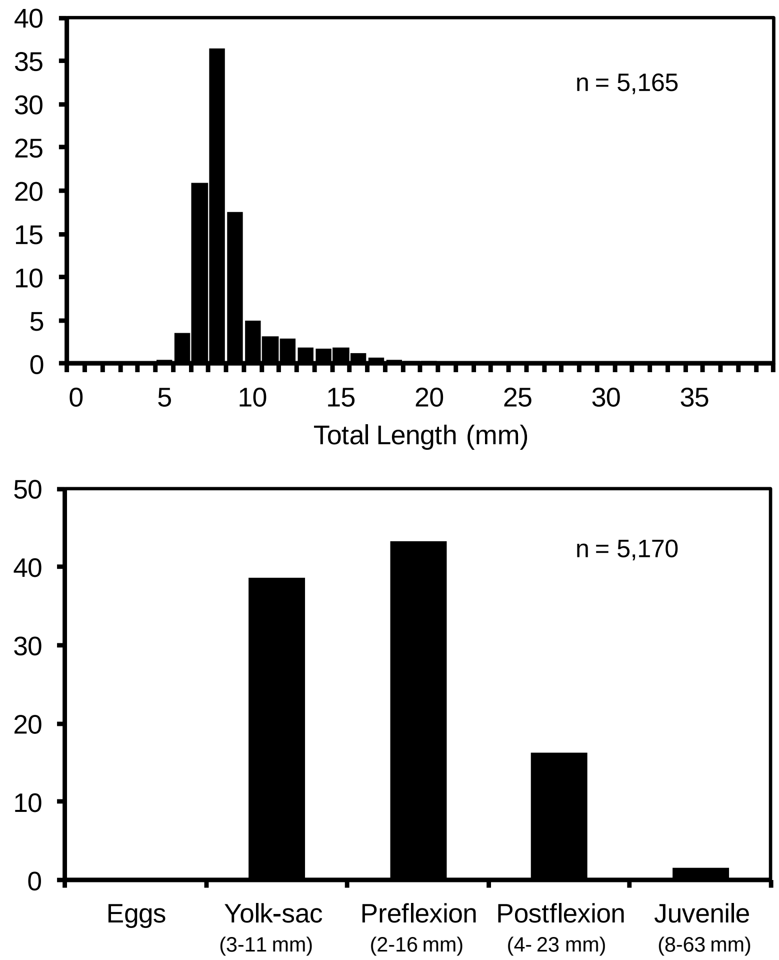



Figure 27. Length-frequency and life stage data of American shad collected in 2008 and 2009. Size ranges (TL) of various developmental stages are in parentheses.

The modal age of American shad larvae captured was approximately 4 days (Boucher 2010), which was used to back calculate the number of egg equivalents ($NE_{eggs}$) that would have produced the observed number of larvae under daily mortality rates of 5%, 10%, and 25%. Adjusted egg numbers were then used to calculate *NA* for the two lifetime fecundity estimates (Table 24).

The potential *NA* loss of American shad during 2008–2009 was highest at SR 50 (424 adults; 212 adults annually) under a variable water withdrawal scenario assuming a lifetime fecundity estimate of 50,000 eggs per female and a daily larval mortality rate of 25% (Table 25). For all the scenarios considered, estimated *NA* loss at SR 50 was at least an order of magnitude higher than *NA* loss at any of the other stations. If higher water withdrawals than those modeled here were allowed at SR 50 during the low discharge conditions (Table 1), NA loss at SR 50 would be considerably higher.

While a 25% daily mortality rate of recently hatched American shad larvae is believed to be the most reasonable estimate of actual mortality during the first few days of life, the lifetime fecundity estimate of 50,000 eggs per female is likely low. However, if a lifetime fecundity estimate of 250,000 eggs per female is assumed along with a 25% daily larval mortality rate, estimated *NA* loss at SR 50 was still 85 adults (43 adults annually). It is also noteworthy that withdrawals from Lake Poinsett, a site being considered as an alternative to SR 50, would not result in any predicted *NA* loss of American shad (Table 25).

Table 24. Estimated total number of American shad larvae potentially entrained during 2008–2009 under various water withdrawal scenarios

| Withdrawal Location | Withdrawal Scenario* | Estimated Total Number of Larvae Entrained (1,000's) | % of Total Larval Transport Entrained† |
|---|---|---|---|
| Lake Poinsett | Variable | 0 | 0 |
| SR 50 | Variable | 4,469 | 7.04 |
| SR 46 | 25 mgd | 242 | 6.82 |
| | 50 mgd | 484 | 13.64 |
| Lake Monroe | 10 mgd | 113 | NC |
| | 20 mgd | 226 | NC |
| Yankee Lake | 25 mgd | 178 | 1.44 |
| | 50 mgd | 356 | 2.87 |
| SR 44 | 25 mgd | 266 | 3.18 |
| | 50 mgd | 512 | 7.36 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

†NC = Not Calculated.

All anadromous herring stocks in the United States, including American shad, are currently managed under plans overseen by the ASMFC and restoration efforts are now underway to rebuild populations in many rivers (ASMFC 1999; ASMFC 2009a; ASMFC 2009b). As a part of this initiative, Florida is required to conduct annual surveys of on-going shad fisheries (Holder et al. 2012). In 2011 the FWC submitted to the ASMFC an American shad sustainable fishing plan for the St. Johns River (Hyle 2011). The recommendation of the FWC was to maintain harvest of American shad from the St. Johns River at current levels. Any increase in American shad harvest not accompanied by a concomitant increase in independent spawning stock abundance estimates would be offset either by a reduced bag limit (currently 10 fish $d^{-1}$) or by spatial or temporal closures of the fishery (Hyle 2011). In 2011 and 2012, recreational harvest of American shad from the St. Johns River was estimated to be 198 and 232 fish, respectively (Hyle 2011; Holder et al. 2012).

As a part of the statewide management plan, Florida will also be required in 2016 to submit a habitat management plan to the ASMFC that includes a summary of current and historical nursery habitats and list potential water resource development projects that may impact those habitats (e.g., water supply withdrawal projects; ASMFC 2009a; ASMFC 2009b). States are required to carefully scrutinize water withdrawal projects and develop management plans that ensure protective flows and levels are maintained and potential entrainment/impingement impacts are minimized (ASMFC 2009a; ASMFC 2009b). Any increase in mortality of adult American shad caused by water development projects may also have to be offset by reduced bag limits in the recreational fishery or by spatial or temporal fishery closures.

Although predicted *NA* losses of American shad presented in this report appear to be relatively small, they obviously could result in a significant increase to the adult mortality caused by recreational fishing. Proposed full water withdrawals at SR 50 alone under the

Table 25. Predicted adult equivalent (*NA*) loss of American shad due to larval entrainment during 2008–2009 combined under various water withdrawal scenarios and larval mortality rates. Annual estimated *NA* loss equals half the values given.

| Withdrawal Location | Withdrawal Scenario* | Estimated Adult Equivalent Loss (*NA*) | | | | | |
|---|---|---|---|---|---|---|---|
| | | Fecundity =50,000 eggs/female | | | Fecundity =250,000 eggs/female | | |
| | | Larval Mortality Rate (per day) | | | Larval Mortality Rate ( per day) | | |
| | | 5% | 10% | 25% | 5% | 10% | 25% |
| Lake Poinsett | Variable | 0 | 0 | 0 | 0 | 0 | 0 |
| SR 50 | Variable | 208 | 245 | 424 | 42 | 49 | 85 |
| SR 46 | 25 mgd | 11 | 13 | 23 | 2 | 3 | 5 |
| | 50 mgd | 23 | 27 | 46 | 5 | 5 | 9 |
| Lake Monroe | 10 mgd | 5 | 6 | 11 | 1 | 1 | 2 |
| | 20 mgd | 11 | 12 | 21 | 2 | 2 | 4 |
| Yankee Lake | 25 mgd | 8 | 10 | 17 | 1 | 1 | 2 |
| | 50 mgd | 17 | 20 | 34 | 2 | 2 | 4 |
| SR 44 | 25 mgd | 12 | 15 | 25 | 2 | 3 | 5 |
| | 50 mgd | 25 | 29 | 50 | 5 | 6 | 10 |

*Variable withdrawal rates used to calculate entrainment at Lake Poinsett and SR 50 are presented in Table 1.

most favorable scenario (250,000 eggs per female, 5% daily larval mortality rate) would increase estimated annual adult loss by 21 fish (9%). With a 25% daily mortality rate and 250,000 eggs per female fecundity, annual adult loss would increase by 43 fish (18%). If a lifetime fecundity of 50,000 eggs per female is assumed, withdrawals at SR 50 alone could increase annual adult loss by more than 200 fish (> 44%) regardless of the daily mortality rate used.

Based on the information presented, SR 50 is least desirable as a water withdrawal site due to potential negative impacts to American shad. Should withdrawals occur at SR 50, they should be cut-off or curtailed during the December through April peak shad spawning season. Given the lack of *NA* loss of American shad at Lake Poinsett, this is a more preferable site for withdrawals as opposed to SR 50.

Without potential entrainment effects at SR 50, potential *NA* losses of American shad are considerably smaller. For example, if worst-case, 50 mgd withdrawals at Lake Poinsett, SR 46, Yankee Lake, and SR 44 were to occur simultaneously (200 mgd total), the total *NA* loss of American shad (lifetime fecundity 250,000 eggs) due to entrainment during 2008 and 2009 would only be 12–23 individuals (6–12 annually) depending on the daily larval mortality rate used (Table 25). Assuming a lifetime fecundity of 50,000 eggs, predicted *NA* losses over 2008–2009 range from 65 to 130 individuals (33 to 65 annually). With the lower lifetime fecundity estimate, total annual adult-loss would increase 17% – 33% over losses that occur due to angling only. With a higher lifetime fecundity estimate, the total increase in *NA* loss due to angling alone would be negligible. However, these worst-case estimates of adult loss based on a withdrawal rate of 200 mgd total are an overestimate since the maximum combined surface water withdrawal rate from the river is not anticipated to exceed 155 mgd.

In the absence of water withdrawals at SR 50, potential entrainment effects on American shad in the St. Johns River are likely minor to moderate. If potential withdrawals at SR 50 are included, potential effects may be significant; however, despite these estimates, potential entrainment of American shad at the SR 46 intake station may be considerably higher under higher flow conditions than were observed in this study. Historical data on American shad spawning suggests the river reach adjacent to the SR 46 withdrawal site constitutes important American shad spawning habitat under high flows (Williams and Bruger 1972). During ichthyoplankton sampling December through April, average flows at SR 50 were low (<282 cfs), a condition associated historically with greater American shad spawning upstream of Lake Harney (Williams and Bruger 1972). However, during higher flows (SR 50 discharge >706 cfs), a condition that did not occur during the ichthyoplankton sampling window of this study, William and Bruger (1972) found the majority of American shad spawning occurred downstream of Lake Harney near the SR 46 site. Thus, there is a special need to minimize ichthyoplankton entrainment at this location.

In addition, the area around the SR 46 withdrawal location is also known to be important spawning habitat for two other anadromous river herrings, hickory shad (*Alosa mediocris*) and blueback herring (*Alosa aestivalis*; Walburg 1960b). Both hickory shad and blueback herring are also regulated by the ASMFC, but little is known of their life history (Harris et al. 2007; Trippel et al. 2007). During the 2008–2009 surveys, hickory shad and blueback herring larvae were most abundant at SR 46 (Table 3); although, the numbers collected were too low to calculate potential *NA* loss due to entrainment. Previous studies however documented substantial spawning of these two species near the proposed SR 46 withdrawal location (Williams et al. 1972; Harris et al. 2007).

Efforts to minimize larval fish entrainment should be included in the design of all intake structures that withdraw water from the St. Johns River. Even though withdrawal effects may be minimal for the majority of fish species, minimizing larval entrainment, by utilizing practical design features is prudent, as well as fundamental to ensuring fish stocks in the river are protected to the greatest extent possible. Potential intake design features that minimize entrainment include, but are not limited to, (1) constructing the intake so that passive ichthyoplankton being transported downriver are deflected away from the actual withdrawal location, (2) installing wedge wire screens with small mesh sizes to filter the water being withdrawn and, (3) limiting inflow velocities of the withdrawal (Gowan et al. 1999). Based on an extensive review of the swimming stamina of small fishes, Gowan (1999) recommended that approach inflow velocities into intake structures should be < 0.8 m $s^{-1}$ (0.25 ft $s^{-1}$). Through careful engineering design, entrainment of all vulnerable ichthyoplankton at each of the proposed water withdrawal locations in the St. Johns River can be minimized. Lacking detailed information on specific intake designs, however, only generalizations about potential effects of water withdrawals at any specific location can be made at this time.

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