APPENDIX 12.E. POTENTIAL WITHDRAWAL EFFECTS ON THE LITTORAL ZONE, MARSH AND FLOODPLAIN SMALL FISHES ASSEMBLAGE

The Littoral Zone, Marsh and Floodplain Small Fishes Assemblage is dominated in numbers and diversity by poeciliids (live-bearers) and cyprinodontids (killifishes). Members of this assemblage are small (<8 cm standard length (SL)) and found almost exclusively in association with dense submersed or emergent vegetation (Barnett and Schneider 1974; Loftus and Kushlan 1987). All of these species generally have short life spans and mature rapidly, have protracted spawning seasons and are tolerant of low DO and high water temperatures (Carlander 1969a; Loftus and Kushlan 1987; Marcy et al. 2005; McLane 1955). They are well adapted for colonizing newly flooded habitats and consequently, they are the most abundant fishes found in St Johns River marshes or on the floodplain. Members of this assemblage may be herbivores, detritivores, or invertivores and most lay eggs on plants (phytophils). Members of this assemblage play an important role in regulating energy flow through the system. They convert primary production, detritus and invertebrates into secondary or tertiary biomass that then becomes available to higher level predators (Loftus and Kushlan 1987). In ecosystems with pulsed hydroperiods like the St. Johns, the concentrating of small fishes that occurs with receding water levels creates an important food source for wading birds (Ogden 1994) as well as for fish predators that occupy more permanently flooded habitats. Since small juvenile sunfishes (e.g., bluegill) are often associated with dense emergent vegetation, they are also included in this assemblage.

The small fishes assemblage can be quite abundant in dense submersed littoral vegetation (SAV) reaching densities up to 2,500,000 fish ha^{-1} (1,011,736 fish ac^{-1}) and biomass up to 619 kg ha^{-1} (552 lbs ac^{-1}) (Barnett and Schneider 1974; Chick and McIvor 1994; Haller et al. 1980). Because withdrawals will not affect SAV in the St. Johns River (see Chapter 9 Submersed Aquatic Vegetation), we will not consider littoral populations of this assemblage further. However, we will assess the potential effects of reduced floodplain inundation predicted under withdrawal scenarios that cause reduced inundation of the floodplain (e.g., the Full1995PS Scenario).

Little is known about the abundance and the population dynamics of the small fish community that occupies the marshes and floodplain of the St. Johns River system. The most extensive sampling of this community has been in the marshes around Blue Cypress Lake where Jordon et al. (1998), reported an average density of small fishes of 280,000 fish ha^{-1} (113,000 fish ac^{-1}) with an average biomass of 35.3 kg ha^{-1} (31.5 lbs ac^{-1}). More limited floodplain sampling by FWC near Lake Washington (Cox et al. 1977; Cox et al. 1976), and Puzzle Lake (Cox et al. 1980) indicated densities of the small fish assemblage of 45,000 to 230,000 fish ha^{-1} (18,000 to 82,600 fish ac^{-1}) with total biomasses ranging from 15 to133 kg ha^{-1} (13.4 to 100.8 lbs ac^{-1}). Although limited sampling of small fishes has occurred in St. Johns River marshes, extensive sampling of this community has been conducted in Everglades (Kushlan 1980; Loftus and Eckland 1994; Loftus and Kushlan 1987; Trexler et al. 2002). In the Everglades, duration of flooding, is the most important factor regulating abundance of small marsh fishes (Kushlan 1976; Kushlan 1980; Loftus and Eckland 1994; Trexler et al. 2002).

The species composition of small fishes in Everglades marshes appears remarkably similar to the species composition in the marshes of the St. Johns River (Figure 12.E-1). Both are dominated by poeciliids, cyprinodontids, elassomatids, and fundulids. Small or young-of-year (YOY) of the larger centrarchid species (e.g., bluegill) are present but are relatively rare. Dominant species in the marshes include mosquitofish, bluefin killifish, least killifish, flagfish, Everglades pygmy sunfish, sailfin molly, and golden topminnow. As conductivity increases in the section of the St. Johns River near Puzzle Lake, flagfish are replaced by sheepshead minnow; a similar species shift in response to conductivity was noted in the Everglades (Loftus and Kushlan 1987).

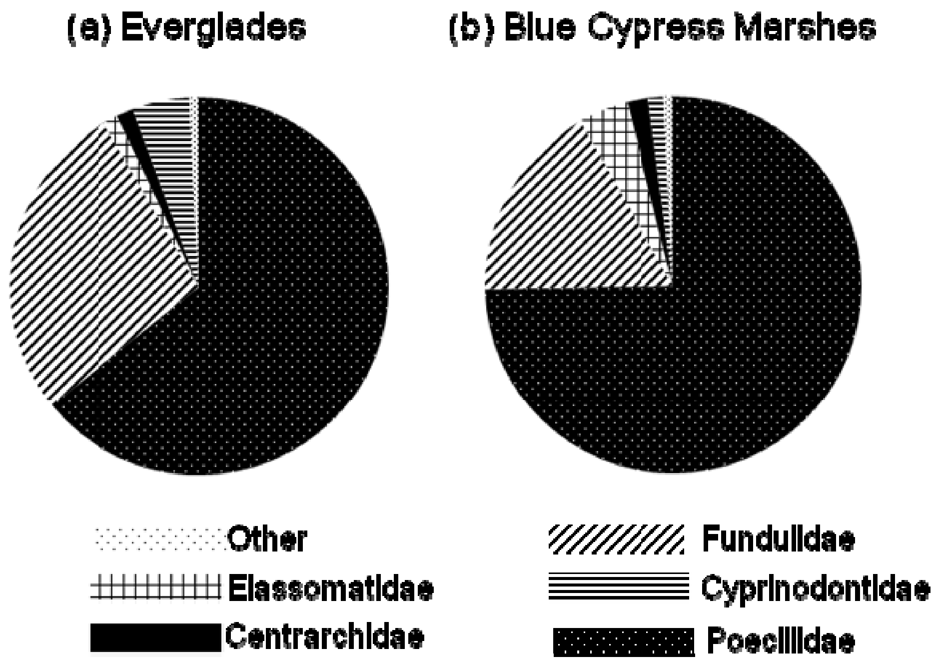

Figure 12.E-1. Community composition of the small fish assemblage in the (a) Everglades (Trexler et. al 2002) and (b) Blue Cypress marshes (Jordan et. al. 1998).

Because of the similarities between the community composition of small fishes in the Everglades and St. Johns River floodplain marshes, we felt predictive relationships between flooding duration and small fish (<8 cm SL) abundance developed from Everglades data, could be useful for assessing potential effects of water withdrawals on the St. Johns River floodplain. Specifically we used output from a pulsed-hydroperiod model developed by DeAngelis et. al. (1997). This model predicts small fish abundance responses in the Everglades to varying periods of flooding and drying for a 25-ha (61.8-ac) spatial cell in which microtopography was variable and a fraction of the cell contained permanent water. Their model predicted there was a minimum flooding threshold of approximately 9 months that the small fish assemblage needed to reach maximum sustainable densities (Figure 12.E-2). If the hydroperiod was less than 6 months, the density of small fishes remained only a few fish m^2. In addition the model suggested that large piscivorous fish do not have a significant impact on small fish populations in the marsh as they do on permanent water bodies, and, that the small fish assemblage repopulated rapidly (within less than a year) after a drydown.

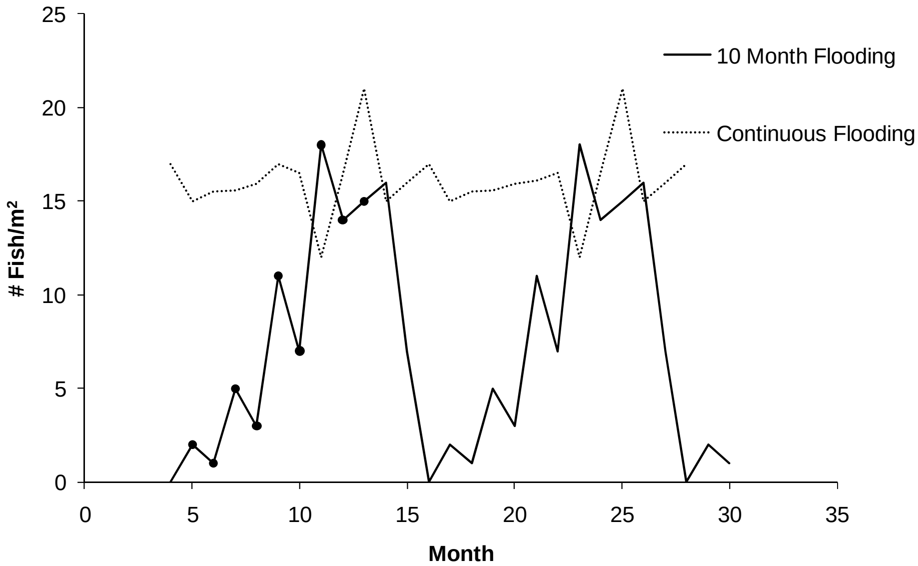

Figure 12.E-2. Simulated small fish densities through the year from the Everglades small fish production model (DeAngelis et.al. 1997) for two different hydroperiods. This figure, adapted from Figures 8d and 8e in DeAngelis et. al. 1997, compares predicted small fish densities under continuous flooding to predicted densities under a hydroperiod of 10 months flooding and two months dry. Solid circles represent points used to develop the regression in Figure 12.E-3. According to DeAngelis et. al. (1997), maximum small fish densities are reached after approximately 9 months of continuous flooding.

We felt that predicted small fish abundances from the Everglades model for a hydroperiod of 10 months of flooding followed by 2 months of drying would be most applicable for predicting abundance on the St. Johns River floodplain. Comparable to the Everglades model physical structure inputs, the St Johns floodplain has relatively low topographic relief and may be flooded extensively for varying periods of the year depending on rainfall. The entire floodplain dries annually for periods ranging from weeks to a few months. Shallow depressional areas are abundant and provide for microtopograhic variability while numerous permanently flooded remnant river channel sloughs, along with the braided river channel and the larger lakes, provide dry season refugia and nearby “seed” sources for floodplain recolonization.

From the results presented in DeAngelis et. al. (1997) (Figure 12.E-2), we generated a linear regression through the predicted monthly small fish densities for each of the first 9 months of continuous flooding (Figure 12.E-3). This regression (r^2=0.78) was then used to generate predicted annual estimates of the maximum abundance of small fishes produced on the St. Johns River floodplain using annual flooding durations predicted by the St. Johns River hydrologic model as the predictive variable .

For our analyses, we used the regression in Figure 12.E-3 to calculate annual maximum small fish density estimates attained along seven MFL transects that are located intermittently along the river channel starting at Lake Poinsett and extending downstream to Lake Woodruff. MFL transect sites for which densities were calculated included two transects on Lake Poinsett (I95 on the southeast corner and County Line on the northwest corner), Toso 528, H1, Lake Monroe T4,

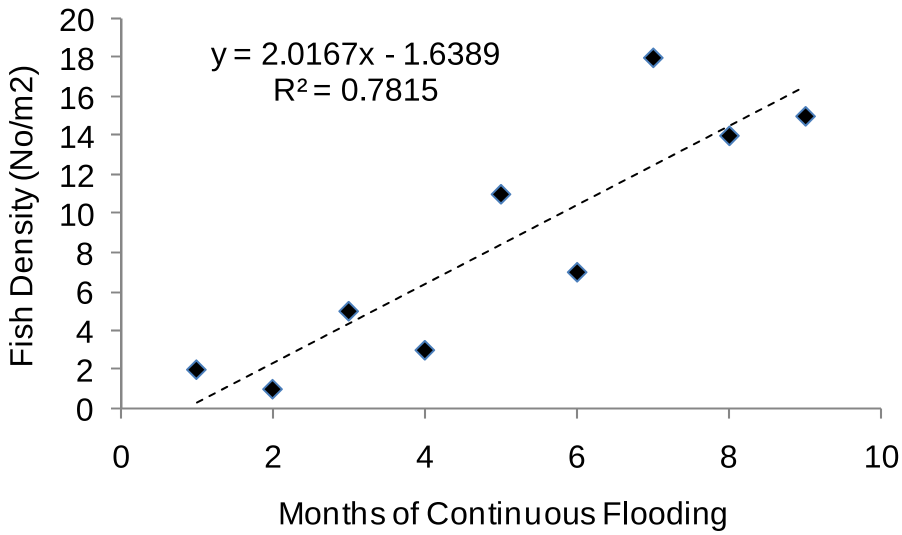

Figure 12.E-3. Relationship between months of continuous flooding and predicted density of the small fish assemblage by the Everglades model (DeAngelis et al. 1997; see Figure 12.E-2).

Pine Island, and Lake Woodruff (See Figure 3-1 on page 9 of Chapter 12 Fish for map of MFL transect locations).

We used four categories of continuous flooding duration to estimate maximum small fish densities attained along each transect cross-section during the annual flooding cycle. The flooding duration categories were: 1) flooded <1 month, 2) flooded 1-6 months, 3) flooded 6-9 months, and 4) flooded > 9 months. Areas flooded < 1 month were assigned a small fish density value of 0. Areas flooded 1-6 months (mean =3.5 months) were assigned a density of 5.4 fish m^2 ((2.0167 X 3.5) – 1.639 = 5.4; Figure 12.E-3). Areas flooded 6-9 months (mean = 7.5 months) and > 9 months were assigned densities of 13.4 fish m^2 and 16.5 fish m^2, respectively. We chose four flooding durations to simplify calculations and because the boundary values for these categories were easily obtainable from the hydrologic model output. In addition, we do not feel further sub-dividing the flooding duration categories would substantially alter the results. For a schematic representation of how we calculated maximum small fish density estimates see Figure 12.E-4. It is important to note that our model calculates only maximum total densities and does not take into consideration the concentration effect of small fishes that will occur as water levels fall, although there is some question as to the extent to which this occurs (Trexler et al. 2002). In addition, our approach does not consider possible habitat structure effects on small fish density.

For each transect, surveyed elevational data were plotted to create a cross-sectional stage-distance curve. For each one cm (0.4 in) elevation above the bankfull water level ,the cumulative distance (m) along each transect was then calculated, and compared to flooding duration data (determined for each modeled June 1- May 31 water year) to determine how long each cm depth interval was flooded annually. The total number of m along the transect occupied by each cm depth interval was then multiplied by a maximum small fish density that was assigned based on its flooding duration. In this way, every meter along each transect across the floodplain was assigned an annual small fish density (Figure 12.E-4). Data were then summed for the entire transect by year and divided by transect length to get an annual mean density estimate (number of fish m^{-2}). For reporting, data were standardized to number fish ha^{-1}. To calculate biomass estimates each small fish was assigned a wet weight of 0.4 g. This is the average weight of small fishes collected in the St. Johns River marsh around Lake Washington and Puzzle Lake (Cox et al. 1980; Cox et al. 1977).

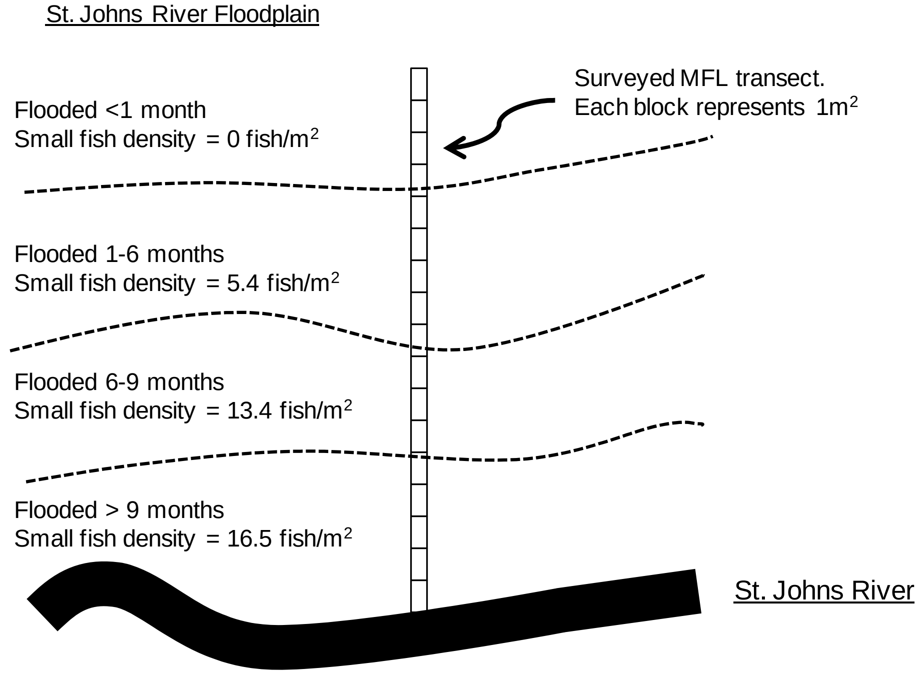

Figure 12.E-4. Aerial view of a schematic diagram showing how maximum small fish densities for each year of the hydrologic model output were generated. Predicted small fish densities were calculated for each 1 m^2 along the transect. Flooding durations along the transect were calculated for each 1 cm (0.4 in) increment above the bankfull water level. Abundance data generated for each 1 m^2 block along the transect were summed and then divided by the total length of the transect to get an average maximum density in fish m^{-2}. The average maximum number of fish m^{-2} was then multiplied by 10,000 to generate the number of fish ha^{-1}.

Our approach was simplistic in that it only considered flooding and drying as an annual event and did not consider potential cumulative impacts of multi-year droughts (DeAngelis et al. 1997). We felt this approach was appropriate because the occurrence and return frequency of droughts (defined as water contained solely within the channel banks) were unaffected by water withdrawals under any scenarios that included a completed Upper Basin Project (e.g., Full1995PS; See Chapter 12 Fish Section 4.1.1 Effects on Hydrology). In addition, drought intensity (defined by the lowest level reached when water levels were solely contained within the channel banks) was also unaffected or increased under any scenario that included a completed Upper Basin Project.

At the Lake Poinsett I95 transect, estimated annual maximum densities of the floodplain small fish assemblage varied widely in response to modeled hydrologic conditions (Figure 12.E-5).

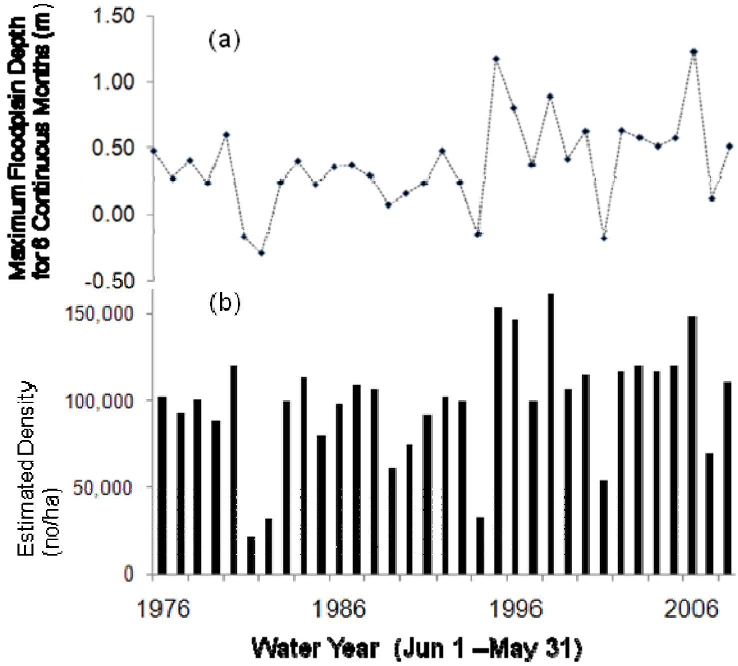

Figure 12.E-5. Plots of (a) maximum water depth (m) on the floodplain that lasted at least 6 continuous months along the Lake Poinsett I95 transect and, (b) mean annual maximum density estimates of small fishes produced on the floodplain (no ha^{-1}) during each corresponding annual wet season. Six-month continuous flooding levels are only a surrogate metric used to indicate the intensity and duration of the entire wet season flooding cycle.

During predicted wet years (e. g.,1995, 1996, and 1998) when the model predicted that most of the floodplain was flooded at least 6 months, estimated maximum mean densities of the small fish assemblage approached 150,000 fish ha^{-1} (60,704 fish ac^{-1}) with a biomass of 60 kg ha^{-1} (54 lbs ac^{-1}). During predicted drought years (e.g.,1980 and1981) when water levels would inundate the floodplain only a few months, maximum mean densities were less than 30,000 fish ha^{-1} (12,141 fish ac^{-1}) with a biomass of 12 kg ha^{-1} (11 lbs ac^{-1}). Extrapolating these estimates to the entire 2210 ha (5,461 ac) floodplain around Lake Poinsett, yields estimates of total maximum abundance on the floodplain of 33.2 X 10^7 individual fish (132,600 kg; 292,383 lbs) during wet years and 6.6 X 10^7 individuals (26,502 kg; 58,477 lbs) during dry years.

We examined the variation among scenarios in estimates of mean density and biomass to evaluate potential differences due to water withdrawals. In this work, we used the percent change from the base scenario (Base1995NN) rather than variation in the actual numbers themselves. Numbers may not be directly comparable because they are influenced by site-specific variability in microtopography. There were other datasets such as output from the digital elevation model (DEM) that also provide a basis for calculating small fish maximum density estimates on the floodplain. Although these additional data sources were limited (both in spatial coverage and availability at the time of our analyses), we were able to generate floodplain small fish densities using early DEM data for an area of the floodplain north of SR 50. Withdrawal effects on maximum small fish abundance generated using the DEM were comparable to those obtained using nearby MFL transects. Under the Full1995NN Scenario maximum abundance of small fishes declined by an average 8.8% compared to the Base1995NN Scenario using the DEM. Declines predicted using the two MFL transects located nearest to SR 50 (Tosohatchee North and H-1) were 7.2% and 10.3% (average= 8.8%), respectively. Thus, we concluded that the two methods provided comparable results. Because using MFL transects allowed a much greater spatial assessment of withdrawal effects and to maintain consistency, we chose to only report results here using the MFL transects.

The Full1995NN Scenario had the largest reductions below the Base1995NN Scenario in estimated maximum floodplain densities of small fishes (Figure 12.E-6). Upstream of Lake Monroe estimated reductions ranged from 9% at the H1 Transect to 11% at the County Line Transect. From Lake Monroe and downstream estimated reductions ranged from 0.2% at the Lake Woodruff Transect to 10% at the Pine Island Transect. Addition of the Upper Basin Project (Full1995PS) resulted in only small reductions in estimated small fish densities around Lake Poinsett (I95 and County Line Transects; Figure 12.E-6) from the Base1995NN Scenario. At the Toso 528 and H1 Transect estimated maximum small fish densities declined approximately 5%. From Lake Monroe downstream, sea level rise caused dramatic increases in flooding durations that resulted in substantially higher estimates of small fish abundances on the floodplain. At the Lake Woodruff Transect, estimated small fish densities under the Full1995PS and Full2030PS Scenarios were more than 60% higher than under the Base1995NN Scenario (Figure 12.E-6). In addition, hydrologic predictions for the Full2030PS Scenario resulted in estimates of maximum small fish densities on the floodplain higher than the Base1995 Scenario at all sites upstream of Lake Monroe (Figure 12.E-6).

Under the Full1995PS Scenario, maximum floodplain densities of small fishes declined approximately 4-5% from the Base1995NN Scenario between Lake Poinsett and at Lake Harney. Roughly spreading this predicted reduction out over the entire 16,314-ha (40,312 ac) of floodplain between these lakes Harney generates a predicted loss of 4.3×10^6 (17,312 kg; 38,174 lbs) to5.7×10^7 (22,846 kg; 50,375 lbs) of small fishes annually. The most important effects of this reduction may the loss of potential food resources to species such as wading birds that utilize the floodplain as important foraging habitat and a reduction in small prey input to the river. Although not presented here, results for the Half1995PS Scenario eliminates the 4-5% in small fish reduction predicted for the Full1995PS Scenario. Higher water levels resulting from 2030 land-use changes (Full2030PS) also eliminates any reduction effect on small fish abundance associated with withdrawals (Figure 12.E-6).

Although flooding duration was found to be the most important factor affecting small fish abundance in the Everglades marshes, there are other factors that also may have an influence. These include vegetation type, density and spatial distribution (Chick and McIvor 1994; Jordan et al. 1998; Trexler et al. 2002), spatial variation in nutrient biogeochemistry (DeAngelis and White 1994), distance from deep water refugia (Trexler et al. 2002), varying patterns of species recovery (DeAngelis and White 1994; Trexler et al. 2002), and anthropogenic nutrient inputs (Trexler et al. 2002). There are undoubtedly differences in these other factors between the Everglades and the St. Johns River basin, as well as differences within the St. Johns River Basin itself, which cannot be accounted for. However, given the overwhelming influence of flooding

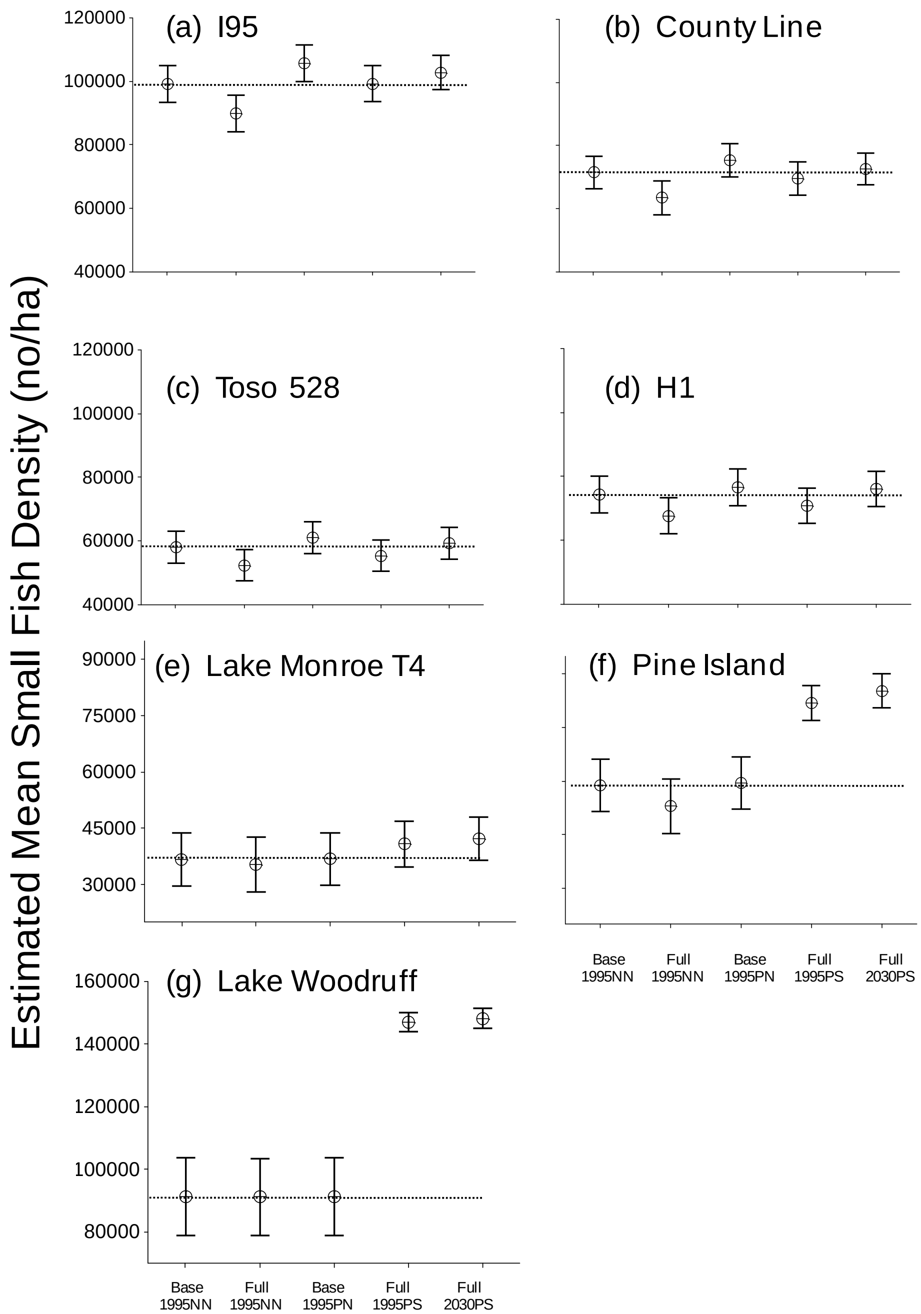

Figure 12.E-6. Estimated mean density (± 1 SE) of small fishes produced on the floodplain at seven MFL transects located in the Upper and Middle Basins from Lake Poinsett to Lake Woodruff for five withdrawal scenarios. The dashed horizontal lines correspond to the Base1995NN Scenario mean for scenario comparison. For the I95, County Line, Toso 528 and H1 MFL Transcts the Full1995PS and Full2030PS scenarios are synonmymous with the Full1995PN and Full2030PN scenarios, respectively. Sea level rise did not influence water levels at these transect locations..

duration on the abundance of small fishes as a group (DeAngelis et al. 1997), we feel our approach is valid for making a relative assessment of effects on abundance and biomass due to water withdrawals. Although abundance estimates of small fishes in the St. Johns River basin (4-28 fish m^{-2}) are very similar to those reported for the Everglades, results presented here are useful only for scenario comparisons at individual sites and do not represent accurate densities on the floodplain at a given point in time.

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