CHAPTER 12 FISH

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ACRONYMS, ABBREVIATIONS, AND CONVERSION FACTORS

ASMFC	Atlantic States Marine Fisheries Commission
CI	95% Confidence interval
CW	Carapace width
DEM	Digital elevation model
DO	Dissolved oxygen
DW	Disk width
EFDC	Environmental Fluid Dynamics Code
EPA	Environmental Protection Agency
ETM	Empirical Transport Model
FIM	Fisheries-Independent Monitoring
FWC	Florida Fish and Wildlife Conservation Commission
HSPF	Hydrologic Simulation Program–FORTRAN
LSJRB	Lower St. Johns River basin
$m^3 s^{-1}$	Cubic meter per second
MFL	Minimum flows and levels
POH	Post orbital head width
PRESS	Predicted residual error of sum of squares
PRESS r^2	PRESS coefficient of determination
r^2	Linear regression coefficient of determination
River km (mi)	Distance in kilometers (miles) from the river mouth
SAB	South Atlantic Bight
SAS	Statistical Analysis System
SAV	Submersed aquatic vegetation
SJRWMD; District	St. Johns River Water Management District
SL	Standard length (total length minus length of caudal fin)
SR	State Road
TL	Total length (tip of snout to tip of longer lobe of caudal fin)
TSS	Total sum of squares
USGS	United States Geological Survey
USJRBP	Upper St. Johns River Basin Project
WSIS	Water Supply Impact Study
YOY	Young-of-the-year

1 ABSTRACT

Water withdrawals have the potential to effect St Johns River fish populations directly by physically removing individuals from the system through entrainment or impingement, or indirectly, through effects on habitat. To evaluate potential fish loss to entrainment we conducted extensive ichthyoplankton (eggs and newly hatched fish larvae) surveys at six locations under consideration as potential withdrawal sites. In 2008 and 2009 over 708,000 fish eggs and larvae representing 16 species were collected. At all locations, gizzard shad (*Dorosoma cepedianum*) and threadfin shad (*Dorosoma petense*) were dominant, together comprising >69% of the total catch. Other abundant species were clown goby (*Microgobius gulosus*), naked goby (*Gobiosoma bosc*), black crappie (*Pomoxis nigromaculatus*), and bluegill (*Lepomis macrochirus*). Highest ichthyoplankton catches occurred at State Road (SR) 46 where Lake Jessup connects to river and the lowest catch rates were in the river channel near SR 50. Although total catch at SR 50 was low, American shad (*Alosa sapidissima*) were larvae more than 10 times more abundant at this site than at any other. Due to the high abundance of American shad eggs and larvae at SR 50, we recommend withdrawals not be taken at this location. At a minimum, any SR 50 withdrawals should be curtailed during the December through April American shad spawning season. All withdrawal intake structures should include design features that minimize potential ichthyoplankton entrainment. Design features include intakes that deflect passive ichthyoplankton away from the structure, the installation of wedge wire screens with small mesh sizes to minimize approach velocities to the intake, and limiting inflow velocities perpendicular to the intake screens.

Predicted declines in flow and water level caused by withdrawals were evaluated by considering potential effects on fish recruitment, growth, mortality, distribution, and abundance. Where predictive models were available, withdrawal effects were quantified by calculating the deviation of the predicted response from the 1995 baseline condition. Water withdrawal scenarios evaluated varied in the following conditions: 1) the volume of water withdrawn (Base = 0,Half = 3.4 m^3 s^{-1} (72.5 mgd), Full = 6.8 m^3 s^{-1} (155 mgd), 2) present (1995) or future (2030) land use; 3) completion of the Upper St. Johns River Basin Project (USJBP) (N = incomplete project, P = complete project) ; and 4) magnitude of sea-level rise (N = current or S = accelerated). As an example, the Full1995PN Scenario designation represents a potential scenario with a full withdrawal of 6.8 m^3 s^{-1} (155 mgd), under 1995 land-use conditions with the USJBP completed and sea-level rise occurring at the current rate.

Maximum proposed water withdrawals of 6.8 m^3 s^{-1} (155 mgd) are predicted to have relatively small effects on water levels and flows in the freshwater reaches of the St. Johns River. Consequently, predicted withdrawal effects on freshwater fishes in these reaches are mostly negligible. Under the worst-case Full1995NN Scenario, mean water levels between Lakes Poinsett and Harney are predicted to drop only 5.5 cm (2 in). The only notable predicted effect on fishes was a 10% average annual decline in the maximum density of small forage fishes produced on the floodplain. Reduction in small fish production was a direct result of withdrawal effects on the spatial extent of the floodplain inundated 6 months or longer. Flow augmentation from the USJRBP with full withdrawals (Full1995PN) increased predicted water levels and flows from the Full1995NN Scenario, however, full withdrawals still caused a 5% reduction in the average annual predicted floodplain production of small fishes. Although chronic annual

reductions in small fish production could potentially affect predator populations that prey on these fishes, wide interannual variability in floodplain inundation will likely make overall effects of flooding reductions on small fish production difficult to detect. Low flow augmentation from the USJRBP along with increased runoff associated with land-use changes reduced potential withdrawal effects on small fish production on the floodplain for all 2030 scenarios considered. Withdrawal effects on water levels and flows during drought conditions under all scenarios were also minimal because modeled withdrawals from the river ceased when total discharge at SR 50 fell below 8.9 $m^3 s^{-1}$ (300 cfs). Seasonality of water level fluctuations and water level recession rates were also minimally affected by withdrawals.

Withdrawal effects on estuarine fishes were evaluated using linear regressions relating monthly distribution and relative abundance of fishes derived from monthly catch data collected over the 10 years from 2001 to 2010 to freshwater inflow. Modeled inflow data for each withdrawal scenario were used as input to the regressions to predict withdrawal effects on monthly distribution, and both monthly and annual relative abundance. Prior to regressing distribution and relative abundance against inflow, individual species were sub-divided by size-class, gear type, and collection zone into sub-groups termed "pseudospecies. Approximately half of the fish species in the estuary that were abundant enough to be analyzed exhibited a significant distributional response to changes in freshwater inflow within at least one specific pseudo-species size-class. For each pseudo-species that showed distributional shifts, their center-of-abundance moved upstream as inflows decreased. The greatest predicted shift in a pseudo-species center-of-abundance under the worst-case 1995NN Scenario was < 2.9 km (1.8 mi). Because of relatively small changes in center-of-abundance we conclude that withdrawals are unlikely to force distributional shifts of pseudo-species away from critical stationary habitat components in the downstream estuary.

The relative abundances of fishes in the St. Johns River estuary appears highly sensitive to variations in freshwater inflow. Of 57 fish and invertebrate species abundant enough to be analyzed, 47 (82%) exhibited a significant abundance response to changes in freshwater inflow within at least one size-class. Sixty-one pseudospecies (representing 34 species) having the strongest relative abundance relationships to freshwater inflow were used to quantitatively predict effects of water withdrawals. In general, freshwater pseudospecies relative abundance declined with decreasing inflow and marine pseudospecies relative abundance increased with decreasing inflow. The response of estuarine pseudospecies (either an increase or a decrease in relative abundance with decreasing freshwater inflow) varied between species and sometimes between different size-classes of the same species. Moderate impacts to the estuarine fish community were predicted for all withdrawal scenarios that did not have an augmentation effect. Relative abundance of juvenile (<150 mm [5.9 in] SL) white catfish (*Ameiurus catus*) and channel catfish (*Ictalurus punctatus*) in the estuary responded negatively to withdrawals. Juveniles of these important commercial and recreational species appear to utilize the estuary as overwintering habitat. How predicted withdrawal reduction effects on juvenile abundance in the estuary potentially relate to future adult catfish abundance in the river is unknown. The Half2030PS Scenario had the least affect on the St. Johns River fish community of all the withdrawal scenarios considered.

2 INTRODUCTION

Fishes are an important biological component of the St. Johns River ecosystem. They occupy many important roles in the food web, ranging from being primary consumers of plants and detritus, to secondary consumers of Crustacea and other invertebrates, to top-level carnivores. In turn, fishes themselves are consumed by a myriad of other species. The fish community of the St. Johns River is a productive, diverse composite of freshwater, estuarine, and marine species populations. It is a biologically unique community in North America because several estuarine species have established nonmigratory breeding populations in upstream freshwater reaches. The St. Johns River also supports some of the most valuable commercial and recreational fisheries in the state (Bass and Cox 1985; DeMort 1990; Holder et al. 2006; McLane 1955).

The purpose of this chapter is to evaluate potential effects of proposed surface water withdrawals on St. Johns River fish populations. Water withdrawals have the potential to affect fish populations indirectly—through effects on hydrology, hydrodynamics, water quality, and habitat—or directly through entrainment and impingement. Understanding the potential effects of water withdrawals on fish communities is essential to making informed management decisions.

This evaluation uses historical and recent empirical fisheries data collected from the river, integrated with a review of peer-reviewed scientific literature concerning hydrologic effects on fish. We focused on reviewing studies conducted in Florida or the southeastern United States. Where appropriate and where sufficient data were available, we developed hydroecological models that relate the status of fish population attributes to hydrologic variation and applied those models to assess the effects of water withdrawals. These hydroecological models allowed us to make quantitative predictions of the potential effects of various withdrawal scenarios. Each section of this chapter (i.e., methods, results, and discussion) presents discussion of the freshwater and estuarine reaches of the river separately.

2.1 Background

The St. Johns River contains a wide array of aquatic and floodplain habitats that support diverse fish assemblages (Evermann and Kendall 1900; McLane 1955; Tagatz 1968). Habitats range from broad forested and herbaceous floodplains to narrow and shallow labyrinthine channels, vegetated sloughs and littoral areas, tributaries, wide shallow lakes, and deep channels (DeMort 1990; McGrail et al. 1998). Tidal flows that extend far upriver beyond the estuary and inflow from saline springs enhance floral and faunal diversity (Anderson and Goolsby 1973; Morris 1995; Pyatt 1959).

The waters of the St. Johns River are generally shallow (< 2.0 m [6.6 ft]) and submersed aquatic vegetation (SAV) is common. Some of the most abundant submersed species are eelgrass (*Vallisneria americana*), pondweed (*Potamogeton* spp.), and hydrilla (*Hydrilla verticillata*). Common floating-leaved species include water lilies (*Nymphaea* spp.) and spatterdock (*Nuphar advena*). Shallow emergent marshes, some of which are quite extensive, surround all of the lakes and occur in many deepwater pockets on the floodplain. Common herbaceous emergent species include bulrush (*Scirpus* spp.), spikerush (*Eleocharis* spp.), maidencane (*Panicum* spp.), cattail (*Typha* spp.), pickerelweed (*Pontederia cordata*), and smartweed (*Polygonum* spp.). Woody species commonly found in the shallow marshes include buttonbush (*Cephalanthus occidentalis*) and Carolina willow (*Salix caroliniana*). Species commonly found on the intermittently inundated floodplain include sand cordgrass (*Spartina bakeri*), soft rush (*Juncus effusus*), dollarwort (*Hydrocotyle* spp.), smartweed, maidencane, willow, and cabbage palm (*Sabal palmetto*). Much of the floodplain of the middle river basin supports cattle grazing, and various pasture grasses are common. Extensive areas of the floodplain are frequently burned to maintain the herbaceous structure. Downstream from Lake Monroe, the river floodplain changes from being dominated by herbaceous plants to being dominated by floodplain swamp (DeMort 1990). Dominant tree species in the floodplain swamp include bald cypress (*Taxodium distichum*.), oak (*Quercus* spp.), red maple (*Acer rubrum*), elm (*Ulmus americana*), red bay (*Perea borbonia*), and Carolina ash (*Fraxinus caroliniana*).

Two hundred and twenty-five fish species have been collected from the St. Johns River (Cox et al. 1980; MacDonald et al. 2009; McLane 1955; Tagatz 1968); 63 freshwater species, 138 euryhaline species, and 24 marine species. Euryhaline species use the estuary for some or all of their life stages. Several species considered to be strictly estuarine inhabitants, including stingray (*Dasyatis* spp.), goby (*Microgobius* spp.), and pipefish (*Syngnathus* spp.), have established subpopulations that spend their entire life cycles within the freshwater portions of the river (Burgess and Franz 1978; Johnson and Snelson 1996).

Since the 1850s, the St. Johns River has supported valuable commercial fisheries (Cary 1885; McLane 1955; Moody 1961). From 1948 to 1953, more than 4.5 million kg (10 million lbs) of fish were commercially harvested by haul seines from Lake George alone (Moody 1961). Other commercial gear types used in the river included trap nets, eel pots, gill nets, otter trawls, wire traps, and trotlines. Dominant species harvested included gizzard shad (*Dorosoma cepedianum*), black crappie (*Pomoxis nigromaculatus*), white catfish (*Ameiurus catus*), channel catfish (*Ictalurus punctatus*), American shad (*Alosa sapidissima*), and American eel (*Anguilla rostrata*) (Hale et al. 1995; Hale et al. 1996; McBride 2000; McLane 1955; Moody 1961). Red drum (*Sciaenops ocellatus*), southern kingfish (*Menticirrhus americanus*), black drum (*Pogonias cromis*), and striped mullet (*Mugil cephalus*) are just a few of the saltwater species that were also commercially exploited (Bass and Cox 1985; Hale et al. 1996). Since the 1950s, most commercial fishing operations have been greatly curtailed. Species still harvested commercially in smaller numbers include striped mullet, black drum, menhaden (*Brevoortia* spp.), flounder (*Paralichthyes* spp.), and white catfish (*Ameiurus catus*).

Recreational sport fishing is also extremely popular in the St. Johns River. The most sought-after freshwater species include largemouth bass (*Micropterus salmoides*), black crappie, bluegill (*Lepomis macrochirus*), redear sunfish (*Lepomis microlophus*), and redbreast sunfish (*Lepomis auritus*) (Bass and Cox 1985; Cheek et al. 1984; DeMort 1990; Holder et al. 2006). Popular saltwater or estuarine species include red drum, spotted seatrout (*Cynoscion nebulosus*), black drum, and Atlantic croaker (*Micropogonias undulatus*) (DeMort 1990). Angler survey data summarized by Bass and Cox (1985) indicate that annual angler sport fishing on the entire St. Johns River in 1975 to 1976 exceeded 2,300 man-hours per river km with yields exceeding 2,200 fish per river km. Compared to other Florida rivers, only the Blackwater River in the panhandle yielded more fish per river km.

Even though it continues to support valuable commercial and recreational fisheries, the fish community of the St. Johns River today is impaired compared to the community present 50 years ago. Native striped bass (*Morone saxatilis*) were extirpated, and populations are now maintained by stocking northern and hybrid strains (M. Hale, FWC, pers. comm. 2011)(Holder et al. 2007). Spawning runs of anadromous shad and herring, that for decades supported extensive commercial and recreational fisheries, have dramatically declined (McBride and Holder 2008). The abundance of other commercially exploited species such as American eel and river catfish have also declined. In the 1960s, fisheries biologists began documenting shifts in the freshwater species composition in the river away from communities dominated by desirable sport fishes (e.g., largemouth bass and other sunfishes) toward communities dominated by undesirable species, such as gizzard shad and gar (*Lepisosteus* spp.) (Cox et al. 1976; Moody 1970). By the 1980s, abundance of sport fishes had declined (Bass and Cox 1985). Factors considered responsible for changes in the river fish community include overexploitation, persistent declines in water quality associated with urban and agricultural development, fish kills, channelization, over drainage and loss of basin wetlands, aquatic weed treatments, loss of habitat, introduction of exotics, and increased concentrations of herbicides, pesticides, and heavy metals (Bass and Cox 1985; Cox et al. 1976; DeMort 1990; McBride and Holder 2008; Moody 1970; Nico and Fuller 1999). Although some community changes (e.g., the extirpation of native striped bass, extensive habitat loss to development) are irreversible, others are not. For example, water quality improvements or habitat restoration could result in increased abundance of sport fishes and shift communities toward more economically or socially desirable species. On the other hand, future anthropogenic impacts that further exacerbate community stressors could cause additional declines in the St. Johns River fishery resources.

2.2 Conceptual Models for Freshwater and Estuarine Analyses

We developed conceptual models showing a plausible chain of causation linking surface water withdrawals to the status of fish community attributes such as abundance, growth, spawning success, community structure, and distribution for fishes in freshwater (Figure 2–1) and estuarine reaches (Figure 2–2).

Potential withdrawal effects on fishes in freshwater river reaches (see Figure 2–1) would stem from reduced water levels and flows, reduced floodplain inundation, and entrainment of eggs and larvae (i.e., direct removal from the system). Inputs to the freshwater fisheries analysis from other working groups include the potential loss of SAV (see Chapter 9 Submersed Aquatic Vegetation), potential declines in freshwater benthic macroinvertebrate density and diversity (see Chapter 11 Benthic Macroinvertebrates), potential for increased phytoplankton blooms and resultant declines in dissolved oxygen (DO) (see Chapter 8 Plankton), and potential for DO declines due to biogeochemical processes (see Chapter 7 Biogeochemistry). Outputs from the freshwater fisheries analysis served as inputs to the Floodplain Wildlife Working Group (see Chapter 13 Floodplain Wildlife).

Water withdrawals could potentially affect fishes in estuarine reaches by reducing freshwater inflow and changing the spatial coverage and distribution of salinity zones. These changes can directly influence estuarine fish distribution, abundance, and community structure (see Figure 2–2). Inputs to the estuarine fisheries analysis from other working groups include potential changes in benthic macroinvertebrate communities (see Chapter 11 Benthic Macroinvertebrates), potential loss of SAV (see Chapter 9 Submersed Aquatic Vegetation), and potential for increased phytoplankton blooms and a resultant decline in DO (see Chapter 8 Plankton). As with the freshwater analysis, outputs from the estuarine analysis serve as input to the Floodplain Wildlife Working Group (see Chapter 13 Floodplain Wildlife).

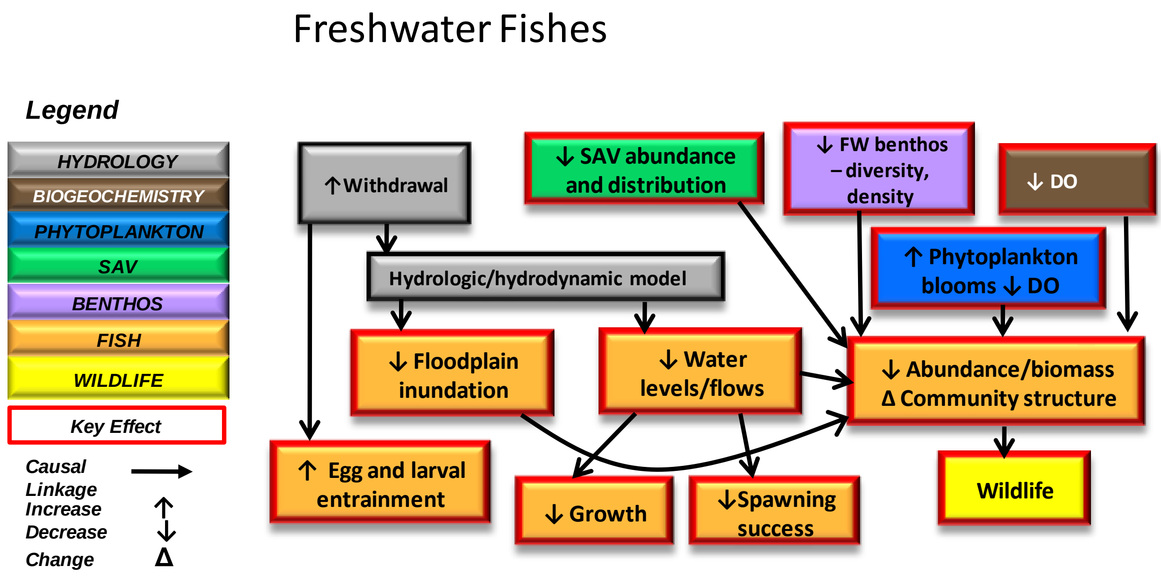

Figure 2–1. Conceptual model of the potential effects of water withdrawals on the freshwater fishes of the St. Johns River.

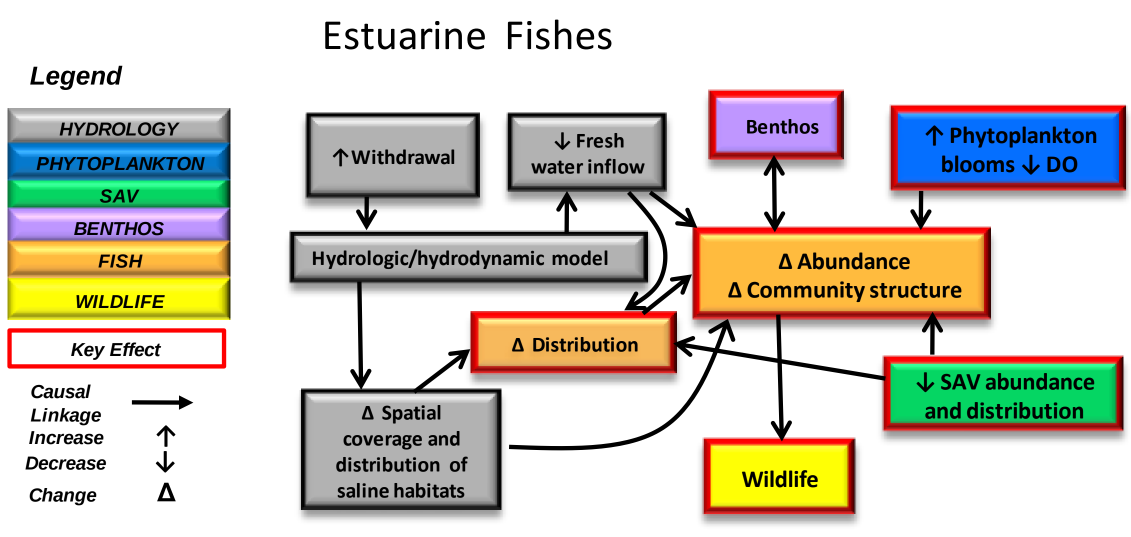

Figure 2–2. Conceptual model of the potential effects of surface water withdrawals on estuarine fishes of the St. Johns River.

3 METHODS

3.1 Analyzing Effects of Water Withdrawals on Fishes in Freshwater Reaches

3.1.1 Modeled Hydrology

Surface water withdrawals can affect many physical, chemical, and biological processes in the river. The severity of these effects depends on the magnitude of the withdrawals relative to the pre-existing flow. The St. Johns River naturally experiences seasonal and episodic variability in both flow and stage. This natural variability leads to significant interannual differences in the spawning success of many species and ultimately to large fluctuations in fish population sizes (Claramunt and Wahl 2000; Houde 1987).

To investigate potential effects of water withdrawals, we compared a number of hydrologic parameters between the baseline scenario and several proposed withdrawal scenarios (Table 3–1). Only data derived from the Hydrologic Simulation Program–FORTRAN (HSPF) model output (see Chapter 3. Watershed Hydrology) were used for comparative analyses. Potential hydrologic effects of withdrawals were analyzed for Lakes Poinsett and Harney and at minimum flows and level (MFL) Transects—Toso528, H1, Lake MonroeT4, Pine Island, and Lake Woodruff (Figure 3–1). For each site except Lake Harney, which lacked ground survey data, we also identified a bankfull elevation, which is defined as the elevation above which water leaves the river channel or lakeshore bank and encroaches on the floodplain. Bankfull elevations were determined from an examination of cross-sectional data and from written descriptions of the MFL transects (Mace 2006; Mace 2007a; Mace 2007b; Mace 2007c). For Lake Poinsett, Toso528, and H1 we analyzed hydrologic data for the entire 34-yr period of record (1975 to 2008) simulated by the HSPF hydrologic model (see Chapter 3. Watershed Hydrology). For all other sites, the Environmental Fluid Dynamics Code (EFDC) hydrodynamic model (see Chapter 6 River Hydrodynamics Results) provided simulated data from 1995 to 2005. In the EFDC hydrodynamic model, effects of sea level rise upstream of Lake Harney were found to be negligible (see Chapter 5 River Hydrodynamics Calibration). Thus, for the fish analysis, for all sites upstream of Lake Harney, the potential effects of the Full1995PN and Full1995PS scenarios are the same and the potential effects of the Half1995PN and Half1995PS scenarios are the same.

Hydrologic parameters compared between simulated model runs included mean water levels and flows, intensity and duration of annual high and low events, seasonality of fluctuations, and water level recession rates. These hydrologic parameters are similar to the critical components of flow and stage proposed by other researchers to regulate ecological processes in aquatic ecosystems (Poff et al. 1997; Richter et al. 1996). The lowest temporal units for both flow and stage used in our analysis were average daily values received as output from the HSPF hydrologic and EFDC hydrodynamic models. Flows were rounded to the nearest hundredth cubic meter per second ($m^3\ s^{-1}$), and water levels were rounded to the nearest centimeter (cm). Mean water levels and flows represent the arithmetic mean of average daily values over the period stipulated, rounded to the nearest cm and $m^3\ s^{-1}$, respectively. For example, monthly mean levels represent the mathematical average of all daily values for that month. Stage duration curves,

Table 3–1. Water withdrawal scenarios used in the fish analysis.

Scenario Name	Land Use	USJRBP Complete	Water Withdrawal	Sea Level Rise	Use in the Fish Analysis
Base1995NN	1995	No	0	0	Baseline condition or scenario
Full1995NN	1995	No	155 mgd (6.8 $m^3 s^{-1}$)	0	An unrealistic worst-case scenario
Full1995PN†	1995	Yes	155 mgd (6.8 $m^3 s^{-1}$)	0	A near-term scenario
Full1995PS†	1995	Yes	155 mgd (6.8 $m^3 s^{-1}$)	+14 cm (5.5 in)	A near-term scenario
Half1995PN†	1995	Yes	77.5 mgd (3.4 $m^3 s^{-1}$)	0	A near-term scenario
Half1995PS†	1995	Yes	77.5 mgd (3.4 $m^3 s^{-1}$)	+14 cm (5.5 in)	A near-term scenario
Full2030PS	2030	Yes	155 mgd (6.8 $m^3 s^{-1}$)	+14 cm (5.5 in)	A long-term scenario
Half2030PS	2030	Yes	77.5 mgd (3.4 $m^3 s^{-1}$)	+14 cm (5.5 in)	A long-term scenario
FwOR1995NN	1995	No	262 mgd (11.5 $m^3 s^{-1}$)	0	An unrealistic worst-case scenario that was used only to evaluate withdrawal effects on salinity in the estuary.
FwOR2030PS	2030	Yes	262 mgd (11.5 $m^3 s^{-1}$)	+14 cm (5.5 in)	A long-term scenario

*See Chapter 6. River Hydrodynamics Results for a detailed discussion of the scenarios.

† For the fish analysis, the Full1995PN and Full1995PS Scenarios are synonymous and the Half1995PN and Half1995PS Scenarios are synonymous for the river upstream of Lake Harney because there the EFDC hydrodynamic model analysis (see Chapter 6 River Hydrodynamics Results) found negligible effects on water levels upstream of the lake from modeled sea level rise.

showing the percent of time that water levels exceeded a given elevation, were also generated for the period of record of each simulation.

Water years were used to quantify intensity and durations of both high (rainy season) and low (dry season) water level and flow conditions (Gordon et al. 1992; Robison 2004). The months assigned to a given water year differed for the two seasons in order to ensure that each annual hydrologic cycle (rainy season versus dry season) was considered in its entirety and was not split between years (Gordon et al. 1992). Because the rainy season in Florida occurs during late summer and fall (July to October), our analyses of high flow and water level conditions use a water year that runs from 1 June to 31 May. As an example, to analyze rainy season conditions in 1976, we would use a water year that started on 1 June 1975 and ended on 31 May 1976. Conversely, because the dry season in Florida occurs during late spring (March to June), the water year used to analyze dry season water levels or flows runs from 1 October to 30 September. Thus, to analyze dry season hydrologic conditions in 1976, we would a water year that started on 1 October 1975 and ended on 30 September 1976.

Maximum flow or stage is defined as the value that is exceeded continuously for a set number of days. For example, the 30-day high stage equals the maximum stage that was exceeded for at least 30 consecutive days during the rainy season. Minimum flow or stage is defined as the value

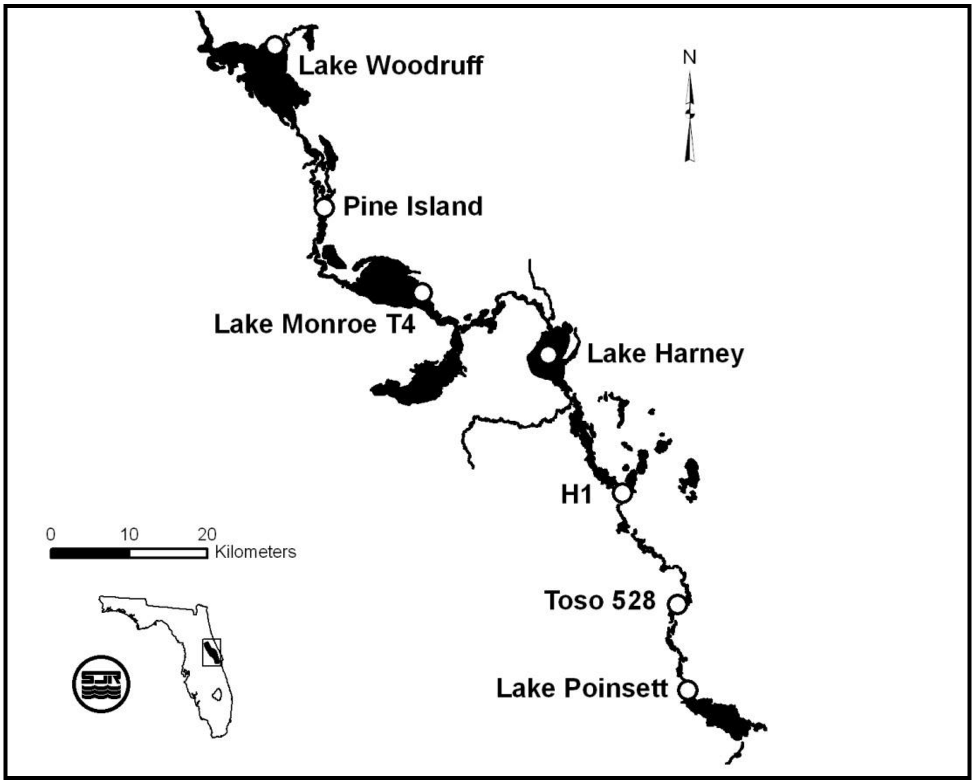

Figure 3–1. Locations of sites used for hydrologic comparisons of surface water withdrawal scenarios, including MFL Transects Toso528, H1, Lake Monroe T4, Pine Island, and Lake Woodruff.

that was not continuously exceeded (i.e., the stage which levels stay below) for a set number of days. For example, the 30-day low stage equals the minimum stage that was not exceeded for at least 30 consecutive days during the dry season. To illustrate, let's assume we want to know for each year of a modeled scenario what was the highest ground elevation in the floodplain that was flooded continuously for at least 1 month (30 days) continuously. To determine this we would use each wet season (1 June to 31 May) water year of the model output. For each year, there are 326 continuous (overlapping) 30-day periods. The first 29 days of the water year have no values (365 – 29 = 326). The lowest value for each of the 30-day periods would be recorded. The highest of those 326 recorded values gives us the highest continuous 30-day water level or, the highest ground elevation that was flooded for at least 30 continuous days, during each water year (Robison 2004). On the other hand, to determine the lowest ground elevation that was continuously dry for at least 1 month, we would use each dry season (1 October to 31 September) water year of the model output. Again, for each water year there are 326 continuous (overlapping) 30-day low values. The highest value for each of the 30-day periods would be recorded. The lowest value of those 326 recorded values then gives us the lowest continuous 30-day water level, or the lowest ground elevation that was exposed for 30 continuous days, for each of the water years (Robison 2004). Possible effects of withdrawals on the seasonality of fluctuations in flow and stage were evaluated using monthly means.

Rates of water level change, particularly the rate at which water levels fall (recession) may be ecologically important, especially during the spring dry season (January to May) when a large number of fish species are spawning and water levels fall almost every day due to lack of rain. Recession rates during the rainy season are less likely to be as important. To evaluate possible effects of withdrawals on water level recession rates, we compared monthly daily averages between scenarios. To calculate a mean monthly daily recession rate, we first calculated a recession rate for each day of a given month by subtracting consecutive daily stage values. Then, only those values that declined from one day to the next were averaged to calculate a monthly mean. Monthly means for each year of model output were then averaged. Because recession is a rate of change, it is not related directly to any particular water level.

3.1.2 Freshwater Species and assemblages

Water withdrawals have the potential to directly, or indirectly, affect reproduction, growth, and/or mortality of fishes. Depending on life history characteristics, withdrawal effects may be species specific. To facilitate our analysis of potential water withdrawal effects within the freshwater reaches of the river, we first created a list of 81 species representing 27 families that may be present in the freshwater portions of the middle and upper basins (Appendix 12.A). This list was derived primarily from McLane (1955) and a review of numerous FWC reports that document fishery work conducted on the St. Johns River from 1958 to 2008 (see Literature Cited). Further review revealed that 20 of the 81 species are present only in tributaries or water bodies not directly connected to the river. These species were excluded from further analysis. In addition, nine species that are mainly estuarine, coastal, or oceanic spawners, and are abundant in the LSJRB, were not considered here but were deferred to the analysis of withdrawal effects on estuarine fishes. Finally, five exotic species also were excluded. This left 47 species that were included in the freshwater analysis (Appendix 12.A).

To further facilitate the analyses of potential water withdrawal effects, the 47 freshwater species were subdivided into five general assemblages (Table 3–2):

- Open Water/Riverine Large Fishes Assemblage,
- Open Water Small Forage Fishes Assemblage,
- Large Sunfishes Assemblage,
- Marsh and Floodplain Large Fishes Assemblage,
- Littoral Zone, Marsh, and Floodplain Small Fishes Assemblage.

Species were assigned to assemblages based on habitat preference, size, and reproductive strategy. This information was derived primarily from species accounts given in McLane (1955), data provided in Carlander (1969a; 1969b), a review of studies conducted by FWC (Bass and Cox 1985; Cox et al. 1981; Cox et al. 1976; Eisenhauer et al. 1993), and personal experience of the authors. A review of important life history characteristics of species within each assemblage is given in the results section. Potential water withdrawal effects were analyzed for each of the assemblages based on the results of the hydrologic data analysis described above. Models relating fish response within the individual assemblages to hydrologic change resulting from water withdrawals were developed only for those withdrawal scenarios where an effect was deemed likely.

Table 3–2. Assemblages of freshwater fish species found within the St. Johns River. (See Appendix 12.A for freshwater species reported within basin boundaries but not considered by these categories.)

Assemblage	Species	Description
Open Water/ Riverine Large Fishes	Channel catfish (*Ictalurus punctatus* , white catfish (*Ameiurus catus)*, longnose gar (*Lepisosteus osseus*, gizzard shad (*Dorosoma cepedianum*), snail bullhead (*Ameiurus brunneus*), American shad (*Alosa sapidissima*), hickory shad (*Alosa mediocris*), blueback herring (*Alosa aestivalis*)	These species are most common in open water habitats. Catfishes and longnose gar are most abundant in the river downstream of Lake Monroe. Flowing tributaries are important. The shad are all anadromous. American shad spawn in the main river channel between Lakes Monroe and Poinsett. Hickory shad and blueback herring spawn in lower river reaches.
Open Water Small Forage Fishes	Threadfin shad (*Dorosoma petense*), tidewater silversides (*Menidia beryllina)*, coastal shiner (*Notropis petersoni*), taillight shiner (*Notropis maculatus*), Seminole killifish (*Fundulus seminolis)*, golden shiner (*Notemigonus chrysoleucas*), pugnose minnow (*Opsopoeodus emiliae*), brook silverside (*Labidesthes sicculus)*	Assemblage consists of small species (< 200 mm [8 in] TL) whose adults use mainly offshore open water habitats. All are obligatory plant spawners that have adhesive eggs. Vegetated littoral areas are important for reproduction. There is no parental care of the young.
Large Sunfishes	Largemouth bass (*Micropterus salmoides)*, bluegill (*Lepomis macrochirus)*, redear sunfish (*Lepomis microlophus)*, black crappie (*Pomoxis nigromaculatus)*, redbreast sunfish (*Lepomis auritus)*, spotted sunfish (*Lepomis punctatus)*	Adults use open water and littoral habitats. Adults utilize the floodplain, but are uncommon >100 m (109 yd) from open water. YOY and juveniles more likely found on floodplain. SAV plays an important role in predator-prey dynamics. All are nest builders who provide parental care. Sunfishes are important recreational species. Most species are intolerant of low DO (< 5 ppm). Redbreast sunfish prefer flowing habitats.
Marsh and Floodplain Large Fishes	Chain pickerel (*Esox niger)*, Florida gar (*Lepisosteus platyrhinchus)*, bowfin (*Amia calva)*, redfin pickerel (*Esox americanus)*, brown bullhead (*Ameiurus nebulosus)*, yellow bullhead (*Ameiurus natalis)*, warmouth (*Lepomis gulosus)*, lake chubsucker (*Erimyzon sucetta*)	Adults and young use the littoral zone, open water pockets in dense vegetation, sloughs, canals, and backwater. These are the dominant large fishes found on the floodplain, although abundance is generally low. They are tolerant of low DO. They spawn in dense vegetation. Most broadcast adhesive eggs with no parental care.
Littoral Zone, Marsh, and Floodplain Small Fishes	Mosquitofish (*Gambusia holbrooki*), rainwater killifish (*Lucania parva)*, least killifish (*Heterandria formosa*), sailfin, molly (*Poecilia latipinna*), flagfish (*Jordanella floridae)*, sheepshead minnow (*Cyprinodon variegates)*, bluefin killifish (*Lucania goodie)*, golden topminnow (*Fundulus chrysotus*), bluespotted sunfish (*Enneacanthus gloriosus)*, dollar sunfish (*Lepomis marginatus)*, Everglades pygmy sunfish (*Elassoma evergladei)*, naked goby (*Gobiosoma bosc)*, clown goby (*Microgobius gulosus)*, speckled madtom (*Noturus leptacanthus)*, marsh killifish (*Fundulus confluentus)*, banded sunfish (*Enneacanthus obesus)*, tadpole madtom (*Noturus gyrinus)*, swamp darter (*Etheostoma fusiforme)*, small juvenile sunfishes (*Lepomis* spp.)	Small fishes (< 8cm TL [3 in]) usually found in association with dense SAV or emergent vegetation. They may be extremely abundant. Many species extensively use the floodplain. Most are tolerant of low DO, have short life spans, mature rapidly, and have protracted breeding seasons. Many species are batch spawners, and can have multiple broods. Most species are tolerant of high water temperatures. Killifishes and gobies are also found in estuarine marshes.

3.1.3 Ichthyoplankton Entrainment

Large numbers of fish worldwide are lost annually to water diversions for power generation, irrigation, and industrial and domestic use (Boreman 1977; EPA 2008; Porak and Tranquilli 1981; Post et al. 2006). Such effects include death or injury to aquatic organisms by impingement (being pinned against screens or other parts of water intake structures) or entrainment (being drawn into water systems and subjected to thermal, physical, or chemical stresses). The exposure of fishes to potential effects from water intake structures varies as a result of their life history characteristics in relation to the location of the intake structure.

Many facilities that withdraw water typically place screens in front of their water intakes to prevent juvenile and adult fishes and debris from entering their systems. In systems with high flow rates and large intake sizes, juvenile and larger fishes may be impinged. Current U.S. Environmental Protection Agency (EPA) recommendations are that intake velocities should not exceed 0.15 m s^{-1} (0.5 ft s^{-1}) to minimize adult and juvenile loss (Boreman 1977; EPA 2004). Adult and juvenile impingement is not anticipated to be a significant issue with the proposed water withdrawal facilities on the St. Johns River because design criteria can be established to ensure sufficiently slow water intake velocities and incorporation of intake structures that enable mobile fishes to avoid filtration screens (Gowan et al. 1999; Zeitoun et al. 1981).

Fish eggs and larvae (ichthyoplankton), however, are too small to be trapped on filtration screens and lack the mobility to avoid being entrained into water systems. Fish eggs and larvae entrained by municipal water withdrawals suffer complete mortality because they are directly removed from the natural ecosystem. This entrainment may cause significant larval mortality, reducing recruitment into the adult population (Boreman et al. 1981; EPRI 1999; EPRI 2002; Gallaway et al. 2007; Van Winkle 2000; Van Winkle and Kadvany 2003). Conversely, although very large numbers of eggs and larvae may be lost in some water systems, the relative proportion of the overall production of eggs and larvae affected may be small, and there may be no obvious effect on the population dynamics of the affected species (Gallaway et al. 2007).

The biological and ecological attributes of the reproductive characteristics of individual species determine the vulnerability of their early life history stages to entrainment. Entrainment loss of ichthyoplankton due to water withdrawals from the St. Johns River will depend heavily on (1) location of the intake facilities relative to spawning sites and larval habitats, (2) proportion of spawning sites and larval habitat near withdrawal facilities, (3) temporal and spatial occurrence of eggs and larvae, (4) larval swimming speeds and size at which larvae are no longer vulnerable to entrainment, (5) intake structure design, (6) intake velocities, and (7) drift rates or sweep velocities past the intake structure.

To evaluate the potential direct loss of fish eggs and larvae by entrainment, intensive larval fish monitoring was conducted in the St. Johns River for 21 months (February 2008 through September 2009) at four regional locations. Sampling was conducted upstream and downstream of all proposed water withdrawal intake sites. For specific details of the ichthyoplankton sampling effort and sample processing see Appendix 12.B.

The effects of impingement and entrainment on fish populations have been studied since the issue was first raised in the early 1970s. Methods to estimate the magnitude of effects have continually been refined, and numerous analytical approaches now are employed to answer different questions about potential effects. Data collected by the St. Johns River larval fish monitoring effort will be extrapolated to population level effects using an Empirical Transport Model (ETM). The ETM is widely used by regulatory agencies and industries to estimate the effects of egg and larval entrainment (Boreman and Goodyear 1988; Boreman et al. 1981; Englert and Boreman 1988; EPRI 1999; EPRI 2002). The ETM evaluates the relative proportion of the egg and larval populations that are physically vulnerable to entrainment. The model estimates the potential survival rate of those larvae, calculates year 1 equivalent losses as the potential numbers of year-old fishes lost due to egg and larval entrainment, and expands the results to consider population-level effects. Our ETM analyses will focus on localized effects at the proposed water withdrawal intake sites.

We collected more than 708,000 individual eggs or fish in the ichthyoplankton sampling effort. Raw sample processing, larval identifications, data entry, and quality assurance and quality control procedures were completed in July 2011.The final step before estimating potential entrainment losses will be to calculate relative abundance (# m^{-3}) from flow meter-derived estimates of the amount of water filtered through each net. Due to the tremendous number of fishes collected and the processing time and effort each individual sample required, our analysis of potential entrainment losses on the adult populations was not be completed by the due date for this chapter. Our final analysis will be completed by August 2012. However, information gathered up to September 2011 will be presented and discussed. Given the recent advances in intake design that minimize larval entrainment, results from our larval studies will be most useful during the siting and design stage of any water withdrawal intakes that may be proposed in the future.

3.2 Analyzing Effects of Water Withdrawals on Fishes in Estuarine Reaches

3.2.1 Salinity

Salinity is recognized as a major factor influencing the distribution and abundance of estuarine organisms (Day et al. 1989). To look specifically at potential withdrawal effects on salinity, we used the EFDC hydrodynamic model to calculate the spatial coverage (ha) of five salinity habitat blocks by year during varying periods of maximum average salinity determined from the baseline scenario (Base1995NN) only. Periods of maximum average salinity (30, 60, and 120 day) for the baseline scenario were determined by examining running daily averages of basin-wide salinity. To investigate withdrawal effects, average salinities for each of the modeled scenarios were calculated for the same daily time intervals as in the baseline calculations. We report salinity as parts per thousand (‰) which is basically equivalent to psu (practical salinity units) and the PSS78 (practical salinity scale) units reported by the EFDC hydrodynamic model (see Chapter 5 River Hydrodynamics Calibration). Salinity categories were categorized as limnetic (< 0.5‰), oligohaline (0.5 ‰ to 4.99 ‰), low mesohaline (5.0 ‰ to 11.99 ‰), high mesohaline (12.0 ‰ to 17.99 ‰), polyhaline (18.0 ‰ to 29.99 ‰), and euhaline (≥30.0‰). The total area considered (about 42,850 ha [105,882 ac]), covered that portion of the EFDC hydrodynamic model from the river mouth to Buffalo Bluff upstream of Palatka (Chapter 5. River Hydrodynamics Calibration). In addition to salinity, we also considered how changes in the coverage and distribution of SAV, resulting from increased salinity due to water withdrawals, could affect the fishes that use these habitats.

3.2.2 Estuarine Fish Sampling

Analysis of potential effects of water withdrawals on the estuarine fish community is based on an expansive data set collected in the lower basin estuary monthly from 2001 to 2010 by the FWC Fisheries-Independent Monitoring (FIM) program (FIM; MacDonald et al. 2009). The FIM program is a systematic and continuous fisheries monitoring program whose purpose is to monitor commercial and recreational species and to collect and integrate essential life history information for use in the management and enhancement of fish populations (FWC-FMRI 2001). FIM program monitoring has been going on at various sites since the 1980s, and data from the program have been used extensively to help set minimum freshwater inflow requirements for several Florida estuaries (MacDonald et al. 2007; MacDonald et al. 2006; Matheson Jr. et al. 2004; McMichael and Tsuo 2002).

The FIM uses a stratified random sampling design and a multi-gear approach to collect data on fishes and invertebrates from a wide range of habitats and life history stages (FWC-FMRI 2010). Stratified random sampling is a common approach designed to reduce statistical error by managing for the influence of habitat and other variables on distribution and abundance. Three types of sampling gear are used: (1) 21.3-m (69.9 ft) center-bag seine, (2) 6.1-m (20.0 ft) otter trawl, and (3) a 183-m (600.4 ft) seine. In general, seines document relative abundance of shallow-water shoreline-associated fishes and invertebrates, while the otter trawl documents relative abundance of species associated with deeper water. In association with each sample, the FWC measured salinity, water temperature, depth, DO, and pH and recorded bottom type, presence of SAV, and shore habitat. These data, used in conjunction with catch, are useful for creating a large body of species-specific habitat use information (MacDonald et al. 2009). In developing their sampling protocol, the FWC divided the St. Johns River estuary into 8 FIM sampling zones (MacDonald et al, 2009). It is important to note that the FWC only sampled fishes along the river main stem and its backwaters and did not sample tributaries or intertidal creeks.

The FWC FIM program began sampling the St. Johns River monthly in May 2001. Originally, sampling was conducted in the five FIM sampling zones (Figure 3–2) that extended from the river mouth upstream to river km 85 (river mi. 53). From January 2004 to July 2005, sampling was reduced to zones 1 to 4; in July 2005, sampling expanded to include zones from 1 to 8. In 2009, the St. Johns Water Management District (SJRWMD; District) contracted with FWC to analyze data they had collected to date. Specific objectives were to (1) produce a descriptive database that could serve as a baseline for comparison with future ecological change and (2) investigate relationships between the abundance and distribution of estuarine organisms and variations in freshwater inflow.

We chose to focus on freshwater inflow rather than salinity as a determinant of fish distribution and abundance because most estuarine fishes have relatively wide salinity tolerances, and for many species, salinity tolerance varies with age and size. In addition, fishes are mobile, and salinities at sites where fish are captured can vary widely over a period of hours due to tidal influence. Thus, relationships between fish responses and withdrawal effects on salinity may be weak and difficult to detect. Conversely, studies from around the world have identified freshwater inflow as a major predictive variable influencing estuary-related fishery yields (Day et al. 1989; Drinkwater 1986). Freshwater inflow is also an easily quantifiable variable that will directly respond to water withdrawals.

Although the FIM program was initially developed to assess the recruitment of young-of-the-year (YOY) fishes into the estuaries and investigate relationships between YOY abundance and future adult harvest, the data are also useful for investigating factors (e.g., freshwater inflow) that influence YOY and juvenile abundance.

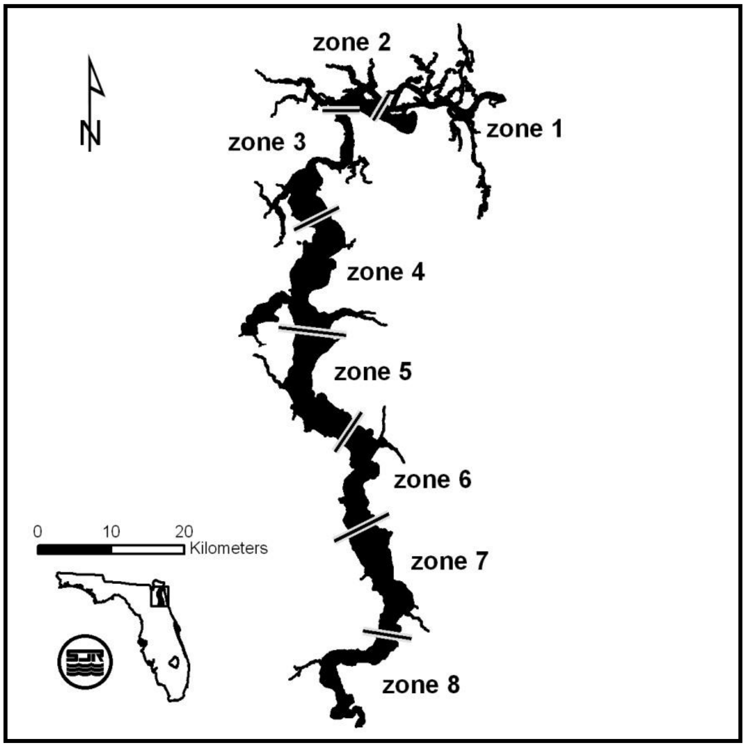

Figure 3–2. FIM zones for sampling the St. Johns River estuary delineated by the FWC Fisheries-Independent Monitoring (FIM) Program.

Species-specific ontogenetic (developmental) changes in habitat, size-specific mortality, and gear avoidance have the potential to confound relationships between inflow and species abundance and distribution. To better control these influences, species were divided into size classes. Monthly abundance and distribution of the various species-specific size classes were then used to determine appropriate months and FIM sampling zones to include in the correlation and regression analyses. Because the two seines and otter trawl sampled different habitats and were selective for different-sized individuals (Hayes 1983), catch could not be combined between gears and correlation and regression analyses had to be conducted for each gear type and species combination individually. For individual species, specific size classes, recruitment period of the size-classes, collection gear, and FIM sampling zones in which the size-classes were collected were separated out into what were termed pseudospecies.

The initial results of the FIM analysis are presented in MacDonald (2009). These results revealed that several pseudospecies exhibited a distributional response to variations in freshwater inflow, whereas more than 60% (27 of 44) of the pseudospecies for which predictive response models could be developed exhibited an abundance response. Applying the predictive models presented in the 2009 FIM report to simulated withdrawal effects on inflow suggested that withdrawals could substantially affect (either positively or negatively depending on the pseudospecies considered) the abundance of some ecologically and economically important estuarine fishes.

During preparation of the Water Supply Impact Study (WSIS), the FIM program collected an additional two years (2009 and 2010) of data. In early 2011, we decided to include this additional data in the final WSIS analyses to increase the number of observations upon which the relationships of inflow to both pseudospecies distribution and abundance were based. As a result, in 2011, the SJRWMD contracted with FWC to conduct updated analyses of the FIM data set. Given the short time frame, the new analyses focused on updating inflow to distribution and abundance relationships and did not update the overall descriptive statistics of the data set. In addition, SJRWMD conducted independent statistical reviews of FIM methods and of the application of FIM results by the SJRWMD to generate potential withdrawal effects (Helsel 2011; Rouhani and Wellington 2011). Recommendations from these reviews were incorporated into the new analyses by both FWC and SJRWMD.

3.2.3 Relationships Between Freshwater Inflows and Estuarine Fish Distribution and Abundance

Potential relationships between freshwater inflow and fish distribution and abundance were examined by correlation and regression analysis. Daily freshwater inflow estimates for the lower St. Johns River were derived from gauged stream flow records at U.S. Geological Survey (USGS) station 2236000 (St. Johns River near DeLand) and USGS station 2243960 (Ocklawaha River at Rodman Dam near Orange Springs, FL). Daily flows from these two sites were summed to approximate total daily inflow into the St. Johns River upstream of the FIM sampling domain. The site at DeLand was chosen to negate the strong tidal fluctuations that confound freshwater flow estimates from gauges located further downstream. It is recognized that freshwater inflow to the river that occurs downstream of the Ocklawaha River could affect these analyses; however, inflow data for most of the tributaries in this reach of the river are not available. In addition, there were strong linear correlations between daily flows at DeLand combined with the Ocklawaha River (1996 to 2005) and daily flows lagged for 30 days (r^2 = 0.87) and 60 days (r^2 = 0.93) at USGS station 08801996, St. Johns River at Jacksonville near the Ortega River (river km 38 [river mi 24]). Daily flows at station 08801996 lagged because this site is approximately 100 km (62 mi) downstream of the Ocklawaha River mouth and 232 km (144 mi) downstream of DeLand. The proposed water withdrawals will have a more direct quantifiable influence on flows at the DeLand and Ocklawaha River sites. Given these strong correlations, proximity to the proposed withdrawal location(s), and the lack of measured inflow data downstream, FWC and the Fish Working Group agreed that flow at DeLand combined with outflow from the Ocklawaha River provides a useful surrogate for total freshwater inflow to the LSJRB estuary.

Relative abundance and distribution summaries were calculated for each gear type deployed by the FIM program in the lower St. John's River. Abundance data were standardized to area sampled prior to analysis (# animals $100m^2$). Distribution at capture was weighted by abundance to determine the central tendency for each species. Only data collected between July 2005 and December 2010 were summarized to ensure that the greatest extent of the river that had been consistently sampled by the FIM program was included. Species with relatively high numbers collected (n > 99) and high percent occurrence (≥ 5%) were considered candidates for species-specific inflow-abundance and inflow-distribution correlation and regression analyses.

Spearman's rank correlation (rho) (Helsel 2011; Helsel and Hirsch 2002) between inflow-abundance and inflow-distribution was assessed for each pseudospecies that had n > 99 animals collected and occurred in ≥ 5% of the samples. Spearman's rho is a nonparametric correlation coefficient, and so is generally applicable as a measure of correlation regardless of data distribution shape or linearity of the relationship. Inflow-abundance correlation was assessed for both monthly and annual recruitment period data, while inflow-distribution correlation was assessed only for monthly data. Inflows were investigated at lag periods of 30, 60, and 90 days (abundance and distribution), and 120, 150, 180, 210, 240, 270, 300, 330, and 360 days (abundance only) prior to the designated sampling period (month or annual recruitment period). Lagged inflow, rather than instantaneous inflow at time of capture, was used in the correlation analyses because survival has been more strongly linked to longer term hydrologic conditions than to inflow at the time of capture (Browder and Moore 1981; Drinkwater 1986; Sutcliffe 1973).

The sampling period of data used in each assessment varied depending on the gear and zones retained for each pseudospecies (Table 3–3). Data collected from the river's main stem were used for distributional responses to inflow, while data from both main stem and backwater areas were included for inflow- abundance responses.

Table 3–3. Fisheries-Independent Monitoring (FIM) program data included in correlation and regression analyses.

<table>
<tr><th rowspan="2">Analysis Periodicity</th><th rowspan="2">FIM Zones</th><th rowspan="2">Sampling Period</th><th colspan="2">Data Included by Dependent Variable Analyzed</th></tr>
<tr><th>Abundance</th><th>Distribution</th></tr>
<tr><td rowspan="2">Monthly
(abundance and distribution)</td><td>1 to 4</td><td>June 2003 to December 2010</td><td rowspan="2">Main stem and backwater areas</td><td rowspan="2">Main stem only</td></tr>
<tr><td>1 to 8</td><td>July 2005 to December 2010</td></tr>
<tr><td>Annual
(abundance only)</td><td>1 to 4</td><td>January 2001 to December 2010</td><td>Main stem and backwater areas</td><td>NA</td></tr>
</table>

NA = not evaluated

Linear regression analyses of dependent variable (abundance or distribution) on lagged inflow were conducted for pseudospecies that had a significant response ($p < 0.05$) and a relatively high Spearman's rho (absolute value ≥0.4). For pseudospecies with absolute values of rho < 0.4, inflow typically accounted for generally < 16% of the variability in the predicted response. Where more than one inflow lag period met the Spearman's rho screening criteria, the inflow with the maximum absolute value of rho was retained for regression analysis. Additionally, pseudospecies with a rho ≥ 0.4 that were within 3% of the maximum absolute value were also retained because they represent equally valid models given the variation in the data (D. Helsel, Practical Stats, pers. comm. 2011).

Four linear regressions corresponding to different transformation scenarios (no transformation, dependent variable transformed, independent variable transformed, and both dependent and independent variables transformed) were conducted on each retained pseudospecies lagged-inflow period combination. In all cases, the transformation applied was a fourth-root transformation (Helsel 2011). This transformation makes right-skewed data more normally distributed, with the advantage over the log transformation that zeros can be directly included without adding an arbitrary constant.

The predicted residual error sum of squares (PRESS) measures the ability of an equation to predict Y values for observations not used in building a regression model. PRESS is a cross validation tool. The PRESS coefficient of determination r^2 was calculated for each linear regression model (Helsel 2011; Helsel and Hirsch 2002). PRESS is numerically computed as if one observation were excluded, the regression computed, and then a prediction made for that excluded observation. The difference between the prediction and the actual observed value is the prediction error. PRESS sums the squares of these prediction errors over all observations in the data set. A PRESS r^2 was calculated by treating the PRESS as the error sum of squares in the calculation of r^2 using the full model's total sum of squares (TSS): PRESS r^2 = (1 - PRESS/TSS). The PRESS r^2 will be considerably lower than r^2 if a regression is overly influenced by a few observations, indicating that r^2 was overly optimistic in what could be achieved for predicting new values of Y using the regression model. Models with the highest PRESS r^2 are not overly influenced by a few observations and best predict values for new observations. The pseudospecies inflow combination and transformation regression with a significant slope ($p < 0.05$), the highest adjusted PRESS r^2, and an $r^2 \geq 0.25$ were used for predicting water withdrawal effects. Linear regressions with $r^2 < 0.25$ were considered weak, even if they were significant ($p < 0.05$).

The regression models that best fit the dependent variable (abundance or distribution) to inflows were evaluated for potential water withdrawal effects. The inflow variables in this analysis were the modeled freshwater inflow (output from the EFDC hydrodynamic model with DeLand and the Ocklawaha River flows combined) for each of the withdrawal scenarios. The modeled inflow data were lagged as appropriate for each biologically relevant pseudospecies regression model. Time periods where the modeled inflows were outside the range of inflows sampled by FIM were excluded from the analysis. The FIM-based regression equations were applied to estimate differences in abundance and distribution for each pseudospecies under the evaluated water withdrawal scenarios. All data calculations were conducted using the Statistical Analysis System (SAS®) software.

4 RESULTS

4.1 Effects of Water Withdrawals on Fishes in Freshwater Reaches

4.1.1 Effects on Hydrology

Water levels and flows were strongly correlated at all sites, independent of modeled scenarios. Because water levels, rather than flow, will be the most important hydrologic variable influencing the majority of the freshwater species we considered, we only report water levels in our hydrologic comparison of scenarios. Flows are considered later in evaluations of those assemblages or species where direct responses to a change in flow can be quantified.

Water Levels

The water level reduction effect due to water withdrawals between the unrealistic worst-case Full1995NN Scenario and the baseline scenario (Base1995NN) was greatest between Lakes Poinsett and Harney (Figure 4–1a). Between Lake Poinsett and Lake Harney, modeled average water levels declined by 4 to 6 cm (1.6 to 2.4 in), average continuous 30-day low water levels declined by 2 to 3 cm (0.8 to 1.2 in), and average continuous 30-day continuous high water levels declined by 4 to 5 cm (1.6 to 2.0 in). From Lake Harney downstream, the effects of water withdrawals under Full1995NN decrease. At Lake Woodruff, declines in the modeled average continuous 30-day low and continuous 30-day high due to withdrawals were all 1 cm (0.4 in) or less. Inundation frequency curves comparing the Base1995NN and Full1995NN Scenarios relative to bankfull conditions at each of the locations considered are presented in Figure 4–2a.

According to the hydrologic model, low-flow augmentation from a completed USJRBP (Base1995PN) would substantially increase average water levels and average continuous 30-day low water levels over Base1995NN (Figure 4–1b). Average continuous 30-day high water levels would remain unchanged. Flow augmentation from the USJRBP was great enough to offset full withdrawal effects (Full1995PS) on average water levels and average continuous 30-day low water levels compared to Base1995NN; however, continuous 30-day high water levels under Full1995PS were still as much as 4 cm (1.6 in) lower than under Base1995NN (Figure 4–1c). Inundation frequency curves for the two scenarios also show this high water reduction effect (Figure 4–2b). Full withdrawals in conjunction with the completed USJRBP and 2030 land use changes (Full2030PS) had barely discernible reduction effects, (<1cm [0.4 in]) (Figure 4–1d and Figure 4–2c) only on the 30-day continuous highs at Lake Poinsett. For the most part water levels in Lake Poinsett as well as at all other sites, especially with regard to the 30-day continuous lows, substantially increased under Full2030PS. Although the data are not presented here, half withdrawal scenarios (Half1995NN and Half1995PS) caused modeled water level declines to be approximately half what they are under their respective full withdrawal scenarios at all sites. Under the Half2030PS Scenario, water levels increased further over Base1995NN than those observed under Full2030PS.

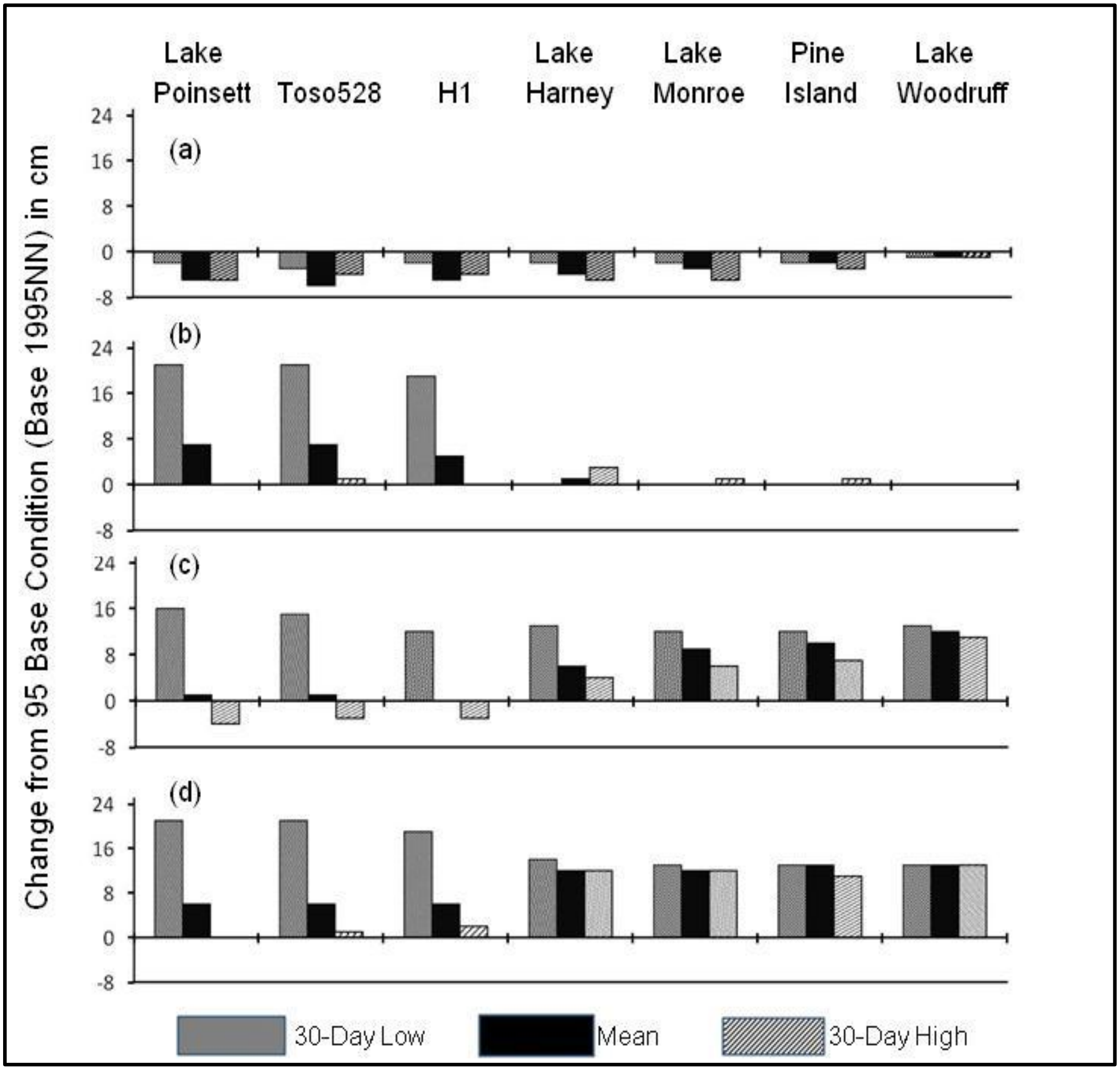

Figure 4–1. Difference in average annual continuous 30-day low, mean, and 30-day high water levels (rounded to the nearest cm) between the baseline scenario (Base1995NN) and (a) Full1995NN, (b) Base1995PN, (c) Full1995PS, and (d) Full2030PS Scenarios at sites along the St. Johns River for the period 1975 to 2008. Sites are arranged from upstream to downstream (see Figure 3–1). Sea level rise did not influence water levels along the Lake Poinsett, Toso528, and H1 Transects (see Table 3–1 Chapter 3 Watershed Hydrology). Howver, sea level rise did increase water levels along the Lake Harney, Lake Monroe, Pine Island and Lake Woodruff Transects (see Chapter 6 River Hydrodynamics Results).

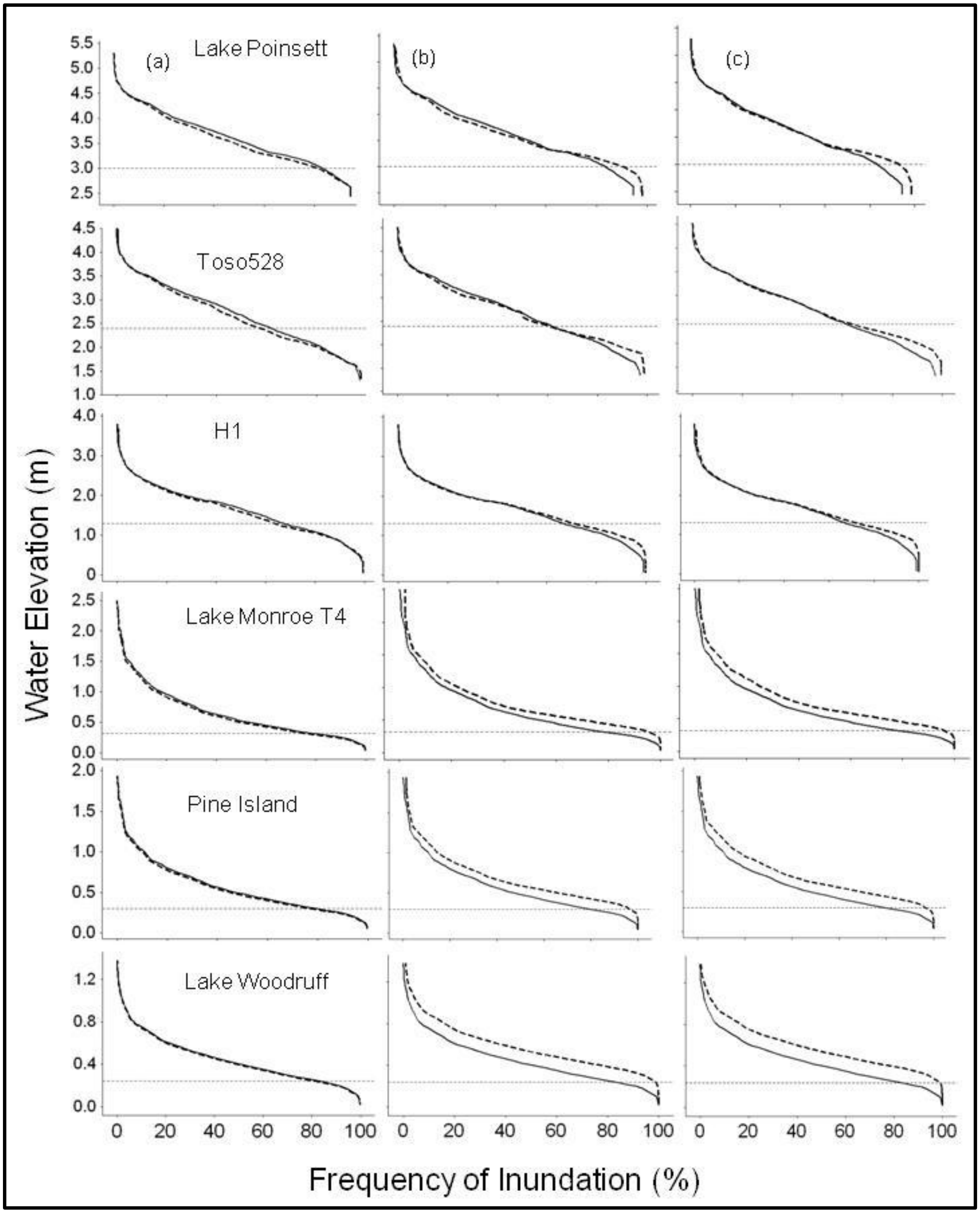

Figure 4–2. Inundation frequency curves for six St. Johns River MFL transects comparing the Base1995NN Scenario (solid lines) to the (a) Full1995NN, (b) Full1995PS, and (c) Full2030PS Scenarios (dashed lines) for the period 1975 to 2008. The dashed horizontal line depicts the bankfull elevation along each transect where water begins entering the floodplain.

The modeled hydrologic output indicates that without the projected increased inflow associated with land-use changes from 1995 to 2030, full withdrawals would decrease the average annual inundation of the floodplain upstream of Lake Harney even after completion of the USJRBP. This condition (Full1995PS) is a potential near-term scenario. Between Lake Poinsett and Lake Harney, the average continuous 30-day high levels are projected to decline 3 to 4 cm (1.2 to 1.6 in), which could affect the fish assemblages that use floodplain habitat in this reach. However, because projected mean and continuous 30-day low water levels would remain unchanged or increase under the Full1995PS Scenario, fish using the more permanently flooded habitats between Lake Poinsett and Lake Harney would experience less severe drying conditions than under Base1995NN. The increases in low, mean, and high water levels from Lake Harney downstream under Full1995PS reflect sea-level rise and would likely benefit freshwater fishes. With projected 2030 land use changes and completion of the USJRBP (Full2030PS, a potential long-term scenario), water levels are predicted to be higher along the entire river; consequently there would likely be no negative effects to freshwater fishes due to full withdrawals under this scenario.

Seasonality of Water Level Fluctuations

Seasonality of water level fluctuations, particularly the timing and duration of the flood pulse, is important to the reproduction and recruitment of many riverine fish species found in tropical and subtropical freshwater ecosystems (Bayley 1995; Junk et al. 1989). To investigate the potential effects of water withdrawals on the seasonality of water level extremes, we used Lake Poinsett as a reference site because it exhibited some of the greatest reduction effects due to modeled withdrawals. In both the baseline Base1995NN Scenario and the worst-case Full1995NN Scenario, low water levels for the year occurred between April and June in 59% of the years and the monthly frequency of occurrence of the 1-day lows between the two scenario was virtually identical (Figure 4–3a). Under the Full1995PS and Full2030PS Scenarios there was a slight shift in the occurrence of the 1-day lows to either earlier or later in the year than under the Base1995NN Scenario, although 50% of the lows still occurred between April and June (Figure 4–3a). The timing of the high water levels of the year were unaffected by withdrawals (Figure 4–3b). Under the Full 1995PS and Full20230PS Scenarios, however, high water levels for the year occurred slightly more often during March and April. Full withdrawals, according to the HSPF hydrologic model, will have negligible effects on the seasonality of water level fluctuations experienced by St. Johns River fishes. Differences between the Base1995NN and the Full1995PS and Full2030PS Scenarios are attributable solely to operation of the USJRBP for flood control and low flow augmentation and 2030 land-use changes on basin runoff characteristics.

Water Level Recession Rates

Modeled average monthly recession rates of the Base1995NN Scenario ranged between -0.8 to -1.4 cm d^{-1} (-0.3 to -0.6 in d^{-1}) at Lake Poinsett and -1.4 to -1.7 cm d^{-1} (-0.6 to -0.7 in d^{-1}) at MFL Transect H1 (Figure 4–4). Recession rates were highest at all sites during the rainy season because recession following flood peaks is higher than recession when water levels are low. Addition of full water withdrawals (Full1995NN) did not appreciably increase average recession rates at either Lake Poinsett or downstream at MFL Transect H1 (Figure 4–4). At MFL Transect H1, the maximum increase in recession rates due to withdrawals occurred in June. If this increase in recession rate of -0.26 cm d^{-1} (-0.1 in d^{-1}) occurred for the entire month, the end of the month water level would fall below the baseline condition by only 6 cm (2.4 in). Results shown in Figure 4–4 are for the worst-case Full1995NN Scenario. Water withdrawals under scenarios with a completed USJRBP and 2030 land-use changes had lesser effects on water recession rates.

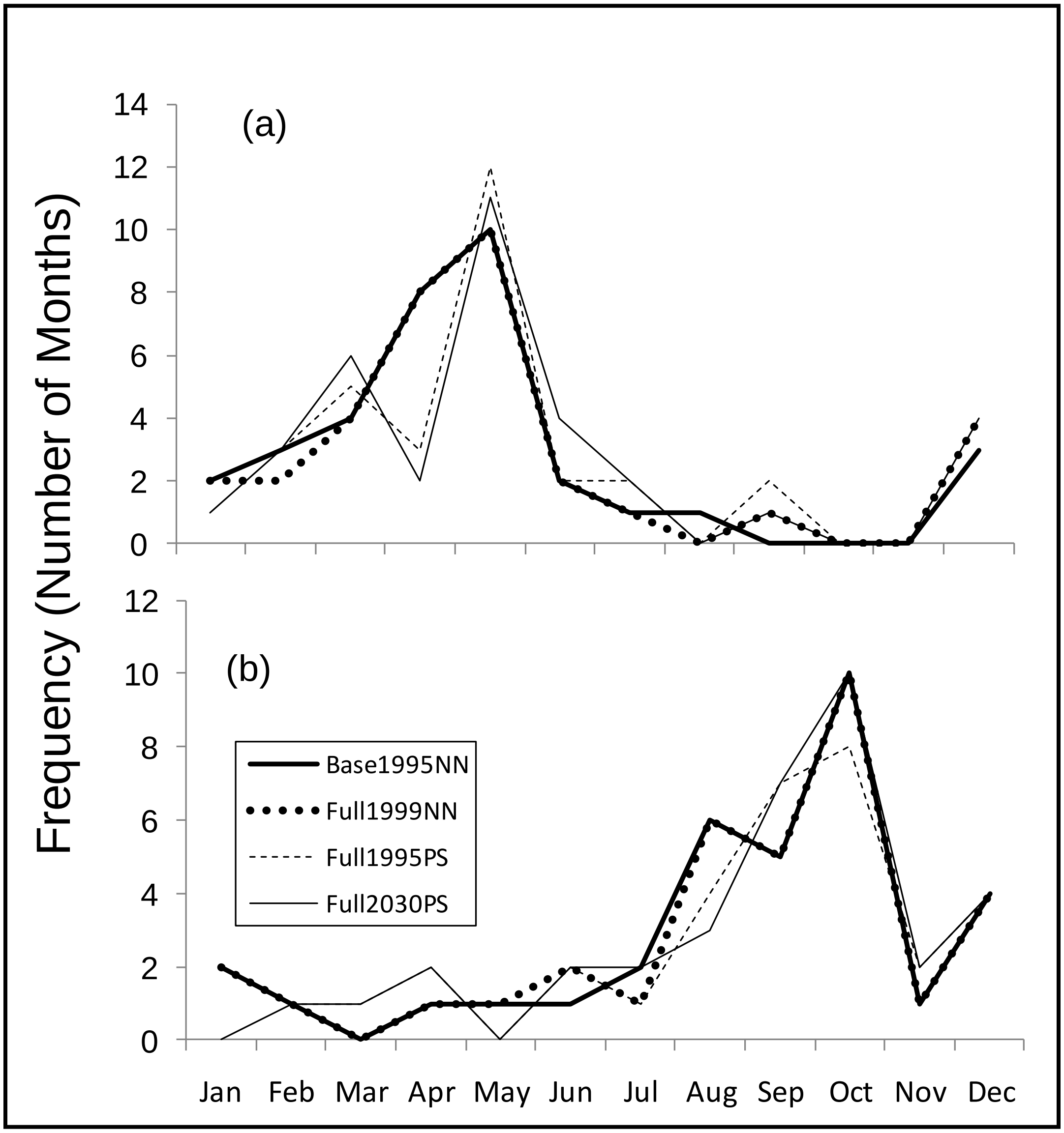

Figure 4–3. Frequency of occurrence of (a) 1-day low and (b) 1-day high water levels by month for the Base1995NN, Full1995NN, Full1995PS, and Full2030PS Scenarios at Lake Poinsett for the period 1975 to 2008.

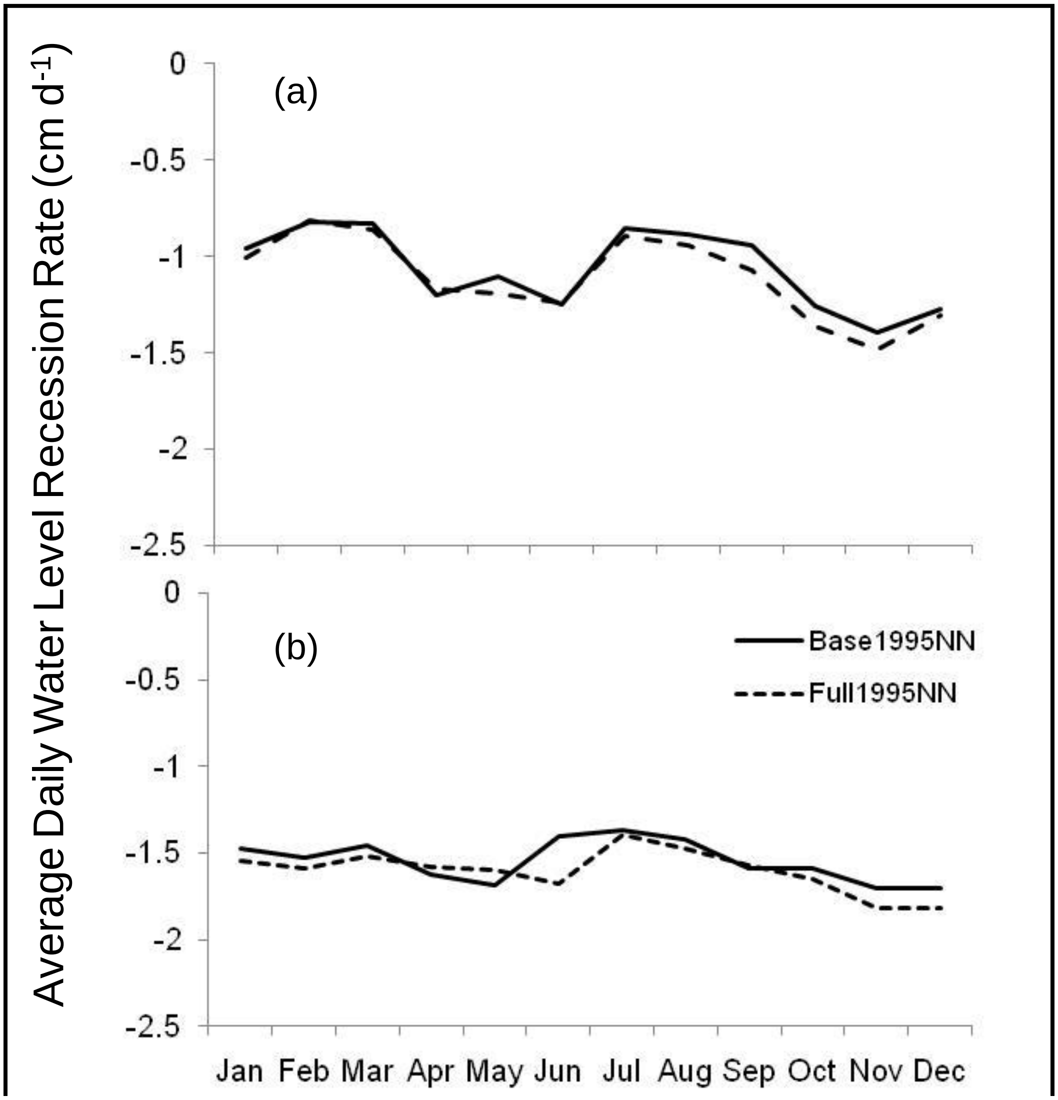

Figure 4–4. A comparison of average monthly recession rates (cm d^{-1}) between the Base1995NN) and Full1995NN Scenarios at (a) Lake Poinsett and (b) MFL Transect H1.

4.1.2 Effects on Freshwater Fish Assemblages

Open Water/Riverine Large Fishes Assemblage

This assemblage is composed of species that mostly occupy the main river channel and the open water areas of the lakes (see Table 3–2, Appendix 12.C). Longnose gar are most abundant in riverine reaches downstream of Lake Monroe where the effects of the withdrawals are small (McLane 1955). Channel and white catfish are found all along the river but appear most abundant between Palatka and Lake Harney (Hale et al. 1986). Channel catfish prefer deeper flowing habitats with structurally complex cover, such as logs and woody debris (McMahon and Terrell 1986), features that are characteristic of the river downstream of Lake Monroe. White catfish share similar habitat preferences as channel catfish (McLane 1955), but are less likely to be found in swifter flowing water (Marcy et al. 2005). Snail bullhead prefer deeper habitats with substantial flow (Marcy et al. 2005). Effects of water withdrawals on all these species in the main river might be difficult to detect based on the small predicted declines in water levels and flows in their preferred habitats, even under the worst-case withdrawal scenario (Full1995NN).

Herring and shad are members of the family Clupeidae and are some of the most abundant species in the St. Johns River. Gizzard shad are likely the most abundant of this group (Cox et al. 1980; McLane 1955). The District currently supports harvesting gizzard shad in several St. Johns River basin lakes to improve water quality and restore more balanced fish communities. Gizzard shad have high fecundity (Bodola 1965; Carlander 1969a) and broadcast adhesive eggs that sink and adhere to rocks and SAV in open water generally less than 2 m (6.6 ft) deep (Miller 1960). Surface water withdrawals will likely have negligible effects on gizzard shad because of their preference for open water habitat, where withdrawal effects will be minimal, and because of the extent of the availability of suitable spawning habitat.

Of special consideration in this assemblage are the three species of anadromous river herring that use the St. Johns River as spawning habitat: American shad, blueback herring, and hickory shad (Appendix 12.C). Of these three species, American shad are the most abundant and intensively studied (McBride 2000; McBride and Holder 2008; Williams and Bruger 1972; Williams et al. 1975). Prior to the 1970s, American shad supported large-scale commercial and recreational fisheries in the St. Johns River. Since then, however, American shad abundance in the river has declined sharply. Today, despite the low numbers of adults entering the river, American shad remain an important sport fish (McBride and Holder 2008).

All anadromous river herring stocks in the United States are currently managed under stock recovery plans overseen by the Atlantic States Marine Fisheries Commission (ASMFC) and restoration efforts are now underway to rebuild populations in many rivers (ASMFC 1999; ASFMC 2009a; ASFMC 2009b). Under current ASMFC plans, all states are required to carefully scrutinize water withdrawal projects and develop management plans that ensure protective flows and levels are maintained and potential entrainment and impingement effects are minimized (ASFMC 2009a; ASMFC 2009b). River herring stocks could be adversely affected by flow reductions if they were great enough to reduce the acreage of suitable spawning habitat, alter critical flow regimes, or block migrations either to or from spawning grounds. After an extensive evaluation of these potential effects (Appendix 12.C), we believe that water withdrawals are not likely to adversely affect any of these parameters. Possible entrainment issues are discussed later in Section 4.1.3 Ichthyoplankton Entrainment.

Open Water Small Forage Fishes Assemblage

All members of this assemblage (see Table 3–2) are small as adults (generally <200mm [8in] total length [TL]) and use open water habitats or habitats adjacent to open water (Loftus and Kushlan 1987; Marcy et al. 2005; McLane 1955). Many are schooling species. Feeding guilds range from planktivores (e.g., threadfin shad) to herbivores (e.g., Seminole killifish) and true omnivores (e.g., golden shiner) (Appendix 12.A). Another common link between the members of this assemblage, besides an adult preference for open water, is that they are all considered phytophils (Simon 1998), or obligatory plant spawners. They broadcast adhesive eggs in vegetated littoral areas with no further parental care. Species composition, density, and spatial coverage of SAV is one component that may influence variability in the population dynamics of these species (Bettoli et al. 1993) .

Other factors influencing population dynamics of the Open Water Small Forage Fishes Assemblage include biotic factors (e.g., such as food availability and predation) and abiotic factors (e.g., such as water quality). Input from other working groups suggests that water withdrawals will likely have a negligible effect on this assemblage. According to the SAV Working Group (see Chapter 9 Submersed Aquatic Vegetation), no appreciable effects to SAV would occur with water withdrawals in potential near- or long-term scenarios. The Biogeochemistry, Plankton, and Benthic Macroinvertebrates Working Groups reported similar conclusions (see Chapter 7 Biogeochemistry, Chapter 8 Plankton, and Chapter 11 Benthic Macroinvertebrates). Effects of water withdrawals on predation pressure on this assemblage are nearly impossible to predict given the numerous factors that influence both predator and prey abundance. Lower water levels could concentrate members of this assemblage and move predators from the vegetated littoral edge into open water, thus increasing predation. However, due to increased inflow resulting from the completion of the USJRB projects and 2030 land-use changes, dry season water levels are predicted to be higher under Full1995PS and Full2030PS than under Base1995NN (Figure 4–1). Because of higher dry season water levels, it is unlikely neither the Full1995PS or Full2030PS Scenarios will cause any detectable change from the baseline condition in either the species composition or abundance of this assemblage.

Large Sunfishes Assemblage

Members of this assemblage (see Table 3–2) constitute some of the most recreationally valuable freshwater fish in the St. Johns River (Appendix 12.D) (Bass and Cox 1985). All are members of the family Centrarchidae (sunfishes), and all build nests and exhibit varying levels of parental care of their eggs and young (Marcy et al. 2005; Warren Jr. 2009). Juvenile and adult sunfishes are generally both invertivores and piscivores (Appendix 12.A). The sunfishes placed in this assemblage generally occupy open water and littoral habitats. Although commonly the dominant top-level predator in warm water communities, sunfish also provide forage for many other species and serve as hosts for sensitive life stages of many freshwater mussels (Warren Jr. 2009). The presence and density of emergent and submersed aquatic macrophytes (plants) can greatly influence the abundance, growth, and distribution of members of this assemblage (Bettoli et al. 1993; Hoyer and Canfield 1996; Loftus and Kushlan 1987; Sammons et al. 2005; Spotte 2007; Ware and Gasaway 1976). Abundant aquatic vegetation can influence population dynamics to such an extent that the effects of relatively modest changes in water level on populations may be undetectable (Bonvechio and Allen 2005). Members of this assemblage exhibit only limited use of seasonally flooded habitats in the St. Johns River. Seasonal hypoxia may play an important role in regulating floodplain use (see Appendix 12.D).

Members of the Large Sunfishes Assemblage nest over a wide range of depths but generally in waters between 0.3 m (1.0 ft) and 3.0 m (10.0 ft). Rapid water level recession rates could cause nest abandonment by the adults or result in exposure and desiccation or stranding of eggs and young fish (Ploskey 1986; Von Geldern Jr. 1971). A comparison of modeled water level recession rates indicate that even under the worst-case scenario (Ful1995NN), changes in recession rates during the peak of the spring spawning season (June) from the baseline condition (Base1995NN) would be negligible (-0.26 cm d^{-1} [-0.1 in d^{-1}])(Figure 4–4). Increased recession rates due to water withdrawals would only drop water levels an additional 6 cm (3.0 in.) over 30 days. This is well below recession rates that would cause nest abandonment by largemouth bass (Von Geldern Jr. 1971)(see Appendix 12.D).

Bonvechio and Allen (2005) investigated relationships between annual and seasonal hydrologic variables and year-class strength of largemouth bass, bluegill, redear sunfish, redbreast sunfish, and black crappie in four rivers and four lakes in Florida. Their results suggest that only redbreast sunfish, a species that prefers flowing water (McLane 1955), may experience a predictable withdrawal effect. Redbreast sunfish may experience a localized reduction in recruitment in the river between Lakes Poinsett and Harney under the Full1995PS and Full2030PS scenarios because of a reduction in fall discharge compared to Base1995NN (Appendix 12.D). This dampening of reproduction however, may be offset to some degree by increased growth due to increased spring discharges (Appendix 12D) (Bonvechio and Allen 2005). Any effects on redbreast sunfish from water withdrawals would likely be difficult to measure. Although redbreast sunfish are common in the upper sections of the river, they are most abundant in the northern (downstream) half and in the Ocklawaha River (McLane 1955). Given this distribution, and the fact that withdrawal effects decrease in a downstream fashion, overall effects of water withdrawals under potential near- and long-term scenarios on redbreast sunfish in the entire river drainage are likely negligible.

A more detailed discussion of potential withdrawal effects on all the other members of this assemblage is presented in Appendix 12.D. Due to model predictions of increased spring discharges and water levels under the Full1995PS and Full2030PS Scenarios as compared to Base1995NN, we believe that water withdrawals will likely have a negligible effect on largemouth bass, bluegill, redear sunfish, black crappie, and spotted sunfish (Appendix 12.D). Increased spring discharges and water levels may actually enhance recruitment of all of these sunfish species (Bonvechio and Allen 2005). In addition, the close association between the abundance of these species and SAV would likely make any abundance responses of these species to small flow or water level changes extremely difficult to detect.

Marsh and Floodplain Large Fishes Assemblage

Adult and juvenile members of this assemblage use the littoral zone, open water pockets in dense vegetation, sloughs, canals, and backwater areas (Table 3–2) (Chick et al. 2004; Loftus and Kushlan 1987; McLane 1955). They are classified in this assemblage because they are commonly the dominant large fishes (>8 cm [3.1 in] standard length [SL]) found in dense marsh and floodplain habitats (Chick et al. 2004; Herke and Horel 1958). Members of this assemblage are well adapted for surviving widely fluctuating water level conditions because they are extremely tolerant of low DO and high water temperatures (Loftus and Kushlan 1987). Most are invertivores or carnivores, and all spawn in association with dense vegetation (Appendix 12.A) (Carlander 1969a; Marcy et al. 2005).

Members of this assemblage are common and often abundant in vegetated littoral zones of lake and riverine habitats of the St. Johns River (Cox et al. 1980; Cox et al. 1977; Cox et al. 1976; Eisenhauer et al. 1993; McDaniel and Cox 1993). Virtually no data are available to look at potential effects of water level fluctuations on abundance of these species. They have little economic or recreational value, and they are commonly characterized in most FWC surveys as rough fish. Consequently, little research has been conducted to investigate potential relationships between biotic or abiotic factors and reproduction, growth, or abundance of these species. Because they are abundant in littoral habitats, have a close association with submersed and emergent aquatic vegetation, are capable of surviving hypoxic events, and use a wide variety of habitats, we conclude water withdrawals even under a worst-case scenario are unlikely to affect these species in a measurable manner.

Littoral Zone, Marsh, and Floodplain Small Fishes Assemblage

Poeciliids (livebearers) and Cyprinodontids (killifishes) dominate this assemblage in numbers (Table 3–2; Appendix 12.E). Members of this assemblage are small (< 8 cm [3.1 in] TL) 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 they consequently are the most abundant fishes found in marshes or on the floodplain. Members of this assemblage are herbivores, detritivores, and/or invertivores, and most are labeled as phytophils (see Appendix 12.A). Members of this assemblage play an important role in regulating energy flow through the ecosystem. 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 River, the concentrating of small fishes that occurs with receding water levels creates an important food source for wading birds (see Chapter 13. Floodplain Wildlife) (Ogden 1994) as well as for predatory fishes that occupy more permanently flooded habitats. Because 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 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 water withdrawals will not affect SAV in the St. Johns River (see Chapter 9. Submersed Aquatic Vegetation), we will not further consider littoral populations of this assemblage. However, we will assess the potential effects of reduced floodplain inundation (see Chapter 4.1.1 Effects on Hydrology).

To assess potential effects of water withdrawal on floodplain small fishes, we developed a model that predicts maximum annual small fish densities on the floodplain from annual flooding durations for each of the withdrawal scenarios (see Appendix 12.E). Central to this analysis were results from a model relating densities of the small fishes assemblage to flooding durations developed for the Everglades (DeAngelis et al. 1997). We felt use of the Everglades model output was appropriate because of the similarities between the community composition of small fishes in the Everglades and St. Johns River floodplain marshes, and because of the hydrologic and physical similarities between the two systems (Appendix 12.E). Our analysis of withdrawal effects was conducted along seven MFL transects. MFL transect sites for which densities were calculated included two transects on Lake Poinsett (I-95 on the southeast corner and County Line on the northwest corner), Toso528, H1, Lake Monroe T4, Pine Island, and Lake Woodruff (see Figure 3-1).

To evaluate withdrawal effects we used the percent change in mean predicted fish densities between each withdrawal scenario and the baseline scenario (Base1995NN) rather than differences between predicted numbers themselves. Other data sets, such as stage-area curves and output from the digital elevation model (DEM) also provide a basis for calculating small fish densities 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 maximum annual floodplain small fish densities using a stage-area curve for Lake Poinsett and early DEM data for an area north of SR 50. Results using both were comparable to those obtained using nearby MFL transect data. For example, results generated by both by the DEM and the use of nearby MFL transect data suggested an average annual 8.8% decline in the maximum abundance of small fishes on the floodplain near SR 50 when comparing Base1995NN to the worst-case Full1995NN Scenario. Because the results using different data sources were similar, and MFL data provided for greater spatial coverage, we only report density data generated from MFL transects in this report.

The worst-case Full1995NN Scenario had the largest reductions below baseline conditions (Base1995NN) in estimated floodplain densities of the small fish assemblage (Figure 4–5). Upstream of Lake Monroe, estimated reductions ranged from 9.3 % at the H1 Transect to 11.3% at the County Line Transect in Lake Poinsett. From Lake Monroe downstream, estimated reductions ranged from 0.2% at Lake Woodruff to 10.0% at Pine Island (Figure 4–5). Under the Full1995PN Scenario, estimated small fish densities along the Lake Poinsett I-95 Transect increased 0.1% over Base1995NN, but along the County Line Transect they declined 2.7% (Figure 4–5). Under the Full1995PN Scenario, estimated small fish densities along the Toso528 and H1 Transects declined 4.6% and 4.7% from the Base1995NN Scenario, respectively. From Lake Monroe downstream, sea level rise (Full1995PS) caused dramatic increases in flooding durations that resulted in substantially higher estimates of small floodplain fish densities compared to Full1995PN (Figure 4–5). At Lake Woodruff, estimated small fish densities under the Full1995PS and Full2030PS were both more than 60% higher than under Base1995NN. Upstream of Lake Monroe, where there was no sea level effect (see Table 3–1), predicted increases in small fish densities under the Full2030PS scenario increased over the Base1995NN scenario from 1.5% to 3.7% (Figure 4–5). Increased small fish densities under the Full2030PS scenario reflect inflow augmentation from both the USJRBP and 1995 to 2030 land-use changes on basin runoff characteristics.

Under the potential near-term Full1995PN scenario, withdrawals are predicted to cause a reduction in the average annual abundance and subsequently the biomass of the floodplain small fish assemblage. Assuming an average mean weight of 0.4 g fish $^{-1}$, average maximum biomass of the Small Fish Assemblage declined approximately 4.7% under Full 1995PN compared to Base1995NN (based on averaged values for the Toso528 and H1 Transects) along the 100 km (62 mi) of river between Lake Poinsett and Lake Harney. Roughly spreading this predicted reduction out over the entire 16,314ha (40,312-ac) of floodplain between these points generates a total predicted loss of 4.3×10^6 (17,312 kg [38,174 lbs]) to 5.7×10^7 (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 use the floodplain as important foraging habitat, and a loss of small prey input to the river. Results for the Half1995PN scenario eliminates the 4.7% in small fish reduction predicted for Full1995PN at the Toso528 Transect and reduces the

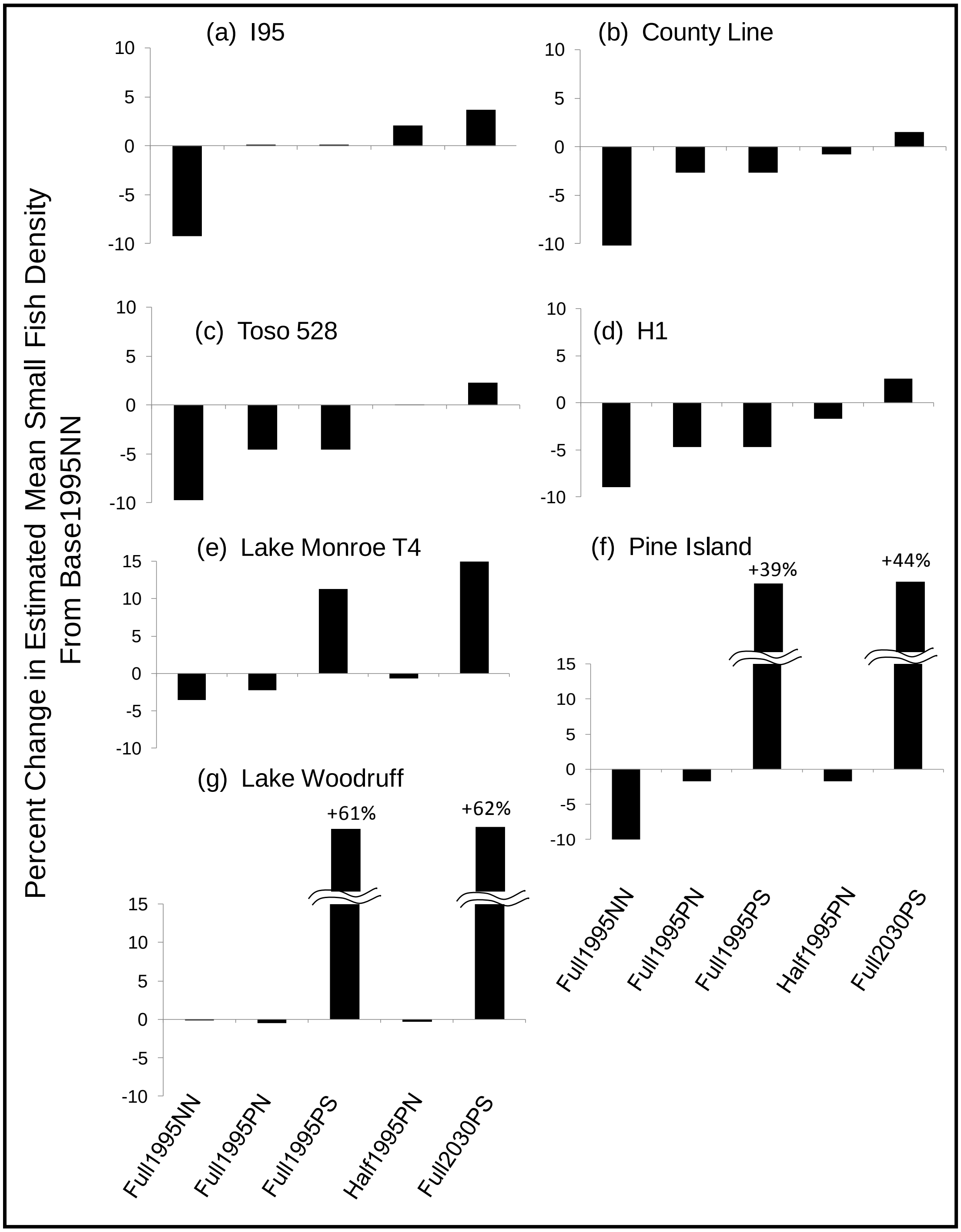

Figure 4–5. Percent change in the modeled mean density of small fishes produced on the floodplain for five withdrawal scenarios compared to Base1995NN at seven MFL transects located in the Upper and Middle St. Johns River basin. Sea level rise did not influence water levels along the I-95, County Line, Toso528, and H1 Transects (see Table 3–1).

predicted reduction to 1.7% at the H1 Transect (see Figure 4–5).

Our approach was simplistic in that it only analyzed flooding and drying on an annual basis and did not consider potential cumulative effects of multiyear droughts (DeAngelis et al. 1997). However, we felt this approach was appropriate because the occurrence and return frequency of droughts (defined as water contained within the channel banks) were unaffected by water withdrawals. In addition, drought intensity (defined by within-channel water levels) became more severe only under the worst-case scenario (Full1995NN) and remained relatively unchanged or even enhanced under all the other scenarios that contained low flow augmentation from the USJRBP (see Chapter 4.1.1 Effects on Hydrology).

Although flooding duration was the most important factor affecting small fish abundance in the Everglades marshes, other factors may also have an influence. These factors include differences in vegetation (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 recovery for individual species (DeAngelis and White 1994; Trexler et al. 2002), and anthropogenic nutrient inputs (Trexler et al. 2002). There are undoubtedly unaccounted for differences in these other factors between the Everglades and the St. Johns River basin, as well as differences between areas within the St. Johns River basin itself. However, given the overwhelming influence of flooding duration on the abundance of small fishes as a group (DeAngelis et al. 1997), we feel our approach is valid for making our relative assessment of effects on fish abundance and biomass due to water withdrawals. Although abundance estimates of small fishes in the St. Johns River basin (4 to 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.

4.1.3 Ichthyoplankton Entrainment

The ichthyoplankton (fish eggs and larvae) sampling effort collected 708,032 individual fish larvae made up primarily of 16 species (Table 4–1). The highest total catches were at SR 46 (river km 310 [river mi 193]) and Lake Monroe (near river km 265 [river mi 165]); average catch per transect sampled at these two sites was 2.5 to 5 times higher than at any other station (Table 4–1). The lowest average catches were at SR 50 (river km 343 [river mile 213]) and Lake Poinsett (river km 378 [river m 235]). Gizzard and threadfin shad dominated the catch at all stations, comprising greater than 69% of the catch at all sites combined. Gobies were the next most abundant group, with clown goby and naked goby together comprising greater than 16% of the total catch. The only other species to comprise more than 2% of the total catch were black crappie, bluegill, and tidewater silverside. While gizzard and threadfin shad dominated the catch at all sites, there were some notable differences in species composition between sites. Although total catch of all fishes was lowest at SR 50, the 11,883 American shad larvae collected at this site were more than 10 fold the number collected anywhere else. The second highest catch of American shad occurred at SR 46 (river km 310 [river mile 193]), along with the highest catches of hickory shad and blueback herring. The greatest number of catfishes (channel and white) also occurred at SR 46. Both clown and naked gobies were rare in Lake Poinsett and at SR 50 (< 100), but numbers increased dramatically from SR 46 to Yankee Lake (> 24,000). Catch of swamp darters was highest in Lake Poinsett and decreased moving downstream, whereas black

Table 4–1. Species composition of ichthyoplankton (larval fish) collected from six St. Johns River sites from February 2008 through September 2009. See Appendix 12.B for sample site descriptions.

Common Name	Scientific Name	Lake Poinsett	SR 50	SR 46	Lake Monroe	Yankee Lake	SR 44
American shad	*Alosa sapidissima*	109	11,883	674	493	313	295
Blueback herring	*Alosa aestivalis*	0	52	145	70	19	11
Hickory shad	*Alosa mediocris*	3	18	100	53	26	14
Gizzard shad	*Dorosoma cepedianum*	38,848	18,237	102,593	29,884	28,319	33,049
Threadfin shad	*Dorosoma petense*	6,590	2,350	103,513	89,009	24,594	16,435
Unidentified *Dorosoma spp.*		44	7	11,296	831	13	4,862
Channel catfish	*Ictalurus punctatus*	0	3	227	1	4	1
White catfish	*Ameiurus catus*	3	27	565	8	31	18
Tailight shiner	*Notropis maculatus*	98	12	62	14	24	51
Tidewater silverside	*Menidia beryllina*	5,757	1,212	3,247	2,638	1,572	643
Rough silverside	*Membras martinica*	0	3	57	93	188	127
Bluegill	*Lepomis macrochirus*	3,095	519	5,277	1,656	1,738	3,337
Redear sunfish	*Lepomis microlophus*	950	370	1,837	346	910	531
Black crappie	*Pomoxis nigromaculatus*	3,575	377	1,939	1,379	3,596	9,800
Clown goby	*Microgobius gulosus*	7	25	17,377	11,658	15,733	5,387
Naked goby	*Gobiosoma bosc*	2	33	14,296	16,867	13,851	4,786
Unidentified gobies		0	33	9,236	931	3,067	2,913
Swamp darter	*Etheostoma fusiforme*	2,712	908	287	36	80	139
Others		175	381	299	130	3,855	158
Totals		61,968	36,450	273,027	156,097	97,933	82,557
Number of transects		6	5	6	4	5	5
Average catch per transect		10,328	7,290	45,505	39,024	19,587	16,511

crappie catch was nearly three times higher at SR 44 than at any other site (Table 4–1).

The temporal distribution of ichthyoplankton catch (all species combined) was similar among sites (Figure 4–6). Eggs and larvae began appearing in late November, peaked in February through May, and generally ended by mid-August. In 2009, the catch of all species in Lake Monroe was markedly lower than in 2008. The temporal distribution of catch also varied among species reflecting different spawning periods.

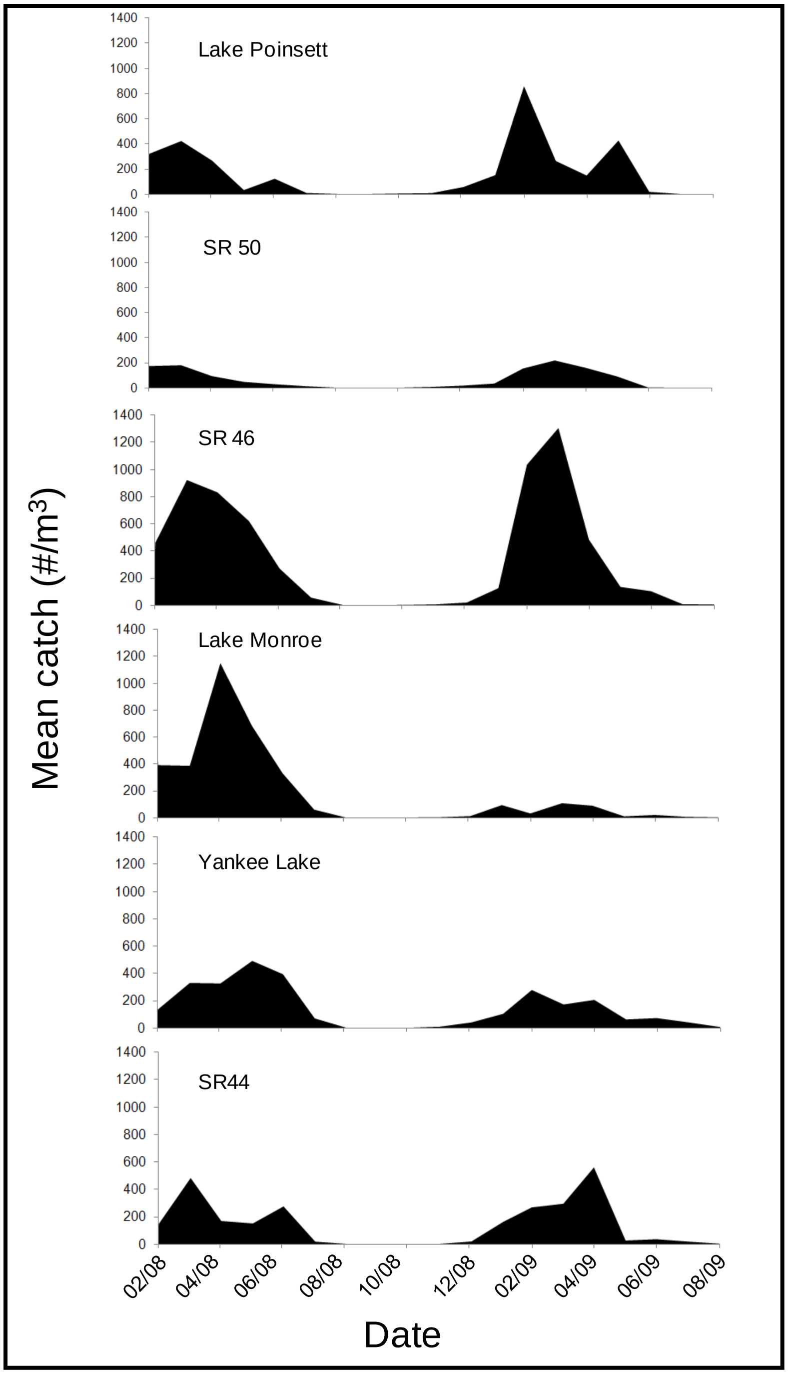

Figure 4–6. Monthly distribution of mean ichthyoplankton (larval fish) catch (number m^3) at six St. Johns River sites from February 2008 through September 2009. See Appendix 12.B for sample site descriptions.

4.2 Effects of Water Withdrawals on Fishes in Estuarine Reaches

4.2.1 Effects on Freshwater inflow

Hydrologic modeling indicates that freshwater inflow into the estuary under the Full1995NN, Half1995PN, and Full1995PN Scenarios, will be lower by an average of 6.7 $m^3 s^{-1}$ (237 cfs), 2.4 $m^3 s^{-1}$ (85 cfs), and 5.7 $m^3 s^{-1}$ (210 cfs), respectively, as compared to Base1995NN (Figure 4–7a). A reduction of 6.7 $m^3 s^{-1}$ (237 cfs) equates to a reduction of approximately 152 mgd. Under all the 1995 withdrawal scenarios, all monthly average inflows are less than the Base1995NN Scenario, with the lowest monthly reductions occurring under the Half1995PN Scenario.

For the Full2030PS Scenario, average yearly inflows to the estuary are higher by 2.4 $m^3 s^{-1}$ (84.7 cfs) than Base1995NN, but this is due to higher inflow in the wet season (Figure 4–7b). During the dry season months (December to May), average freshwater inflows under Full2030PS are 1.3 $m^3 s^{-1}$ (46cfs) less than Base1995NN. During the wet season (June to November), freshwater inflows under Full2030PS exceed Base1995NN by 6.1 $m^3 s^{-1}$ (215 cfs). Under Half2030PS, average annual inflow to the estuary increases by 5.7 $m^3 s^{-1}$ (201 cfs) over Base1995NN; while dry season inflow is approximately the same as Base1995NN, wet season inflows are approximately 10.1 $m^3 s^{-1}$ (357 cfs) higher. Under FwOR2030PS, average annual freshwater inflows are 2.3 $m^3 s^{-1}$ (81 cfs) lower than Base1995NN; although wet season inflows are 1.5 $m^3 s^{-1}$ (53 cfs) higher, dry season inflows are 6.2 $m^3 s^{-1}$ (219 cfs) lower (Figure 4–7b).

4.2.2 Effects on Salinity

The EFDC hydrodynamic model output indicates that water withdrawals would have little effect on the overall spatial coverage of various salinity habitats in the LSJRB estuary (Figure 4–8, Figure 4–9, and Figure 4–10). This is consistent with the conclusions reached by the Submersed Aquatic Vegetation Working Group (Chapter 9 Submersed Aquatic Vegetation).

The scenario that would cause the greatest salinity increases in the estuary is FwOR1995NN. Under the FwOR1995NN Scenario, the average annual spatial coverage of lowest salinity limnetic (open water) habitat is only reduced by 538 ha (1,329 ac) or 3.9% from the Base1995NN Scenario when comparing differences in highest mean 30-day salinity (Figure 4–8a). Concurrently, the highest salinity polyhaline and euhaline habitats under FwOr1995NN increased over Base1995NN by only 122 ha (301 ac, 2.0%) and 246 ha (605 ac, 11.5%), respectively (Figure 4–8e and Figure 4–8f). Predicted salinity habitat changes between the two scenarios were less when comparing changes in the highest 60- and 120-day salinity averages (Figure 4–9 and Figure 4–10).

Under the potential long-term scenario Full2030PS, the spatial coverage of limnetic habitat increased slightly as compared to Base1995NN regardless of the salinity duration considered (Figure 4–8, Figure 4–9, and Figure 4–10). With the increase in limnetic habitat under Full2030PS Scenario, there was a corresponding decline in the coverage of oligohaline habitat. Spatial coverage of all other habitats remained virtually unchanged between the Base1995NN and the Full2030PS Scenarios. Adding an additional withdrawal from the Ocklawaha River (FwOR2030PS scenario) caused a slight decline as compared to Base1995NN in the spatial coverage of limnetic and oligohaline habitats, and a slight increase in polyhaline and euhaline habitats. Based on these analyses, we conclude that water withdrawals under all the withdrawal scenarios modeled will have a negligible effect on the spatial coverage of the various salinity habitats as defined here. Given the overlap of the confidence intervals, it is also unlikely that any changes that do occur will be statistically detectable.

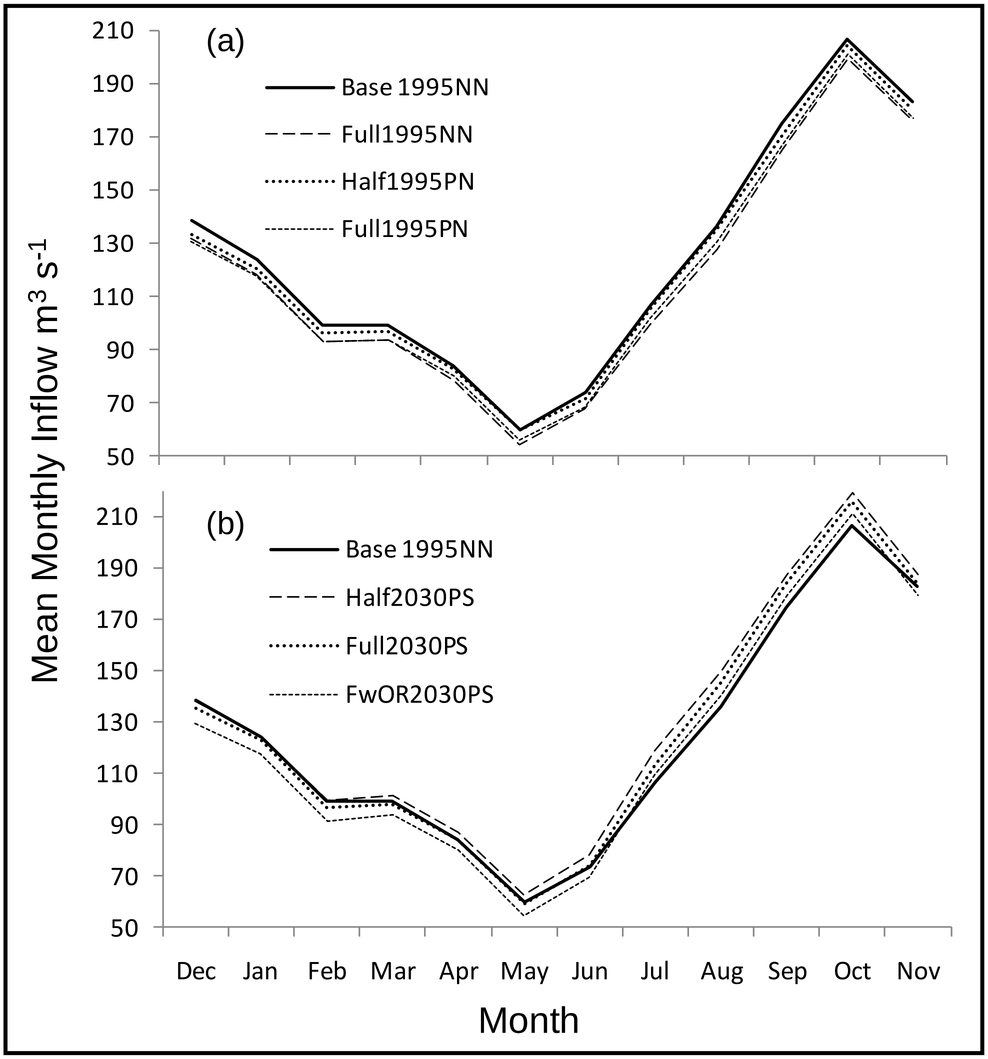

Figure 4–7. Predicted mean monthly inflows ($m^3\ s^{-1}$) for the period 1995 to 2005 simulated by the EFDC hydrodynamic model for (a) 1995 and (b) 2030 withdrawal scenarios compared to Base1995NN.

4.2.3 Effects on Submersed Aquatic Vegetation (SAV)

SAV provides important habitat for many species of fishes and macroinvertebrates in the Lower Basin estuary (CSA (Continental Shelf Associates) 1993; Jordan 2000; MacDonald et al. 2009). SAV occurs in shallow areas from river km 45 to the upstream limit of the estuary (see Chapter 9 Submersed Aquatic Vegetation). In the estuary, the dominant SAV species is American eelgrass (*Vallisneria americanum*). Jordan (2000) provided a comprehensive survey of fishes found in LSJRB SAV habitats using both throw traps and seines. SAV supported significantly more individuals and more species than did adjacent sand flats in nearshore habitats. Brown bullhead,

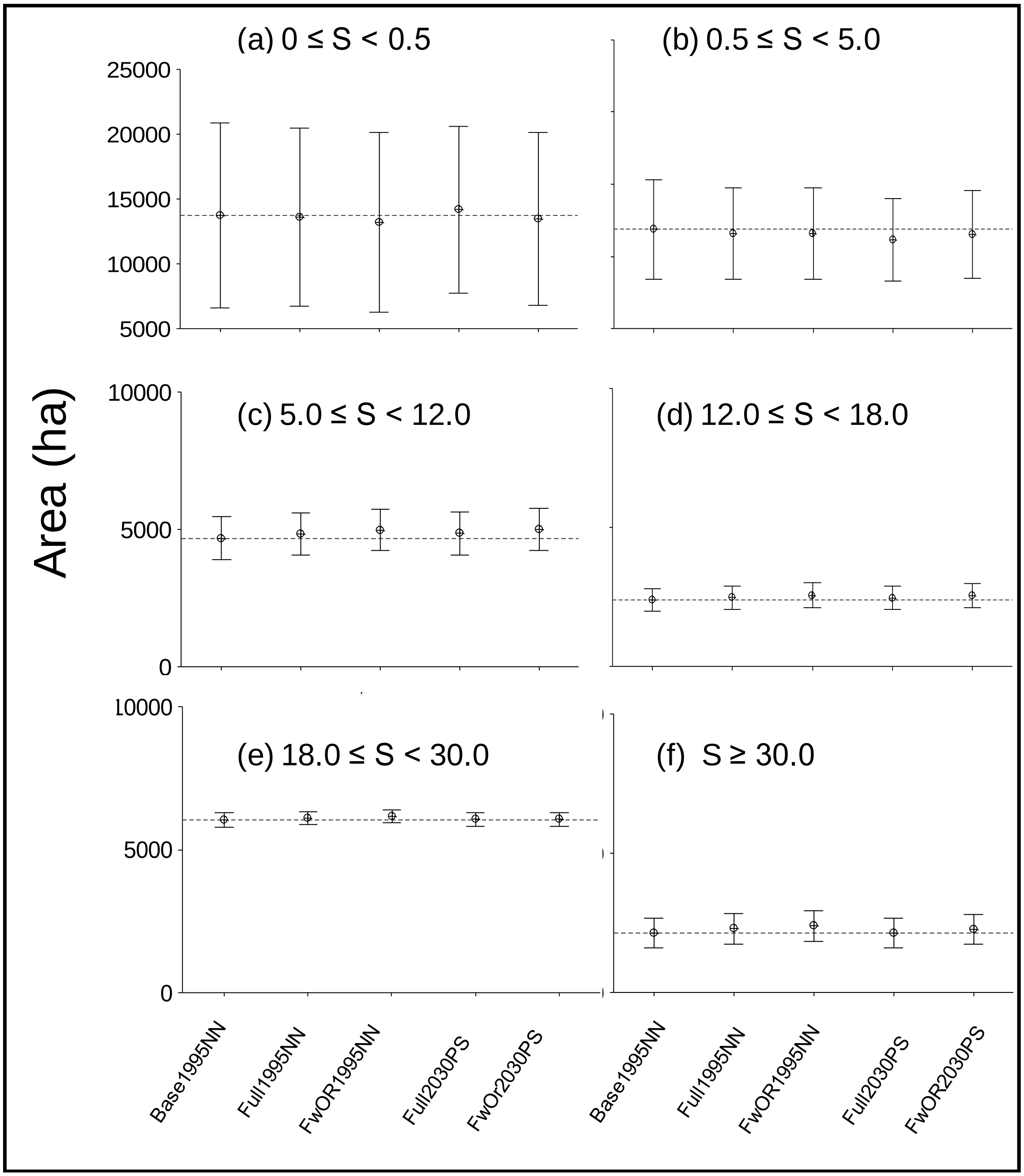

Figure 4–8. Mean annual areal coverage (ha) ±95% CI of various salinity habitat units (S) in the lower St. Johns River basin (LSJRB) estuary (river km 0 to 160 [river mi 99]) for five modeled scenarios. Annual values were derived for the years 1996 to 2005 and were calculated for the exact same date range that was used to calculate the highest mean 30-day salinity of the base scenario (Base1995NN). Salinity habitat units reflect those used in the FIM analysis (a) limnetic (0 ‰ ≤ S < 0.50 ‰), (b) oligohaline (0.5 ‰ ≤ S < 5.0‰), (c) low mesohaline (5.0 ‰ ≤ S < 12.0 ‰), (d) high mesohaline (12.0 ‰ ≤ S < 18.0 ‰), (e) polyhaline (18.0 ‰ ≤ S < 30.0 ‰), and (f) euhaline (S ≥30.0 ‰). (Dashed lines represent mean area for the Base1995NN Scenario.)

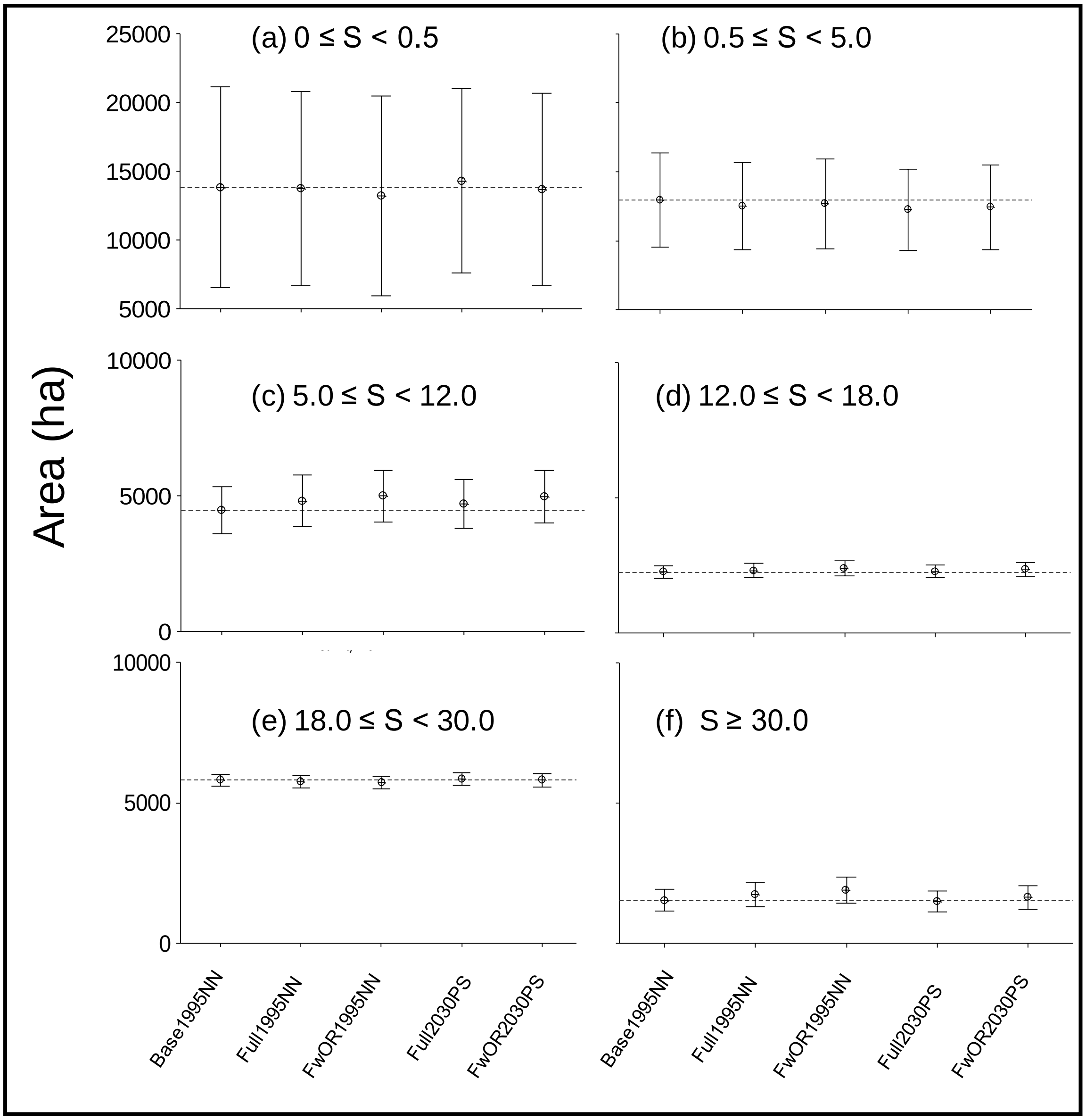

Figure 4–9. Mean annual areal coverage (ha) ±95% CI of various salinity habitat units (S) in the lower St. Johns River basin (LSJRB) estuary (river km 0 to 160 [river mi 99]) for five modeled scenarios. Annual values were derived for the years 1996 to 2005 and were calculated for the exact same date range that was used to calculate the highest mean 60-day salinity of the base scenario (Base1995NN). Salinity habitat units reflect those used in the FIM analysis (a) limnetic (0 ‰ ≤ S < 0.50 ‰), (b) oligohaline (0.5 ‰ ≤ S < 5.0‰), (c) low mesohaline (5.0 ‰ ≤ S < 12.0 ‰), (d) high mesohaline (12.0 ‰ ≤ S < 18.0 ‰), (e) polyhaline (18.0 ‰ ≤ S < 30.0 ‰), and (f) euhaline (S ≥30.0 ‰). (Dashed lines represent mean area for the Base1995NN Scenario.)

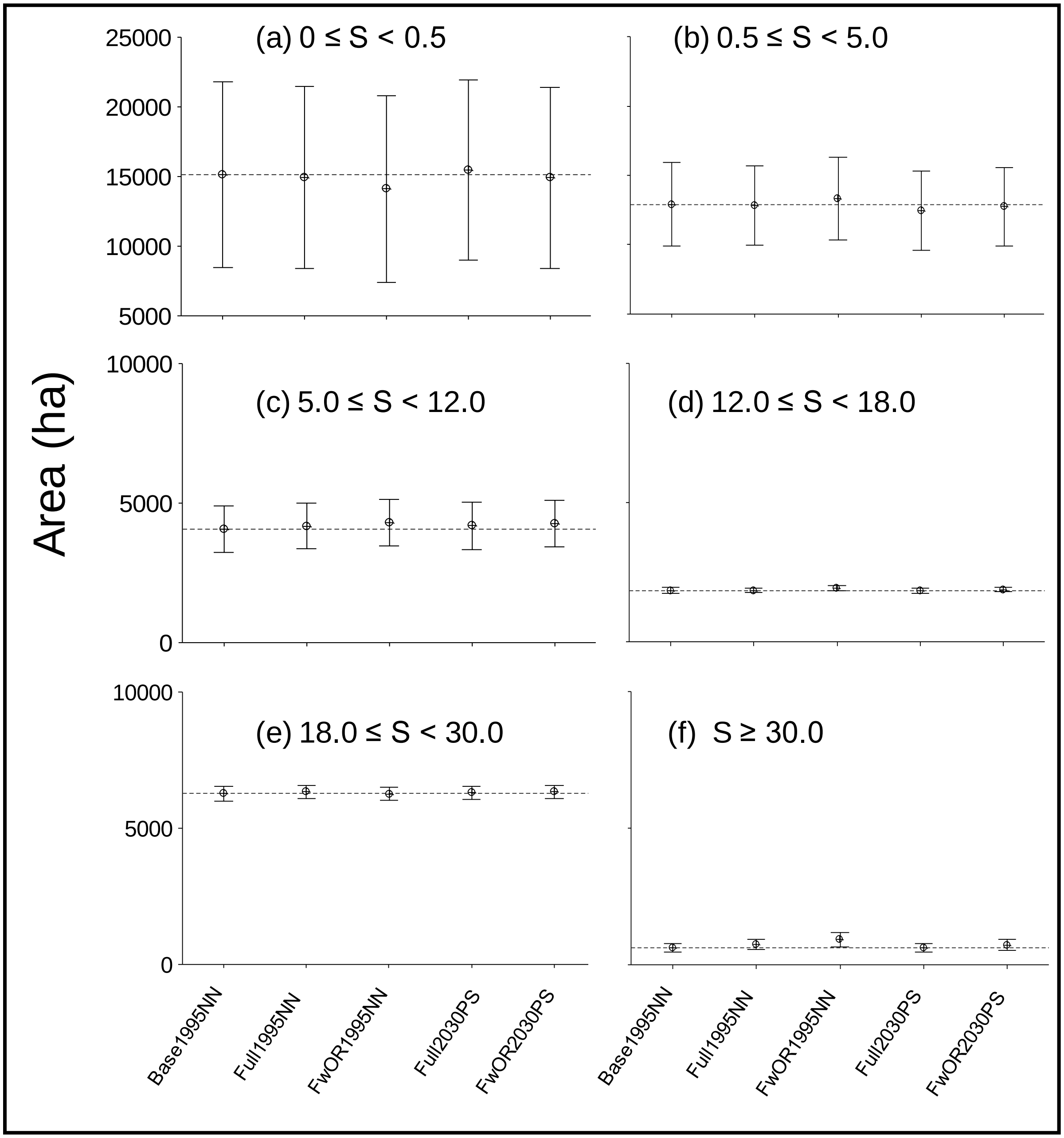

Figure 4–10. Mean annual areal coverage (ha) ±95% CI of various salinity habitat units (S) in the lower St. Johns River basin (LSJRB) estuary (river km 0 to 160 [river mi 99]) for five modeled scenarios. Annual values were derived for the years 1996 to 2005 and were calculated for the exact same date range that was used to calculate the highest mean 120-day salinity of the base scenario (Base1995NN). Salinity habitat units reflect those used in the FIM analysis (a) limnetic (0 ‰ ≤ S < 0.50 ‰), (b) oligohaline (0.5 ‰ ≤ S < 5.0‰), (c) low mesohaline (5.0 ‰ ≤ S < 12.0 ‰), (d) high mesohaline (12.0 ‰ ≤ S < 18.0 ‰), (e) polyhaline (18.0 ‰ ≤ S < 30.0 ‰), and (f) euhaline (S ≥30.0 ‰). (Dashed lines represent mean area for the Base1995NN Scenario.)

largemouth bass, naked goby, rainwater killifish, Seminole killifish, and various sunfishes were significantly more abundant in SAV. Clown goby, tidewater silverside, and spot, however, were significantly more abundant on adjacent unvegetated sand flats. Finally, Atlantic croaker, bay anchovy, and freshwater goby were distributed randomly between SAV and open sand habitats, and did not exhibit a preference for either (Jordan 2000). Of the species that were most abundant in SAV, only rainwater killifish and naked goby are associated primarily with brackish water (Loftus and Kushlan 1987). Fish community composition in SAV did not vary in a consistent fashion along the estuarine gradient, and fish densities within SAV were related more to the presence and health of SAV than to salinity (Jordan 2000).

Similar relationships among fish species distribution and SAV and salinity in the St. Johns River were found by MacDonald et al. (2009). Additional freshwater species found to be more abundant in SAV in the FIM analyses include channel catfish, white catfish, longnose gar, gizzard shad, and golden shiner. Other species associated with brackish water that were found to prefer SAV habitats were Atlantic needlefish, gulf pipefish, and pinfish. Results from these studies suggest that SAV may be more important as physical habitat to freshwater than to estuarine fishes in the LSJRB. Although the Submersed Aquatic Vegetation Working Group found that withdrawals did not influence the distribution or abundance of SAV in the LSJRB (see Chapter 9 Submersed Aquatic Vegetation), relative abundance data presented in Jordan (2000) and the FIM report (MacDonald et al. 2009) could be used to develop predictive models that quantify potential effects on fishes associated with SAV habitat losses or gains.

4.2.4 Correlation and Regression Analyses of Estuarine Fish Responses

From 2001 through 2010, the FWC FIM program conducted 7,467 sampling events in the LSJRB estuary, and collected 854,233 individuals representing at least 160 species (MacDonald et al. 2009). More than 80% of the 21.3-m seine catch was comprised of bay anchovy, Atlantic silverside, spot, rainwater killifish, striped mullet, white shrimp, striped anchovy, Atlantic croaker, menhaden, and bluegill. The nekton collected in the 183-m seine tended to be larger bodied animals than in the 21.3-m seine; menhaden, striped mullet, pinfish, spot, white mullet, and Atlantic croaker were dominant (> 65% of the total catch). Fish collections with the 6.1-m otter trawl were dominated by Atlantic croaker, bay anchovy, spot, and white shrimp (> 78% of the total catch).

For all gear types combined, 444 pseudospecies were abundant enough to qualify for the Spearman’s correlation analysis (> 99 individuals collected by the gear and at least a 5% frequency of occurrence in all samples). A pseudospecies designation includes a species-specific size class grouped by collection gear, FIM zone of collection, and recruitment period. The 444 pseudospecies represented 57 individual species; many species were analyzed more than once because they were represented by more than one size group or were collected by more than one gear type. Spearman’s correlation was conducted on 3,912 combinations of abundance (both monthly and annual) versus lagged inflow and 354 combinations of center-of-distribution versus lagged inflow. See Appendix 12.F for a summary of all Spearman’s correlations and linear regression analyses conducted.

Of 57 fish and invertebrate species abundant enough to be analyzed, 47 (82%) exhibited a significant abundance response to changes in freshwater inflow within at least one size-class.

There were two invertebrate species, blue crab and white shrimp, that were included in our statistical analyses because they were part of the FIM data set. We present results for blue crab and white shrimp in summary tables in this chapter but defer discussion of the results to the Macroinvertebrate Working Group (see Chapter 11 Benthic Macroinvertebrates). Seven hundred and eighty-seven Spearman's correlations relating pseudospecies relative abundance to inflow were significant ($p < 0.05$); 65 Spearman's correlations relating center-of-abundance to inflow were also significant ($p < 0.05$). Screening to exclude pseudospecies whose highest absolute rho values were less than 0.4 reduced the number of abundance to inflow combinations to 638 and the number of center-of-distribution to inflow combinations to 58. Combinations were then screened to include only those whose rho values were within ±3% of the highest rho value for each individual pseudospecies, which reduced the number of abundance to inflow combinations to 191, and the number of center-of-abundance to inflow combinations to 32. Because both raw and transformed data were used in the linear regression analyses, the total number of regressions run on abundance to inflow data and abundance to center-of-abundance data were 764, and 128, respectively. Finally, regression analyses were screened to select regressions with the highest PRESS r^2, leaving 61 pseudospecies abundance to inflow regressions and 20 pseudospecies center-of-abundance to inflow regressions for use in quantitatively predicting water withdrawal effects (Table 4–2). The 61 pseudospecies abundance to inflow regressions represented 34 species, and the 20 pseudospecies center-of-abundance to inflow regressions represented 14 species. Eleven species exhibited both an abundance and distributional response to inflow. Regression statistics for all pseudospecies, including those with $r^2 < 0.25$, are presented in Appendix 12.F.

4.2.5 Estuarine Fish Distribution Responses

Two freshwater and 12 estuarine or marine pseudospecies exhibited strong distribution responses to freshwater inflow (Table 4–2). For all of these pseudospecies, the center-of-abundance shifted downstream in response to increasing freshwater inflow. This is consistent with the hypothesis that higher freshwater inflows should expand freshwater habitat in the estuary, while contracting the habitat for more salt-tolerant species. Conversely, decreasing freshwater inflows due to water withdrawals should contract freshwater habitat and expand higher salinity habitats. A large number of pseudospecies exhibited weak distribution responses to freshwater inflow that are not discussed here but can be found in Appendix 12.F. These weak pseudospecies responses in large part mirrored the distributional shifts of other size groups of the same species that had strong responses.

The lag times used to relate distributional response to freshwater inflow were all relatively short (30 to 90 days). These lag times correspond to the estimated time it takes the gauged inflows used in the regression analyses to reach the lower reaches of the St. Johns River estuary. By using only a limited number of relatively short lagged inflows, our analyses of distribution-to-inflow relationships can only elucidate more short-term responses to freshwater inflow, and do not identify distribution responses of pseudospecies that respond more markedly to long-term inflow patterns.

Applying modeled inflow output to the regressions for the 20 pseudospecies that had strong distribution responses to freshwater inflow indicates that the predicted median center-of-abundance for each pseudospecies will move upstream in response to reduced inflows expected

Table 4–2. Responses to freshwater inflow of fishes and invertebrates (represented by pseudospecies) in the St. Johns River estuary. Only pseudospecies with strong abundance and distribution responses were used to quantitatively predict withdrawal effects. Weak distribution responses ($r^2 < 0.25$) are not reported.

Common Name (*Scientific Name*)	**Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)**	**Monthly (m) or Annual (a) Response to Inflow**	**Best-fit (Highest PRESS r^2) Lag Time (days)**	**Response**
Strong distribution response to freshwater inflow (significant linear regression with r^2>0.25) All distribution responses were calculated on monthly data only.				
Atlantic bumper (*Chloroscombrus chrysurus*)	50 to 110	m	30	↓
Atlantic croaker (*Micropogonias undulatus*)	26 to 40	m	30	↓
	41 to 60	m	60	↓
	61 to 85	m	30	↓
Atlantic weakfish (*Cynoscion regalis*)	41 to 75	m	30	↓
Bay whiff (*Citharichthys spilopterus*)	51 to 90	m	60	↓
Channel catfish (*Ictalurus punctatus*)	50 to 100	m	90	↓
Clown goby (*Gobiodon okinawae*)	0 to 28	m	30	↓
Hogchoker (*Trinectes maculatus*)	20 to 45	m	90	↓
Naked goby (*Gobiosoma bosc*)	20 to 35	m	60	↓
Pinfish (*Lagodon rhomboids*)	131 to 160	m	90	↓
Redear sunfish (*Lepomis microlophus*)	0 to 125	m	60	↓
Silver perch (*Bairdiella chrysora*)	0 to 30	m	60	↓
	31 to 55	m	60	↓
	56 to 85	m	90	↓
Spot (*Leiostomus xanthurus*)	61 to 90	m	30	↓
White shrimp (*Penaeus setiferus*)	0 to 15	m	30	↓
	0 to 15	m	90	↓
	4 to 11	m	30	↓
Striped mullet (*Mugil cephalus*)	31 to 45	m	30	↓
Strong abundance response to freshwater inflow (significant linear regression with r^2 > 0.25)				
Atlantic croaker (*Micropogonias undulates*)	131 to 170	m	120	–
Atlantic thread herring (*Opisthonema oglinum*)	70 to 110	m	60	–
Atlantic weakfish (*Cynoscion	41 to 75	m	90	–

Common Name (*Scientific Name*)	**Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)**	**Monthly (m) or Annual (a) Response to Inflow**	**Best-fit (Highest PRESS r^2) Lag Time (days)**	**Response**
regalis)				
Bay anchovy (*Anchoa mitchilli*)	36 to 600	a	360	–
Bay whiff (*Citharichthyes spilopterus*)	50 to 70	a	90	–
	71 to 100	a	300	–
Blue crab (*Callinectes sapidus*)	91 to 170	m	180	–
	111 to 180	m	180	–
Bluegill (*Lepomis macrochirus*)	20 to 65	m	90	+
	20 to 65	m	300	+
Channel catfish (*Ictalurus punctatus*)	50 to 100	m	180	+
	150 to 275	a	150	+
Clown goby (*Gobiodon okinawae*)	29 to 36	m	270	–
	37 to 56	m	300	–
Freshwater goby (*Ctenogobius shufeldti*)	0 to 50	m	360	+
	30 to 55	m	360	+
	30 to 55	a	360	+
Fringed flounder (*Etropus crossotus*)	50 to 85	m	60	–
	61 to 90	m	30	–
Golden shiner (*Notemigonus chrysoleucas*)	0 to 50	m	180	–
Gulf flounder (*Paralichthyes albigutta*)	60 to 180	m	360	–
	60 to 180	a	150	–
Gulf pipefish (*Syngnathus scovelli*)	0 to 120	m	210	–
Hogchoker (*Trinectes maculates*)	20 to 45	m	60	+
	20 to 45	a	60	+
Irish pompano (*Diapterus auratus*)	60 to 110	a	360	+
Mummichog (*Fundulus heteroclitus*)	0 to 34	m	360	–
	0 to 34	a	30	–
Naked goby (*Gobiosoma bosc*)	20 to 35	m	210	–
	20 to 35	a	240	–
Pigfish (*Orthopristis chrysoptera)*	80 to 130	m	360	–
Pinfish (*Lagodon rhomboids*)	36 to 70	m	300	–
	101 to 130	m	330	–
	131 to 160	m	360	–
	101 to 130	a	300	–
	131 to 160	a	30	–
Rainwater killifish (*Lucania parva*)	0 to 32	m	270	–
Redbreast sunfish (*Lepomis auritus*)	20 to 110	a	360	+
	131 to 190	a	30	–
Redear sunfish (*Lepomis microlophus*)	0 to 125	m	360	+
Silver perch (*Bairdiella chrysora*)	31 to 55	m	90	–
	80 to 100	m	90	–

Common Name (*Scientific Name*)	Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)	Monthly (m) or Annual (a) Response to Inflow	Best-fit (Highest PRESS r^2) Lag Time (days)	Response
Silversides (*Menidia* spp.)	41 to 55	m	60	–
Southern flounder (*Paralichthyes lethostigma*)	0 to 50	m	30	+
	51 to 100	m	210	+
	126 to 325	a	150	–
Southern puffer (*Sphoeroides nephelus*)	70 to 170	a	210	–
Spot (*Leiostomus xanthurus*)	41 to 60	m	60	+
	60 to 90	m	60	–
	60 to 90	a	30	–
	91 to 120	m	60	–
Spotted seatrout (*Cynoscion nebulosus*)	31 to 50	m	150	–
	51 to 110	m	300	–
	210 to 325	a	60	+
Striped burrfish (*Chilomycterus schoepfi*)	40 to 110	a	270	–
Striped mullet (*Mugil cephalus*)	31 to 45	a	210	+
Tidewater mojarra (*Eucinostomus harengulus*)	91 to 110	m	360	–
White catfish (*Ameiurus catus)*	25 to 100	m	300	+
	101 to 200	m	360	+
White mullet (*Mugil curema*)	31 to 80	m	180	+
	100 to 130	m	150	–
Weak abundance response to freshwater inflow (Spearman's rho > 0.40, but linear regression r^2 < 0.25. For pseudospecies with nonsignificant regression, lag times are highest Spearman's rho's.				
Atlantic needlefish (*Strongylura marina*)	325 to 500	a	30	–
	325 to 500	a	120	–
Atlantic croaker (*Micropogonias undulatus*)	60 to 100	m	240	–
Bay anchovy (*Anchoa mitchilli*)	20 to 35	m	180	–
Crevalle jack (*Caranx hippos*)	50 to 200	m	240	–
Lookdown (*Selene vomer)*	30 to 80	a	270	–
	30 to 80	a	300	–
	30 to 80	a	330	–
Pinfish (*Lagodon rhomboides*)	70 to 100	m	360	–
Redear sunfish(*Lepomis microlophus*)	20 to 80	m	90	+
Red drum (*Sciaenops ocellatus*)	0 to 35	m	210	–
	0 to 35	m	240	–
	36 to 80	m	360	–
Silver jenny (*Eucinostomus gula)*	65 to 90	m	120	–

Common Name (*Scientific Name*)	Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)	Monthly (m) or Annual (a) Response to Inflow	Best-fit (Highest PRESS r^2) Lag Time (days)	Response
Silver perch (*Bairdiella chrysora*)	0 to 30	m	120	–
	100 to 130	a	90	+
	131 to 170	m	30	+
Silversides(*Menidia* spp.)	25 to 40	m	120	–
Southern flounder (*Paralichthyes lethostigma)*	0 to 50	m	60	+
Spot(*Leiostomus xanthurus*)	26 to 40	m	30	+
Tidewater mojarra (*Eucinostomus harengulus)*	60 to 90	m	120	–
White shrimp (*Penaeus setiferus*)	12 to 27 (POH)	m	210	+
Very weak abundance response to freshwater inflow (Spearman's rho < 0.40; p < 0.05) Lag times are highest Spearman rho's.				
Atlantic needlefish (*Strongylura marina)*	100 to 175	m	60	–
Atlantic stingray (*Dasyatis Sabina*)	125 to 325 (DW)	m	90	–
Blue crab (*Callinectes sapidus*)	10 to 50 (CW)	m	360	+
	40 to 90 (CW)	m	30	–
	81 to 110 (CW)	m	120	–
Bluegill (*Lepomis macrochirus*)	60 to 120	m	360	+
Channel catfish (*Ictalurus punctatus*)	101 to 350	m	360	+
Clown goby (*Microgobius gulosus*)	19 to 28	m	270	–
Darter goby (*Ctenogobius boleosoma)*	0 to 35	m	30	+
Gizzard shad (*Dorosoma cepedianum)*	201 to 375	m	270	–
Hogchoker (*Trinectes maculatus*)	0 to 30	m	30	+
	31 to 60	m	360	+
	46 to 75	m	300	+
Largemouth bass (*Micropterus salmoides*)	126 to 250	m	360	+
Lined sole (*Achirus lineatus*)	10 to 60	m	30	+
Mummichog (*Fundulus heteroclitus*)	35 to 60	m	360	–
Silver perch (*Bairdiella chrysora*)	10 to 40	a	30	–
Southern flounder (*Paralichthyes lethostigma*)	126 to 325	m	120	–
Spotted seatrout (*Cynoscion nebulosus*)	0 to 30	m	180	–
Striped anchovy (*Anchoa hepsetus*)	0 to 45	m	30	–

Common Name (*Scientific Name*)	Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)	Monthly (m) or Annual (a) Response to Inflow	Best-fit (Highest PRESS r^2) Lag Time (days)	Response
Threadfin shad (*Dorosoma petense*)	75 to 125	m	230	–
White mullet (*Mugil curema)*	151 to 200	m	360	–
No abundance response to freshwater inflow.				
Atlantic bumper (*Chloroscombrus chrysurus*)	50 to 110	m,a		NR
Atlantic silverside (*Menidia menidia*)	0 to 75	m, a		NR
Blackcheek tonguefish (*Symphurus plagiusa*)	20 to 110	m, a		NR
Eastern mosquitofish (*Gambusia holbrooki*)	0 to 32	m		NR
Ladyfish (*Elops saurus*)	150 525	m, a		NR
Longnose gar (*Lepisosteus osseus*)	675 to 950	m, a		NR
Atlantic Menhaden (*Brevoortia tyrannus*)	20 to 40	m, a		NR
Seminole killifish (*Fundulus seminolis*)	0 to 80	m		NR
Southern kingfish (*Menticirrhus americanus*)	10 to 40	m, a		NR
Striped mullet (*Mugil cephalus*)	0 to 30	m, a		NR
White mullet (*Mugil curema*)	0 to 30	m, a		NR

Note:
CW = carapace width (mm)
DW = disk width (mm)
POH = post orbital head width (mm)
\+ = increasing abundance with increasing inflow
– = decreasing abundance with increasing inflow
↑ = upstream movement in distribution response to increasing inflow
↓ = downstream movement in distribution response to increasing inflow
NR = no response

under the three withdrawal scenarios that lack augmentation effects (Full 1995NN, Half 1995PN, and Full1995PN) (Table 4–3). The smallest predicted upstream movement occurred under Half1995PN. However, predicted differences even under the worst-case scenario (Full1995NN) were relatively small (< 3 km [1.9 mi]) (Table 4–3).

Under the Half2030PS Scenario, inflow augmentation caused a downstream shift in the center-of-abundance of all pseudospecies (Table 4–3). Under Full2030PS, seven pseudospecies exhibited small upstream distribution shifts, and 13 pseudospecies moved downstream. Under FwOR2030PS, the number of pseudospecies moving upstream increased to 13 with some pseudospecies (e.g., 31 to 45 mm [1.2 to 1.8 in] SL striped mullet) exhibiting shifts in distance similar to that observed under Full1995NN. Box and whisker plots of distributional responses of the 20 pseudospecies as compared to Base1995NN are presented in Appendix 12.G.

4.2.6 Estuarine Fish Abundance Responses

Of the 61 pseudospecies that exhibited a strong relative abundance to freshwater inflow response, 10 pseudospecies (representing six species) are considered strictly freshwater fishes, although many are known to tolerate mild salinities. Of the freshwater species, bluegill, channel catfish, white catfish, and redear sunfish had pseudospecies whose relative abundance only increased in the estuary in response to increasing freshwater inflow (Table 4–2). Exceptions to this trend by freshwater fishes were redbreast sunfish and golden shiner, which both had some pseudospecies size-groups that increased in relative abundance with increasing inflows while other size-groups of the same species decreased in relative abundance with increasing inflows. Many responses of freshwater as well estuarine pseudospecies were documented only within a few FIM sampling zones and do not necessarily reflect an estuarine-wide response. For example, small bluegill (20 to 65 mm [0.8 to 2.6 in] SL) only increased in relative abundance in response to freshwater inflows in FIM sampling zones 3 and 4 (see Figure 3–2 for sampling zones). These zones are generally located at the shifting interface between oligohaline and low mesohaline salinity habitats. Specific FIM sampling zones to which all pseudospecies responses were determined are presented in Appendix 12.F.

The relative abundance of 39 estuarine or marine pseudospecies declined in response to increasing freshwater inflows, while 12 pseudospecies increased (Table 4–2). Hogchoker, Irish pompano, and freshwater goby were the only species that increased in abundance with increasing freshwater inflow across all size groups sampled; however, this included only one size group (60 to 110 mm [2.4 to 4.3 in] SL) of Irish pompano (see Table 4–2). Several species exhibited differing size-specific responses to increasing freshwater inflow. Notable species that had this response included spot, striped mullet, southern flounder, spotted seatrout, and white mullet (see Table 4–2). For example, relative abundance of small spot (26-60 mm [1.0 to 2.4 in] SL) increased with increasing inflow whereas as the relative abundance of the next size group (61 to 90 mm [2.4 to 3.5 in] SL) decreased in response to increasing freshwater inflow. The smallest striped mullet size class (0 to 30 mm [0 to 1.2 in] SL) exhibited no response to freshwater inflow, but the next larger size class (31 to 45 mm [1.2 to 1.8 in] SL) strongly increased in relative abundance with increasing freshwater inflows. Small southern flounder (< 100 mm [3.9 in] SL) also increased in abundance with increasing freshwater inflows, while larger size classes (> 126 mm [5.0 in] SL) declined in abundance. In contrast, spotted seatrout had an opposite response; the relative abundance of small spotted seatrout (< 100 mm [3.9 in] SL) declined with increasing freshwater inflows, while the relative abundance of larger sizes (>210 mm [8.3 in] SL) increased. Relative abundance of several other important estuarine species, such as Atlantic croaker, bay anchovy, Atlantic weakfish, pinfish, and silverside, all declined with increasing freshwater inflows across all size groups sampled (Table 4–2) .

Freshwater pseudospecies relative abundance responses to freshwater inflows were generally strongest for lag times exceeding 180 days (see Table 4–2). This suggests that changes in relative abundance of freshwater species are more dependent on interannual variability in inflow patterns as opposed to short-term events. Changes in relative abundance of a number of estuarine pseudospecies also were most strongly related to lagged inflows exceeding 180 days, but a substantial number of pseudospecies responded most strongly to lagged inflows of 120 days or less. The shorter-term responses of these pseudospecies may indicate a more seasonal inflow effect on recruitment success that is less sensitive to interannual variability.

To help simplify our overall analyses of potential water withdrawal effects on fishes in the estuary, we subdivided the 57 species of fishes collected frequently enough by the FIM sampling program to be analyzed for inflow responses into assemblages based on habitat recorded at the time of capture, and life history characteristics (Table 4–4)(MacDonald 2009). Freshwater fishes were assigned to the five assemblages described previously (see Section 3.1.2 Freshwater Fish Species and Assemblages; Table 3–2). Estuarine and marine species were subdivided into six additional assemblages (Table 4–2). Because of ontogenetic shifts in habitat use with increasing size, some species can occupy more than one assemblage. For example, striped mullet recruiting to the estuary at sizes < 44 mm (1.7 in) SL are found almost exclusively in open water and therefore were assigned to the Open Water Small Estuarine Fishes Assemblage. However, as striped mullet grow beyond this size, they recruit to shallow nursery areas and become most abundant in estuarine marshes (MacDonald et al. 2009; Martin and Drewery 1978). Therefore, striped mullet > 44 mm (1.7 in) SL were assigned to the Estuarine Marsh Fishes Assemblage (Table 4–4). Finally, as striped mullet grow bigger than 110 mm (4.3 in) SL they again occupy more open water habitats eventually returning to the ocean to spawn. As a result, striped mullet > 100 mm (4.3 in) SL were assigned to the Marine Fishes Assemblage (Table 4–4). Because our marine and estuarine fish assemblages only contain species that were abundant in the FIM collections, assemblage species lists cannot not be considered comprehensive lists of all species present in the estuary that could be assigned to each assemblage.

Applying modeled inflow output to the regressions relating abundance responses to inflow indicates that water withdrawal could cause some substantial declines in the relative abundance of a few important freshwater pseudospecies found in the estuary (Table 4–5). Most notable of these for the worst-case scenario Full1995NN is a predicted 23% decline in the relative abundance of both small (<100 mm [3.9 in] SL) channel catfish and white catfish throughout the upper reaches of the estuary (FIM sampling zones 3 through 8). The relative abundance of larger channel catfish (150 to 275 mm [5.9 to 10.8 in] SL) is also predicted to decline nearly 43%, but only in FIM zone 4, which is located near the shifting interface between oligohaline and low mesohaline salinity habitats. Lag times associated with these catfish pseudospecies responses were generally long (>150 days) suggesting that interannual variability inflow is most important in determining relative abundance.

Under the Full1995NN Scenario, small bluegill (20 to 65 mm [0.8 to 2.6 in] SL), redear sunfish (<125 mm [4.9 in] SL), and redbreast sunfish (20 to 110 mm [0.8 to 4.3 in] SL) are also predicted to decline substantially in the transition zone between oligohaline and low mesohaline habitats (Table 4–5). Surprisingly, in FIM zone 4, larger redbreast sunfish (131 to 190mm [5.2 to 7.5 in] SL) are predicted to increase in relative abundance by 6%. Given that the response of larger redbreast sunfish is associated with a short lag time (30 days), this change in relative abundance may represent a response to instantaneous flow as opposed to longer-term salinity shifts. Lag times most strongly correlated with relative abundance of all the small sunfish

Table 4–3. Predicted changes in median center-of-abundance for the 20 pseudospecies whose distribution was influenced by freshwater inflows ($p < 0.05$, $r^2 > 0.25$). For the withdrawal model scenarios, the magnitude of the upstream (+) or downstream (-) change (Δ) in center-of-abundance is presented in km (mi) as compared to the Base1995NN Scenario predicted center-of-abundance. For regression statistics see Appendix 12.F.

Common Name	Scientific Name	Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)	r^2	Base 1995NN km (mi)	ΔFull1995NN km (mi)	ΔHalf1995PN km (mi)	ΔFull1995PN km (mi)	ΔHalf2030PS km (mi)	ΔFull2030PS km (mi)	ΔFwOR2030PS km (mi)
Channel catfish[3]	*Ictalurus punctatus*	50 to 100	0.28	101.5 (63.1)	+1.0 (0.6)	+0.3 (0.2)	+0.8 (0.5)	-1.2 (-0.7)	-0.8 (-0.5)	-0.2 (-0.1)
Redear sunfish[2]	*Lepomis microlophus*	0 to 125	0.33	53.8 (33.5)	+0.2 (0.1)	+0.1 (0.1)	+0.2 (0.1)	-0.1 (0.0)	0.0 (0.0)	+0.3 (0.2)
Naked goby[1]	*Gobiosoma bosc*	20 to 35	0.27	82.2 (51.1)	+1.6 (1.0)	+0.8 (0.5)	+1.6 (1.0)	-0.3 (0.2)	+0.6 (0.3)	+2.0 (1.2)
Atlantic bumper[2]	*Chloroscombrus chrysurus*	50 to 110	0.38	23.1 (14.3)	+0.8 (0.5)	+0.2 (0.1)	+0.6 (0.4)	-1.1 (0.7)	-0.7 (0.4)	-0.1 (0.1)
Striped mullet[1]	*Mugil cephalus*	31 to 45	0.33	77.6 (48.2)	+2.9 (1.8)	+0.8 (0.5)	+2.1 (1.3)	-0.6 (0.4)	+0.6 (0.4)	+2.8 (1.7)
Bay whiff[3]	*Citharichthyes spilopterus*	51 to 90	0.30	56.2 (34.9)	+1.7 (1.0)	+0.3 (0.2)	+1.2 (0.7)	-1.4 (0.8)	-0.6 (0.3)	+0.7 (0.4)
Clown goby[1]	*Microgobius gulosus*	0 to 28	0.27	87.3 (54.2)	+0.5 (0.3)	+0.2 (0.1)	+0.4 (0.2)	-0.5 (0.3)	-0.4 (0.2)	-0.1 (0.0)
Hogchoker[3]	*Trinectes maculatus*	20 to 45	0.32	74.9 (46.5)	+0.9 (0.6)	+0.4 (0.2)	+0.9 (0.6)	-0.9 (0.6)	-0.6 (0.4)	0.0 (0.0)
Atlantic croaker[3]	*Micropogonias undulatus*	26 to 40	0.32	98.5 (61.2)	+0.6 (0.4)	+0.2 (0.1)	+0.4 (0.3)	-0.1 (0.0)	+0.2 (0.1)	+0.6 (0.4)
Atlantic croaker[3]	*Micropogonias undulatus*	41 to 60	0.35	95.0 (59.0)	+0.7 (0.4)	+0.2 (0.1)	+0.5 (0.3)	-0.1 (0.0)	+0.2 (0.1)	+0.7 (0.4)
Atlantic croaker[3]	*Micropogonias undulatus*	61 to 85	0.47	86.1 (53.5)	+1.0 (0.6)	+0.3 (0.2)	+0.8 (0.5)	-0.3 (0.2)	+0.2 (0.1)	+0.9 (0.6)

Common Name	Scientific Name	Range of Standard Lengths (SL) for Pseudospecies Size Group (mm)	r^2	Base 1995NN km (mi)	ΔFull1995NN km (mi)	ΔHalf1995PN km (mi)	ΔFull1995PN km (mi)	ΔHalf2030PS km (mi)	ΔFull2030PS km (mi)	ΔFwOR2030PS km (mi)
Atlantic weakfish[3]	*Cynoscion regalis*	41 to 75	0.34	77.1 (47.9)	+1.1 (0.7)	+0.3 (0.2)	+0.8 (0.5)	-1.2 (0.7)	-0.7 (0.4)	0.0 (0.0)
Silver perch[1]	*Bairdiella chrysora*	0 to 30	0.59	51.9 (32.2)	+2.7 (1.6)	+0.6 (0.4)	+1.6 (1.0)	-0.9 (0.6)	+0.2 (0.1)	+2.5 (1.5)
Silver perch[1]	*Bairdiella chrysora*	31 to 55	0.73	51.8 (32.2)	+2.5 (1.6)	+0.6 (0.4)	+1.8 (1.1)	-1.2 (0.7)	+0.2 (0.1)	+2.2 (1.4)
Silver perch[1]	*Bairdiella chrysora*	56 to 85	0.64	54.8 (34.1)	+2.3 (1.4)	+0.6 (0.3)	+1.9 (1.2)	-1.9 (1.2)	-0.8 (0.5)	+1.0 (0.6)
Spot[3]	*Leiostomus xanthurus*	61 to 90	0.28	80.0 (49.7)	+1.4 (0.9)	+0.2 (0.1)	+0.9 (0.6)	-1.2 (0.7)	-0.3 (0.2)	+1.0 (0.6)
Pinfish[2]	*Lagodon rhomboides*	131 to 160	0.60	30.3 (18.8)	+1.6 (1.0)	+0.4 (0.2)	+1.3 (0.8)	-2.0 (1.2)	-1.2 (0.8)	0.0 (0.0)
White shrimp[1] *(August to November)*	(*Penaeus setiferus*)	0 to 15 POH	0.48	34.9 (21.7)	+0.8 (0.5)	+0.2 (0.1)	+0.6 (0.4)	-1.1 (0.7)	-0.8 (0.5)	-0.3 (0.2)
White shrimp[1] *(June to July)*	(*Penaeus setiferus*)	0 to 15 POH	0.66	32.6 (20.3)	+2.1 (1.3)	+0.5 (0.3)	+1.3 (0.8)	-1.2 (0.8)	0.0 (0.0)	+1.6 (1.0)
White shrimp[3]	(*Penaeus setiferus*)	4 to 11 POH	0.42	58.0 (36.0)	+1.8 (1.0)	+0.2 (0.1)	+1.2 (0.7)	-1.8 (1.0)	-1.0 (0.6)	+0.5 (0.3)

Note:
SL = standard length
POH = post orbital head width
\+ = upstream
\- = downstream
Δ = change in center-of-abundance
Gear type = [1] 23.1-m seine, [2] 183-m seine, [3] 6.1-m otter trawl

Table 4–4. Estuarine and marine fishes assemblages in the St. Johns River.

Estuarine and Marine Assemblages	**Species***	**Description**
Open Water Small Estuarine Fishes	Bay anchovy (*Anchoa mitchilli*), striped anchovy (*Anchoa hepsetus*), Atlantic silverside (*Menidia menidia*), silverside spp. (*Menidia* spp.), Atlantic menhaden (*Brevoortia tyrannus*), Atlantic bumper (*Chloroscombrus chrysurus*), juvenile striped mullet (*Mugil cephalus*) < 44 mm (1.7 in) SL, juvenile white mullet (*Mugil curema*) < 44 mm (1.7 in) SL, Atlantic thread herring (*Opisthonema oglinum*)	The most abundant assemblage in the estuary consists primarily of small species or pseudospecies (< 250 mm [9.8 in] SL) that use open water and riverine or backwater habitats. Most are planktivorous and have high ecological value as prey.
Estuarine Marsh Fishes	Mummichog, rainwater killifish (*Lucania parva*) juvenile striped mullet (*Mugil cephalus*) >44 mm (1.7 in)SL, juvenile white mullet (*Mugil curema*) >44 mm (1.7 in)SL	Mummichog were the most abundant species sampled from this assemblage. Striped and white mullet move into the estuarine marsh at sizes > 44mm (1.7 in) SL where they become epiphytic and detritivore feeders. All species have high ecological value as prey.
Estuarine Benthic Fishes	Atlantic stingray (*Dasyatis sabina*), southern flounder (*Paralichthyes lethostigma*), gulf flounder (*Paralichthyes albigutta*), hogchoker (*Trinectes maculatus*), lined sole (*Achirus lineatus*), bay whiff (*Citharichthyes spilopterus*), blackcheek tonguefish (*Symphurus plagiusa*), fringed flounder (*Etropus crossotus*), tidewater mojarra (*Eucinostomus harengulus*), naked goby (*Gobiosoma bosc*), silver jenny (*Eucinostomus gula*), darter goby (*Ctenogobius boleosoma*), freshwater goby (*Ctenogobius shufeldti*)	With the exception of Atlantic stingray and the flounders, adults in this assemblage are generally small (< 250 mm [9.8 in] SL). All are benthic and use river channel and backwater habitats. Flounder have high economic value as they are harvested both recreationally and commercially.
Sciaenid Fishes	Silver perch (*Bairdiella chrysora*), spotted seatrout (*Cynoscion nebulosus*), Atlantic weakfish (*Cynoscion regalis*), spot (*Leiostomus xanthurus*), southern kingfish (*Menticirrhus americanus*), Atlantic croaker (*Micropogonias undulatus*), red drum (*Sciaenops ocellatus*)	Sciaenidae is a family of fish commonly called drums and croakers because of the repetitive drumming sounds they make. Red drum, spotted seatrout, black drum, southern kingfish, weakfish, and croaker are some of the most sought-after recreational fishes in the estuary.
Estuarine Invertebrates	Blue crab(*Callinectes sapidus*) , white shrimp (*Penaeus setiferus*)	Both species have high economic value. For discussion see Chapter 11 Benthic Macroinvertebrates.
Marine Fishes	Crevalle jack (*Caranx hippos*), striped burrfish (*Chilomycterus schoepfi*), Irish pompano (*Diapterus auratus*), ladyfish (*Elops saurus*), pinfish (*Lagodon rhomboides*), pigfish (*Orthopristis chrysoptera*), lookdown (*Selene vomer)*, southern puffer (*Sphoeroides nephelus*), Atlantic needlefish (*Strongylura marina*), gulf pipefish (*Syngnathus scovelli*), striped mullet (*Mugil cephalus*) > 110 mm (4.3 in)SL	Members of this assemblage are considered primarily marine, although Atlantic needlefish and gulf pipefish are common in freshwater reaches of the river. Both pinfish and pigfish have high ecological value as prey.

*Assemblages include only those species with pseudospecies that were abundant enough to be analyzed for freshwater inflow responses. Species lists in each assemblage should not be considered comprehensive of all species present in the estuary that could be assigned to that assemblage. Freshwater fishes in the estuary were assigned to freshwater assemblages (see Table 3–2).

Table 4–5. Predicted percent change in median monthly and/or annual relative abundance as compared to Base1995NN Scenario for 61 pseudospecies whose abundance was strongly influenced by freshwater inflow ($p < 0.05$; $r^2 > 0.25$). Zones indicate the only FIM sampling zones for which the predicted changes are appropriate. For regression statistics see Appendix 12.F. Box and whisker plots comparing each withdrawal scenario to the Base1995NN Scenario are presented in Appendix 12.G.

Assemblages and Species	Range of Standard Lengths (SL) for Pseudospecies Size Range (mm)	FIM Zones	Period Analyzed	Temporal Response	r^2	Full1995 NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030 PS
Open Water/Riverine Large Fishes (Freshwater Assemblage)											
Channel catfish [3] (*Ictalurus punctatus*)	50 to 100	4 to 8	Sep to Jan	Monthly	0.67	-23.20%	-7.77%	-17.40%	29.94%	17.20%	-0.90%
Channel catfish [2] (*Ictalurus punctatus*)	150 to 275	4	May to Sep	Annual	0.61	-42.68%	-28.45%	-52.06%	7.03%	-21.92%	-63.82%
White catfish [3] (*Ictalurus catus*)	25 to 100	3 to 8	Sep to March	Monthly	0.69	-22.60%	-9.79%	-20.90%	24.46%	10.50%	-8.30%
White catfish [3] (*Ictalurus catus*)	101 to 200	3 to 8	Jan to Dec	Monthly	0.35	-12.80%	-4.91%	-11.30%	10.70%	4.60%	-5.70%
Open Water Small Forage Fishes (Freshwater Assemblage)											
Golden shiner [1] (*Notemigonus chrysoleuca*)	0 to 50	6 to 8	May to July	Monthly	0.27	28.40%	10.90%	25.10%	-5.16%	4.90%	26.50%
Large Sunfishes (Freshwater Assemblage)											
Bluegill [1] (*Lepomis macrochirus*)	20 to 65	3 to 4	Aug to Nov	Monthly	0.52	-32.90%	-13.95%	-30.20%	33.66%	13.60%	-16.20%
Bluegill [1] (*Lepomis macrochirus*)	20 to 65	3 to 4	Aug to Nov	Annual	0.61	-11.29%	-4.57%	-9.91%	6.03%	0.33%	-6.74%
Redbreast sunfish [1] (*Lepomis auritus*)	20 to 110	3 to 4	Jan to Dec	Annual	0.74	-9.74%	-4.71%	-8.95%	8.34%	3.83%	-3.14%
Redbreast sunfish [2] (*Lepomis auritus*)	131 to 190	4	Sep to April	Annual	0.77	6.01%	1.30%	4.59%	-11.27%	-6.42%	-2.43%
Redear sunfish [2] (*Lepomis microlophus*)	0 to 125	4	Nov to Jun	Monthly	0.38	-36.90%	-17.37%	-34.10%	39.32%	14.10%	-16.70%

Assemblages and Species	Range of Standard Lengths (SL) for Pseudospecies Size Range (mm)	FIM Zones	Period Analyzed	Temporal Response	r^2	Full1995 NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030 PS
Open Water Small Estuarine Fishes (Estuarine Assemblage)											
Atlantic thread herring [2] (*Opisthonema oglinum*)	70 to 110	1 to 2	Aug to Oct	Monthly	0.33	36.90%	4.75%	25.00%	-31.71%	-16.70%	-0.20%
Bay anchovy [3] (*Anchoa mitchilli*)	36 to 60	1 to 4	May to Jan	Annual	0.64	7.08%	2.23%	5.34%	-4.38%	-1.28%	3.78%
Striped mullet [1] (*Mugil cephalus*)	31 to 45	1 to 4	March to Jun	Annual	0.52	-9.94%	-3.75%	-8.28%	8.93%	3.90%	-3.36%
Silverside [1] (*Menidia* spp.)	41 to 55	1 to 8	Jan to Dec	Monthly	0.26	4.08%	1.36%	3.27%	-2.60%	-0.55%	2.58%
Estuarine Marsh Fishes (Estuarine Assemblage)											
Mummichog [1] (*Fundulus heteroclitus*)	0 to 34	1 to 2	Dec to Jan	Monthly	0.45	29.70%	11.47%	25.30%	-19.90%	-9.10%	8.60%
Mummichog [1] (*Fundulus heteroclitus*)	0 to 34	1 to 2	Dec to Jan	Annual	0.76	6.41%	4.24%	8.53%	0.41%	2.60%	7.20%
Rainwater killifish [1] (*Lucania parva*)	0 to 32	4 to 8	March to Jun	Monthly	0.71	19.00%	6.86%	15.20%	-11.74%	-5.40%	6.40%
White mullet [1] (*Mugil curema*)	31 to 80	1 to 2	June to July	Monthly	0.32	-17.50%	-6.31%	-15.90%	7.77%	-3.00%	-19.20%
White mullet [2] (*Mugil curema*)	100 to 130	2 to 4	Oct to Jan	Monthly	0.28	10.80%	3.36%	7.60%	-11.71%	-7.60%	-1.40%
Estuarine Benthic Fishes (Estuarine Assemblage)											
Bay whiff [2] (*Citharichthyes spilopterus*)	50 to 70	1 to 4	June to July	Annual	0.50	40.67%	5.56%	20.28%	-12.53%	-0.07%	21.64%
Bay whiff [2] (*Citharichthyes spilopterus*)	71 to 100	1 to 4	June to Sep	Annual	0.56	8.12%	3.23%	6.52%	-4.82%	-1.93%	3.99%
Clown goby [3] (*Microgobius gulosus*)	29 to 36	4 to 8	Sep to April	Monthly	0.32	10.20%	3.35%	8.40%	-8.89%	-4.20%	2.30%
Clown goby [3] (*Microgobius gulosus*)	37 to 56	4 to 8	Oct to April	Monthly	0.31	17.70%	5.37%	13.50%	-13.16%	-5.50%	4.70%
Naked goby [1] (*Gobiosoma bosc*)	20 to 35	1 to 8	Dec to April	Monthly	0.47	13.80%	4.66%	10.30%	-10.84%	-5.60%	2.40%
Naked goby [1] (*Gobiosoma bosc*)	20 to 35	1 to 4	Dec to April	Annual	0.58	8.29%	2.27%	5.74%	-7.25%	-3.84%	1.41%

Assemblages and Species	Range of Standard Lengths (SL) for Pseudospecies Size Range (mm)	FIM Zones	Period Analyzed	Temporal Response	r^2	Full1995 NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030 PS
Freshwater goby [1] (*Ctenogobius shufeldti*)	0 to 50	1 to 4	Nov to March	Monthly	0.29	-30.90%	-13.96%	-28.30%	32.63%	11.90%	-13.80%
Freshwater goby [3] (*Ctenogobius shufeldti*)	30 to 55	3 to 4	Dec to March	Monthly	0.42	-35.60%	-15.45%	-32.00%	37.82%	14.50%	-13.90%
Freshwater goby [3] (*Ctenogobius shufeldti*)	30 to 55	3 to 4	Dec to March	Annual	0.62	-42.59%	-25.04%	-42.13%	56.78%	19.76%	-19.26%
Fringed flounder [2] (*Etropus crossotus*)	50 to 85	1 to 3	Sep to Dec	Monthly	0.26	18.00%	5.33%	14.30%	-19.85%	-15.40%	-6.10%
Fringed flounder [3] (*Etropus crossotus*)	61 to 90	1 to 3	Aug to Dec	Monthly	0.28	13.20%	2.60%	8.90%	-16.73%	-16.10%	-11.50%
Gulf flounder [3] (*Paralichthyes albigutta*)	60 to 180	1 to 3	March to Sep	Monthly	0.27	40.50%	15.00%	34.80%	-19.69%	-8.90%	12.40%
Gulf flounder [2] (*Paralichthyes albigutta*)	60 to 180	1 to 3	March to Sep	Annual	0.47	27.32%	8.65%	44.27%	-11.32%	-3.49%	14.07%
Hogchoker [3] (*Trinectes maculatus*)	20 to 45	3 to 6	Sep to March	Monthly	0.37	-4.40%	-2.83%	-4.50%	3.58%	2.20%	-0.60%
Hogchoker [3] (*Trinectes maculates*)	20 to 45	1 to 4	Sep to March	Annual	0.60	-8.00%	-2.07%	-5.97%	13.28%	8.35%	3.19%
Irish pompano [2] (*Diapterus auratus*)	60 to 110	1 to 4	Oct to Dec	Annual	0.60	-10.47%	-3.96%	-9.12%	9.04%	3.21%	-4.66%
Southern flounder [3] (*Paralichthyes lethostigma*)	0 to 50	1 to 5	Feb to May	Monthly	0.35	-18.90%	-5.94%	-15.70%	2.30%	-6.10%	-22.70%
Southern flounder [3] (*Paralichthyes lethostigma*)	51 to 100	1 to 8	April to Sep	Monthly	0.27	-19.70%	-7.87%	-16.50%	10.10%	0.30%	-12.00%
Southern flounder [2] (*Paralichthyes lethostigma*)	126 to 325	1 to 3	Feb to Nov	Annual	0.55	3.64%	1.61%	3.06%	-2.68%	-1.16%	0.98%
Tidewater mojarra [2] (*Eucinostomus harengulus*)	91 to 110	4	May to Jun	Monthly	0.46	72.80%	18.65%	48.70%	-29.30%	-5.60%	28.30%

Assemblages and Species	Range of Standard Lengths (SL) for Pseudospecies Size Range (mm)	FIM Zones	Period Analyzed	Temporal Response	r^2	Full1995 NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030 PS
Sciaenid Fishes (Estuarine Assemblage)											
Atlantic croaker [2] (*Micropogonias undulatus*)	131 to 170	1	July to Sep	Monthly	0.33	17.00%	3.36%	13.30%	-10.80%	-3.30%	6.10%
Atlantic weakfish [3] (*Cynoscion regalis*)	41 to 75	1 to 8	June to Nov	Monthly	0.42	15.70%	3.34%	10.30%	-15.17%	-6.90%	1.80%
Silver perch [2] (*Bairdiella chrysora*)	80 to 100	4	Sep to Oct	Monthly	0.26	64.80%	9.13%	40.00%	-42.44%	-27.40%	-5.30%
Silver perch [1] (*Bairdiella chrysora*)	31 to 55	1 to 6	May to July	Monthly	0.42	18.60%	4.49%	13.40%	-7.08%	1.20%	17.10%
Spot [3] (*Leiostomus xanthurus*)	41 to 60	1 to 4	April to June	Monthly	0.32	-19.00%	-5.17%	-13.60%	4.52%	-2.70%	-21.30%
Spot [2] (*Leiostomus xanthurus*)	60 to 90	1 to 4	April to June	Monthly	0.38	16.20%	4.34%	12.20%	-4.25%	2.10%	16.70%
Spot [2] (*Leiostomus xanthurus*)	60 to 90	1 to 4	April to June	Annual	0.49	8.42%	2.26%	6.12%	-1.53%	1.23%	7.29%
Spot [2] (*Leiostomus xanthurus*)	91 to 120	1 to 4	May to Sep	Monthly	0.32	13.30%	2.86%	9.80%	-10.46%	-4.40%	5.40%
Spotted seatrout [1] (*Cynoscion nebulosus*)	31 to 50	1 to 6	June to Oct	Monthly	0.65	56.80%	10.71%	42.40%	-34.25%	-14.90%	23.10%
Spotted seatrout [1] *Cynoscion nebulosus*)	51 to 110	1 to 6	July to Dec	Monthly	0.39	47.90%	16.48%	36.90%	-27.57%	-12.50%	17.30%
Spotted seatrout [2] *Cynoscion nebulosus*)	201 to 325	1 to 2	Nov to March	Annual	0.58	-3.70%	-1.81%	-2.99%	4.72%	4.15%	1.70%
Estuarine Invertebrates (Estuarine Assemblage)											
Blue crab [2] (*Callinectes sapidus*)	91 to 170 CW	2 to 4	April to Oct	Monthly	0.26	13.60%	4.39%	11.50%	-6.40%	-0.80%	9.20%
Blue crab [3] *Callinectes sapidus*)	111 to 180 CW	1 to 8	June to Dec	Monthly	0.47	8.90%	2.20%	6.80%	-7.62%	-3.10%	2.60%
Marine Fishes (Estuarine Assemblage)											
Gulf pipefish [1] (*Syngnathus scovelli*)	0 to 120	4 to 8	May to Oct	Monthly	0.65	30.50%	8.27%	23.90%	-14.70%	-3.10%	15.40%
Pigfish [2] (*Orthopristis chrysoptera*)	80 to 130	2 to 4	July to Sep	Monthly	0.30	41.30%	7.83%	25.80%	-32.54%	-22.60%	-0.10%

Assemblages and Species	Range of Standard Lengths (SL) for Pseudospecies Size Range (mm)	FIM Zones	Period Analyzed	Temporal Response	r^2	Full1995 NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030 PS
Pinfish [1] *Lagodon rhomboides* ()	36 to 70	1 to 5	April to July	Monthly	0.29	7.00%	2.73%	6.60%	-4.27%	-1.80%	2.30%
Pinfish [2] (*Lagodon rhomboides*)	101 to 130	1 to 4	July to Oct	Monthly	0.33	13.40%	4.95%	12.60%	-8.49%	-2.70%	6.50%
Pinfish [2] (*Lagodon rhomboides*)	101 to 130	1 to 4	July to Oct	Annual	0.57	8.03%	3.83%	8.48%	-3.87%	-1.41%	3.72%
Pinfish [2] (*Lagodon rhomboides*)	131 to 160	1 to 4	Aug to Oct	Monthly	0.39	10.40%	4.89%	9.70%	-7.63%	-2.80%	4.40%
Pinfish [2] (*Lagodon rhomboides*)	131 to 160	1 to 4	Aug to Oct	Annual	0.63	11.73%	1.11%	7.94%	-14.29%	-9.75%	-2.42%
Southern puffer [2] (*Sphoeroides nephelus*)	70 to 170	1 to 3	Aug to Nov	Annual	0.74	16.44%	4.16%	12.30%	-9.49%	-2.99%	11.57%
Striped burrfish [2] (*Chilomycterus schoepfi*)	40 to 110	1 to 2	June to Oct	Annual	0.67	54.98%	32.86%	66.66%	-13.34%	2.36%	41.25%

Note:
Gear type = [1] 23.1-m seine, [2] 183-m seine, [3] 6.1-m otter trawl
CW = carapace width

pseudospecies were 300 days or longer, suggesting interannual flow variability is important in determining annual abundance of these pseudospecies.

Under the worst-case scenario (Full1995NN) the majority of estuarine and marine pseudospecies increased in predicted relative abundance (Table 4–5). Most notable were predicted increases in the relative abundance of small spotted seatrout (31 to 150 mm [1.2 to 5.9 in] SL, > 48%) in FIM sampling zones 1 through 6, Atlantic thread herring (0 to 110 mm [0 to 4.3 in] SL, 37%) in FIM zones 1 and 2, tidewater mojarra (91 to110 mm [3.6 to 4.3 in] SL, 73%) in FIM zone 4, and blue crab (< 180 mm [7.1 in] CW, > 8.9%) throughout the entire estuary (FIM zones 1 through 8).

Notable estuarine pseudospecies that had predicted declines in relative abundance under the Full1995NN Scenario as compared to Base1995NN included striped mullet (31 to 45mm [1.2 to 1.8 in] SL, 10%) in FIM zones 1 through 4, white mullet (31 to 80 mm [1.2 to 3.1 in] SL, 18%) in FIM zones 1 and 2, freshwater goby (< 55 mm [2.2 in] SL, 31%) in FIM zones 1 through 4, and southern flounder (<100 mm [3.9 in] SL, 18%) throughout the estuary (FIM zones 1 through 8).

Under the most likely near-term withdrawal scenario (Full1995PN), predicted changes in relative abundance for all freshwater, estuarine, and marine pseudospecies compared to the Base1995NN Scenario were generally slightly less, but still of a similar magnitude to those predicted for Full1995NN (Table 4–5). Similarities in water withdrawal effects between the Full1995NN and Full1995PN Scenarios reflect the minor contribution that flow augmentation from the USJRBP will have on freshwater inflow to the estuary. Reducing withdrawals by half (Half1995PN) generally reduced predicted changes in relative abundance of all the freshwater, estuarine, and marine pseudospecies by slightly more than half of that predicted for the Full1995PN Scenario.

Under the Half2030PS and Full2030PS Scenarios, flow augmentation from 2030 land use changes resulted in a nearly complete reversal of the relative abundance responses predicted for all freshwater, estuarine, and marine pseudospecies under the Full1995NN Scenario (Table 4–5). The only instances where predicted relative abundance responses were not reversed were for channel catfish (150 to 275 mm [5.9 to 10.8 in] SL, FIM zone 4), which were predicted to decline 22% as compared to a decline of 43% under Full1995NN, and for small white mullet (31 to 80 mm [1.2 to 3.1 in]SL, FIM zones 1 and 2), which were predicted to decline 3% as compared to an 18% decline under Full1995NN.

Adding Ocklawaha River water withdrawals (FwOR2030PS) resulted in many pseudospecies having changes in relative abundance that were similar, but not always as extreme as, those observed under the Full1995NN Scenario (Table 4–5). One notable exception under FwOR2030PS is that the predicted relative abundance of southern flounder (< 50mm [2.0 in] SL, FIM zones 1 through 5) declined 23% under FwOR2030PS as compared to a predicted decline of 19% under Full1995NN. Greater predicted declines in relative abundance under the FwOR2030PS Scenario as compared to the Full1995NN Scenario also occurred for channel catfish (150 to 275 mm [5.9 to 10.8 in] SL, FIM zone 4) and white mullet (31 to 80 mm [1.2 to 3.1 in] SL, FIM zones 1 and 2). See Appendix 12.G for box and whisker plots of predicted percent changes in abundances from the Base1995NN Scenario for all pseudospecies and model scenarios.

5 DISCUSSION

5.1 Withdrawal Effects on Fishes in Freshwater Reaches

Maximum proposed water withdrawals of 155 mgd (6.8 $m^3 s^{-1}$) appear to have relatively small effects on water levels and flows in the freshwater segments of the St. Johns River. Consequently, predicted effects of these changes in water levels and flows on most freshwater fishes that occupy these reaches are correspondingly small. This conclusion is supported by the lack of a withdrawal effect on SAV (see Chapter 9 Submersed Aquatic Vegetation) which likely has a large influence on the population dynamics of many freshwater fishes found in the river. In addition, withdrawals were found to have no effect on water quality (see Chapter 7 Biogeochemistry), wetland vegetation (see Chapter 10 Wetland Vegetation), and on the benthic invertebrates (see Chapter 11 Benthic Macroinvertebrates) which serve as important prey for many freshwater fish.

5.1.1 Effects on the Small Floodplain Fish Assemblage

Under the worst-case scenario (Full1995NN), mean water levels in the St. Johns River between Lakes Poinsett and Harney are predicted to drop approximately 5.5 cm (2 in.), and the only notable predicted effect on fishes is a nearly 10% reduction in production of small fishes on the floodplain. Although this is a chronic reduction that would potentially affect predator populations that prey on these fishes, the wide interannual variability in floodplain inundation will likely make overall effects of this reduction difficult to detect. Water withdrawals under the Full1995NN Scenario do not affect seasonality of water level fluctuations or water level recession rates and only minimally affect the intensity and duration of extreme low and high water events.

Although flow augmentation from the USJRB projects with full withdrawals (Full1995PN) increased predicted water levels and flows during low flow periods, the reduction of floodplain inundation during the wet season caused by withdrawal still reduced floodplain production of small fishes by approximately 5%. This reduction in floodplain production under Full1995PN is a direct result of withdrawal effects on the spatial extent of marshes and that were flooded for 6 months and longer. Part of this effect may be due to the aggressive intake schedule that was modeled for Taylor Creek once river flows at SR 50 exceeded 8.9 $m^3 s^{-1}$ (300 cfs), which is the proposed low flow cutoff for water withdrawals.

The HSPF hydrologic model input could also be a factor contributing to the predicted decline of small fish abundance on the floodplain under the Full199PN Scenario. In the HSPF model, Taylor Creek withdrawals were removed from Lake Poinsett, but the discharges that regulated the amount of water withdrawn from Lake Poinsett was based on measured river discharge at SR 50 (see Chapter 3 Watershed Hydrology). SR 50 is approximately 36 km (22 mi) downstream of Lake Poinsett. This spatial difference between the point where discharges that trigger withdrawals are measured and the location where withdrawals occur could affect hydrologic model predictions, particularly during low-flow conditions. Discharge measurements that trigger withdrawals need to be established for points at, or in close proximity to, the site where modeled withdrawals occur. Establishing a low-flow cutoff for discharge from Lake Poinsett as opposed to SR 50 could potentially reduce predicted withdrawal effects on the duration of floodplain inundation that in turn, would reduce the predicted effect of water withdrawals on the floodplain production of small fishes.

5.1.2 Entrainment Effects on Ichthyoplankton

Entrainment of ichthyoplankton (fish eggs and larvae) may be the greatest potential effect that withdrawals will have on freshwater fishes in the St. Johns River. Entrainment of ichthyoplankton in water intake structures will occur regardless of the withdrawal scenario considered, because the volume of water withdrawn remains constant. Obviously, entrainment will be less under the half withdrawal scenarios as compared to the full withdrawal scenarios.

We hope to complete the process of investigating potential entrainment effects on individual species by August 2012. Although our final predictions are not yet available, some general conclusions can be drawn from data analyzed to date. In both 2008 and 2009, ichthyoplankton abundance at all sites was greatest between January and August. Since few ichthyoplankton were collected after August, we conclude that entrainment at any intake site will not be an issue for withdrawals that occur from September through November. Gizzard shad and threadfin shad larvae were by far the most abundant ichthyoplankton collected in the St. Johns River. It is unlikely that larval entrainment of either species would result in significant declines in adult abundances because larvae and adults are widely distributed, both species have long spawning seasons, and, in the case of gizzard shad, they are highly fecund (Carlander 1969a). Likewise, we do not believe entrainment will have significant effects on important sunfish species given their high abundance, widespread distribution, protracted spawning season, and close association with SAV.

We have concerns about potential entrainment effects on anadromous river herrings, particularly American and hickory shad; larvae of both species were collected at every site sampled in 2008 and 2009 (see Table 4–1). All anadromous herring stocks in the United States are currently managed under plans overseen by the Atlantic States Marine Fishery Commission (ASMFC) and restoration efforts are now underway to rebuild populations in many rivers (ASFMC 1999; ASFMC 2009a; ASFMC 2009b). As a part of the management plan Florida is required to submit a habitat management plan that includes a summary of current and historical nursery habitats and list potential water resource development projects which may impact those habitats (e.g. water supply withdrawal projects; ASFMC 2009a; ASFMC 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 (ASFMC 2009a; ASFMC 2009b).

One factor affecting entrainment is the abundance of eggs and larvae at the intake location. The low total ichthyoplankton catch (all species combined) at SR 50 (see Table 4–1) appears to suggest this location may be optimal for locating an intake structure (see Appendix 12.B for site locations). However, over 11,800 American shad larvae were collected at the SR 50 site in 2008 and 2009 (see Table 4–1). This was more than 10-fold the number of American shad larvae collected at any other site. The high abundance of American shad larvae at SR 50 makes this site least desirable as a water withdrawal location. American shad eggs and larvae were present in the river from December through April (Boucher 2008), with peak occurrence generally in March. If withdrawals occur at SR 50, then they should be reduced or eliminated during these months to avoid removal of American shad eggs and larvae. Another option would be to locate the intake at the outflow of Lake Poinsett instead of SR 50, where the catch of American shad larvae was the lowest of any site sampled and where the second fewest combined numbers of fish eggs and larvae of all other species were collected (see Table 4–1).

Finally, the proposed intake location on the river near the SR 46 sampling site between Lake Monroe and Lake Harney also warrants special consideration because of potential entrainment of river herring eggs and larvae (see Appendix 12.B for site locations). Hickory shad and blueback herring eggs and larvae were most abundant at SR 46. Although American shad were more in 2008 and 2009 at SR 50 than at SR 46, historical data suggests the river reach adjacent to the SR 46 site provides important American shad spawning habitat under higher flows (Williams and Bruger 1972). During our ichthyoplankton sampling, average flows from December through April at SR 50 were low (< 8 $m^3 s^{-1}$ [282 cfs]) , a condition associated with greater American shad spawning upstream of Lake Harney (Williams and Bruger 1972). Their observation agrees with our conclusion that under the low flow conditions encountered in 2008 and 2009 larval American shad were most abundant upstream of Lake Harney at SR 50. However, during higher flows (SR 50 discharge > 20 $m^3 s^{-1}$ [706 cfs]), a condition that did not occur during our ichthyoplankton sampling window, William and Bruger (1972) found the majority of American shad spawning occurred downstream of Lake Harney near the SR 46 site.

The potential for greater American shad spawning to occur at the SR 46 site under higher flows is further supported by our analyses of the potential availability of spawning habitats under differing flow regimes (see Appendix12.C). Selection of spawning sites by American shad is related directly to flow velocity and depth (Stier and Crance 1985). Measurements we recorded when SR 50 discharges exceeded 21.7 cm^3 s (766 cfs) indicates that at these higher discharges, velocities and depths throughout the river reach between Lake Monroe and Lake Harney should be optimal for American shad spawning (see Appendix 12.C). At lower SR 50 discharges (< 6.4 m^3 s [225 cfs]) however, velocities throughout this same reach are suboptimal (see Appendix 12.C). This may explain why fewer American shad larvae were collected at SR 46 than at SR 50 in 2008 and 2009. At discharges < 6.4 m^3 s (225 cfs), optimal velocities and depths for American shad spawning occurred just upstream of the SR 50 site (see Appendix 12.C). Besides the potential for American shad spawning under higher flow conditions, the overall high abundance and diversity of all ichthyoplankton at the SR 46 site (see Table 4–1), also supports a need to minimize potential entrainment at this location.

Potential intake design features that minimize entrainment include, but are not limited to, constructing the intake to deflect passive ichthyoplankton, using wedge wire screens with small mesh sizes that minimize approach velocities, and limiting inflow velocities perpendicular to the screens (Gowan et al. 1999). Based on the swimming stamina of small fishes (Gowan et al. 1999) recommended inflow velocities should be < 0.076 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.

5.2 Withdrawal Effects on Fishes in Estuarine Reaches

5.2.1 Importance of Freshwater Inflow to Estuary Productivity

The importance of freshwater inflow in determining the productivity of river-dominated estuarine ecosystems is well known (Cross and Williams 1981; Day et al. 1989; Hobbie 2000; Mann 2000). Exported nutrient and detrital material carried by river flow are rapidly recycled within estuaries and may contribute to high fish production both within the estuary proper (Livingston 1997) and within offshore coastal waters (Elliott and Hemingway 2002; Mann 2000; Mann and Lazier 1996). Freshwater inflows also help chemically and physically structure the estuary to provide critical fish nursery habitat (Day et al. 1989; Elliott and Hemingway 2002; Livingston 1990; Livingston 1997). It is widely accepted that changes in freshwater inflow patterns due to water diversion or withdrawal could potentially have dramatic effects on an estuary and its associated fishery composition and yield (Browder and Moore 1981; Day et al. 1989; Elliott and Hemingway 2002; Hensley and Williams 2010; Livingston 1997; Mann 2000; Mann and Lazier 1996).

Although relationships between fish production and some aspect of freshwater inflows have been demonstrated for estuaries throughout the world, specific causal mechanisms remain poorly understood (Day et al. 1989; Mann 2000). Many studies have suggested that increased nutrient loading associated with higher inflows results in higher primary production (Day et al. 1989). Higher primary production increases invertebrate production and ultimately results in higher fishery yield (Day et al. 1989; Elliott and Hemingway 2002). Exogenous organic carbon inputs may also be an important energy supply to estuaries that fluctuates with varying inflow (Jassby et al. 1993).

A long-term (13-yr) study of responses of estuarine fishes in the Apalachicola River to changes in river runoff found that fish associations were strongly dependent on interannual patterns of inflow (Livingston 1997). Fish responses were primarily affected by biological interactions, such as changing predator-prey dynamics and competition. Under prolonged drought, fish abundance, biomass, species richness, and trophic diversity were reduced. These responses were related to changes in nutrient cycling associated with reduced river inflow that were ultimately reflected through the food web (Livingston 1997; Livingston et al. 1997). Other studies, however, suggest fish responses to inflow may reflect flow effects on physical habitat rather than trophic energy transfer (Kimmerer 2002). Fish abundance responses to inflow are also not always positive. In a study of fish abundance relationships to hydrologic variables within several northeastern estuaries, Rose and Summers (1992) reported instances of positive, negative, and negligible effects of freshwater flow on estuarine fish populations. Abundance of some marine and estuarine species may respond negatively to inflow simply due to the downstream displacement of habitat, downstream displacement of weaker swimming individuals, osmotic stress, or a combination of factors (Howarth et al. 2000; Kimmerer 2002; Paperno and Brodie 2004; Peebles 2002b).

Mechanisms underlying flow effects on estuarine fishes are inherently complex and may vary among estuaries, species, and even different size classes of the same species (Kimmerer 2002; Livingston 1988). Differential responses of species-specific size groups may reflect ontogenetic (developmental) shifts in salinity tolerances, habitat use, and feeding patterns (Elliott and Hemingway 2002; Greenwood et al. 2007; Livingston 1988).

In our analyses, we looked specifically at the effects of freshwater inflow on distribution and abundance of selected size classes of individual species (i.e., pseudospecies) of both freshwater and estuarine/marine species in the St. Johns River estuary. Such correlative analyses cannot reveal causal mechanisms for observed responses, but they are useful for quantifying collective effects. Given the complex suite of ecologically significant aspects of estuarine habitats affected by variations in freshwater inflows (e.g., salinity, nutrient loading, allochthonous carbon loading), it is reasonable to conclude that the regressions of distribution and relative abundance against freshwater inflow reflect causation rather than spurious correlation. As such, we believe our models are appropriate for use as tools to assess the potential effects of increased surface water withdrawals on estuarine fish populations.

The 183-m seine captures larger and more mobile subadults and adults (MacDonald et al. 2009). This accounts for the larger size classes tested for inflow responses using this gear. The 21.3-m seine and the 6.1-m otter trawl are generally more efficient at sampling smaller juveniles and subadults (< 200 mm [7.9 in] SL). Gear avoidance by larger individuals may be one reason that the numbers of large fish of most species collected by the 21.3-m seine and the 6.1-m otter trawl were insufficient to test for inflow responses. Whereas smaller species (e. g., bay anchovy, silversides) may be susceptible to capture in proportion to their abundance at all ages, other species (e.g., striped mullet, Atlantic weakfish) may only be captured in proportion to their abundance during their first or second year of life. In our analyses, we generally assume that the year class strength of an individual species is established early during their first year of life (Houde 1987) and that reductions in larval, juvenile, and subadult abundance that occur in response to water withdrawals may subsequently result in fewer individuals becoming adults. However, we also recognize there is a high degree of uncertainty associated with this assumption. Empirical data that establishes precisely when during the first year of life year class strength of an individual species is formed, along with data that relates pseudospecies relative abundance to future adult abundance, is lacking. Thus, we do not make quantitative predictions about withdrawal effects on adult abundance but focus our analyses on withdrawal effects only at the pseudospecies level.

Estuarine fish communities of the St. Johns River are similar to those reported for the Indian River Lagoon (Paperno and Brodie 2004) and for smaller estuaries along the Florida Gulf coast (MacDonald et al. 2007; MacDonald et al. 2006; Matheson Jr. et al. 2004; Matheson Jr. et al. 2005; Peebles 2002a). In general, species richness is high with the catch numerically dominated by anchovies, silversides, mojarras, gobies, striped mullet, spot, croaker, menhaden, and rainwater killifish. Similarities between fish communities occur despite rather large differences in geomorphology and runoff characteristics between the estuaries (Livingston 1990). For all of these estuaries, freshwater inflow is an important factor influencing species composition and relative abundance of fishes (MacDonald et al. 2007; MacDonald et al. 2006; Matheson Jr. et al. 2004; Matheson Jr. et al. 2005; Peebles 2002a).

5.2.2 Effects on Distribution

Approximately half of the nekton (fish) species in the St. Johns River that were abundant enough to be analyzed exhibited a significant distributional response to changes in freshwater inflow within at least one specific pseudospecies size class. For each pseudospecies that showed distributional shifts, the center-of-abundance moved upstream as inflows decreased. This response likely reflects changing salinity (MacDonald et al. 2007; MacDonald et al. 2009), however, changes in prey distribution could also be a factor (Livingston 1997; Peebles 2002b). Decreased inflows allow saltwater intrusion further upstream which tends to contract available fresher habitat and expand saltier habitat. Similar upstream movements of freshwater, estuarine, and marine pseudospecies with decreased freshwater inflow have been reported for estuaries at the mouth of the Hillsborough River (MacDonald et al. 2006), Alafia River (Matheson Jr. et al. 2004), Little Manatee River (MacDonald et al. 2007), Weeki Watchee River (Matheson Jr. et al. 2005) and Peace River (Peebles 2002a).

Distributional shifts in response to decreased inflows could be important for a species if they alter the area of overlap between its preferred salinity regime (dynamic habitat) and its preferred vegetation or bottom types (static habitat), reducing access to preferred habitats essential to growth and survival (Browder and Moore 1981; Peterson 2003). In the LSJRB estuary, distributional shifts could potentially have the greatest effect on those species or pseudospecies whose center-of-abundance is downstream of Jacksonville (river km 40 [rm 25]), where the salinity gradient from saltwater to freshwater river is most dramatic and, where extensive intertidal marshes and several tributaries (static habitat) connect to the river. Tidal creeks and salt marshes provide critical nursery habitat for many important commercial and recreational fishes that spawn offshore or in the estuary itself (Boesch and Turner 1984; Chambers 1992; Day et al. 1989). It is generally believed that the survival of the young of many species depends on their finding physiological optima first (e.g., optimal salinity) and then behaviorally searching out the appropriate life stage-dependent physical habitat, such as vegetated marshes (Peterson 2003). Salinity or other physiochemical shifts that cause forced selection of suboptimal static habitats may cause reduced growth and survival (Jassby et al. 1995).

The smallest size classes (generally < 50 mm [2.0 in] SL) of YOY spotted seatrout, striped mullet, pinfish, spot, Atlantic croaker, striped anchovy, bay anchovy, red drum, and white shrimp all use estuarine nursery habitats downstream of Jacksonville (MacDonald et al. 2009). None of these small pseudospecies that were abundant enough to be analyzed exhibited significant distributional responses to decreases in freshwater inflow. In addition, of those pseudospecies (all size classes) that did exhibit a distributional response, shifts in center-of-abundance were small (≤ 2.9 km [1.8 mi]), even under the worst-case withdrawal scenario (Full1995NN). We conclude that withdrawal scenarios analyzed here are unlikely to force distributional shifts of pseudospecies away from critical static habitat components in the downstream estuary. However, our conclusion of negligible effects on distribution only applies to the river main stem and its main backwaters, not to downstream tributaries or intertidal creeks. We lack data to conduct an analysis of potential withdrawal effects in these latter habitats, which may provide important nursery habitat for a number of estuarine species (e. g. Allen and Barker 1990; Rogers et al. 1984; Rozas and Zimmerman 2000).

5.2.3 Effects on Relative Abundance

The relative abundances of fishes in the St. Johns River estuary appear to be highly sensitive to variations in freshwater inflow. Of the 57 species abundant enough to be analyzed, 47 (82%) exhibited a significant abundance response to changes in freshwater inflow within at least one size class. In general, freshwater pseudospecies abundances in the estuary declined with decreasing inflow, marine pseudospecies abundances increased with decreasing inflow, and estuarine pseudospecies abundances either declined or increased, depending on the species and the size class. Similarly, in other Florida estuaries greater than 50% of the pseudospecies that could be tested exhibited abundance responses to freshwater inflow, and the response patterns of the three groups (freshwater, marine, and estuarine) were similar to those observed in the St. Johns River (MacDonald et al. 2007; MacDonald et al. 2006; Matheson Jr. et al. 2004; Matheson Jr. et al. 2005; Peebles 2002a; Peebles 2002b).

5.2.4 Effects on Relative Abundance of Freshwater Estuarine Species

White Catfish and Channel Catfish

Two potentially significant responses of freshwater fishes in the St. Johns River estuary to full withdrawals were predicted declines in the abundance of both YOY and juvenile white and channel catfish. Under the Full1995PN Scenario, modeled declines in relative abundance of 25 to 100 mm (1 to 4 in) SL and 101 to 200 mm (4 to 8 in) SL white catfish in FIM zones 3 through 8 were 20.9% and 11.3%, respectively. In addition, channel catfish 50 to 100 mm (2 to 4 in) SL in FIM zones 4 through 8 and 150 to 275 mm (6 to 11 in) SL in FIM zone 4 had modeled declines of 17.4% and 52.1%, respectively.

White and channel catfish have historically been two of the most heavily harvested commercial fishes in the St. Johns River (Hale et al. 1995), although net restrictions have greatly reduced this harvest in the past decade (McCarthy and Pyati 2011). There is also a large recreational fishery for both species. Neither channel nor white catfish were abundant enough in other Florida estuary surveys for analyses of inflow effects.

Both white and channel catfish are most abundant in the LSJRB in Lake George and in the river main stem and its tributaries between Lake George and Palatka (Hale et al. 1986; McCarthy and Pyati 2011). White catfish are the most abundant catfish species, generally comprising > 80% of the total catfish commercially harvested (Hale et al. 1986; Hale et al. 1995). Historically, the lowest commercial harvest of both catfish species generally occurred downstream of Palatka, partly because of lower effort in this section and partly because the river is wide and concentrations of catfish may have been harder to locate (Hale et al. 1986). The majority of the white catfish harvest occurred upstream from Palatka to Lake George, where the river is narrower and has higher flow velocities. Both white and channel catfish adults prefer deeper flowing habitats with structurally complex cover, such as logs and woody debris (McMahon and Terrell 1986), features that are characteristic of this stretch of the river and its tributaries.

Although data on the spatial distribution of white and channel catfish spawning are not available for the St. Johns River, it seems likely that most spawning occurs in the river upstream of the estuary. McLane (1955) most frequently collected newly hatched white and channel catfish 10 to 25 mm (0.3 to 1.0 in) SL in moderate currents in the main river channel and in deeper portions of connecting lakes. Other studies have also documented that larval channel catfish are most common in main river channel habitats and least common in backwater habitats (Floyd et al. 1984; Holland-Bartels and Duval 1988). White catfish larvae, however, are reported to be most common along shallow shoals adjacent to main stem habitats and are more likely to be found in backwaters (Hughes and Carlson 1986). Both catfish species have protracted spawning seasons with the majority of spawning in the St. Johns River occurring between April and September (McLane 1955).

Both white and channel catfish are tolerant of mild salinities of up to 11 ‰ (Perry and Avault 1969). In the 183-m seine and the 6.1-m otter trawl, both white and channel catfish in the St. Johns River were common in salinities up to 12‰, but were most abundant in salinities < 0.5 ‰ (MacDonald et al. 2009). Highest catches of both species of catfish were in the 6.1-m trawl. The most abundant size group of white and channel catfish captured by the otter trawl was 50 to 150 mm (2 to 5.9 in) SL. Newly hatched channel and white catfish <50 mm (2 in) SL were rare. The absence of newly hatched catfish suggests that juveniles captured in the trawl were moving into the estuary from either upstream or from tributaries, and that little successful reproduction may occur in the estuary itself. Because the 6.1-m otter trawl frequently captured other fish species < 25 mm (1 in) SL and because newly hatched white and channel catfish are reported to prefer habitats near the mid channel (McLane 1955), gear selectivity is not likely the reason for the absence of small catfish in our samples.

White and channel catfish 50 to 150 mm (2 to 5.9 in) SL in size annually first appeared in the 6.1-m otter trawl catches during September and were most abundant in trawl catches through February. Catfish of this size range are likely YOY that were spawned during the previous spring and summer spawning season (Carlander 1969a). During the months of peak catfish spawning (April to August) both white and channel catfish <150 mm (5.9 in) SL were mostly absent in the trawl catch (MacDonald et al. 2009). Inflow lag times best associated with the abundance response to inflow for catfish 50 to 150 mm (2 to 5.9 in) SL were > 150 days for channel catfish and > 300 days for white catfish. These long lag times suggest seasonal or interannual variability in freshwater inflow plays an important role in determining the fall and winter abundance of small juvenile catfish in the estuary.

Two hypotheses could help explain the relationship between small catfish abundance in the estuary and freshwater inflow. First, the number of young white and channel catfish that move into the estuary may reflect flow variation influences on reproductive success in upstream, freshwater reaches, with higher flows favoring reproduction and leading to higher upstream abundances of YOY. These higher abundances result in a higher numbers of YOY that migrate to the estuary. An alternative hypothesis is that upstream or tributary reproduction is not necessarily related to flow variability, but that higher flows over long periods simply flush a larger fraction of the young spawned upstream into the estuary. Unfortunately, data on the abundance of YOY catfish upstream of the estuary are not available to test either hypothesis. In addition, our use of continuous lagged inflows prior to capture to estimate relative abundance in this analysis does not lend itself to looking specifically at the effects of seasonal or interannual freshwater inflow variability on catfish abundance.

Data on adult movements suggest that there are no discrete subpopulations of either white or channel catfish in the St. Johns River, and that both species should be viewed as having a single river-wide population (Hale et al. 1986). Movement of juveniles into the estuary in fall and winter may reflect enhanced food availability in the estuary during the winter (Heard 1975). Tagging studies on the St. Johns River populations indicates tagged adults of both species appear to have a greater tendency to move upstream. The greatest tendency for upstream movement was reported for adult fish that were tagged in the estuary between Doctors Lake and Palatka (Hale et al. 1986). Other studies suggest that adult white catfish migrate to more brackish water in the fall and winter and then move upstream into freshwater to spawn in the spring (Heard 1975; Hughes and Carlson 1986). Heard (1975) speculated that enhanced food availability in the estuary explains this seasonal movement pattern. The 10 yrs of FIM data analyzed in this study identifies a distinct annual downstream movement pattern into the estuary of some proportion of YOY white and channel catfish in the river in fall and winter. Hale et al. (1986) did not find this seasonal component to movement, but they only tagged adult fish > 150 mm (6 in) TL.

Investigating potential relationships between freshwater flow and reproductive success of white and channel catfish upstream of the estuary, along with quantifying relationships between upstream and estuarine abundance, is critical to more fully understanding potential water withdrawal effects on these important species. In this regard, we plan to conduct a more detailed assessment of abundance of young catfish in the estuary relative to seasonal and annual freshwater inflow for more discrete time periods (e.g., during the spawning season). In addition, we plan to look more closely at spatial and temporal occurrence in the 6.1-m otter trawl catches to see if annual downstream movement patterns are discernible. Finally, 15 years of quarterly 6.1-m otter trawl data collected by the FWC from 1980 to 1995 are being obtained to determine if a relationship between freshwater flow upstream of the estuary and river-wide indices of catfish year class strength can be established.

The predicted 52% decline in larger (150 to 275 mm [5.9 to 10.8 in] SL) channel catfish relative abundance in FIM zone 4 likely reflects changing salinity within a relatively small spatial scale. FIM zone 4 encompasses the transitional zone between oligohaline habitat (0.5 ‰ to 5.0‰ salinity) and low mesohaline habitat (5.0 ‰ to 12.0 ‰ salinity). Although common in low mesohaline habitats, catfish clearly were more abundant in the lower salinity oligohaline habitats (MacDonald et al. 2009). Other freshwater pseudospecies whose predicted relative abundances were negatively affected by reduced freshwater inflow in transitional FIM zone 4 included small (< 125 mm [4.9 in.] SL) bluegill, redbreast sunfish, and redear sunfish. Although juvenile bluegill have been shown to tolerate salinity up to 10‰ (Peterson et al. 1993), most sunfish and other freshwater fishes prefer salinities < 5 ‰ (Peterson and Meador 1994).

5.2.5 Effects on Relative Abundance of Marine and Estuarine Species

Water withdrawals under all the 1995 scenarios modeled have the potential to affect all marine and estuarine pseudospecies that exhibit a significant abundance response to freshwater inflow, because modeled inflows in each scenario are lower than Base1995NN during every month of the year (see Figure 4–7). Under the 2030 scenarios, however, full withdrawals only reduced inflows from late winter through spring. Only the Half2030PS scenario did not reduce any average monthly inflow compared to the Base1995NN scenario (Figure 4–7).

The YOY of several important marine or estuarine fish species that spawn in offshore or nearshore oceanic waters recruit to the St. Johns River estuary during the late winter or early spring. These include pinfish, spot, Atlantic croaker, red drum, southern flounder, striped mullet, white mullet, and Atlantic menhaden (MacDonald et al. 2009). Similar seasonal recruitment patterns for these species have been observed in other Florida estuaries (MacDonald et al. 2007; MacDonald et al. 2006; Matheson Jr. et al. 2004; Matheson Jr. et al. 2005; Peebles 2002a; Peebles 2002b). Eggs and larvae of these fishes are carried into the estuary by complex transport mechanisms, and the effects of freshwater inflow on stratification and other chemical–physical factors likely play an important role (Day et al. 1989; Mann 2000; Norcross and Shaw 1984). Organic molecules in the water column from freshwater inflows may also provide important cues used by the larvae of offshore spawners to locate estuarine nurseries (Kristensen 1963).

Southern Flounder

Southern flounder larvae (< 100 mm [4 in] SL) captured by the 6.1-m otter trawl exhibited a significant positive response to freshwater inflow. A similar correlation of increasing abundance with increasing freshwater inflow was also indicated by the monthly catch of < 50 mm TL (2 in) southern flounder in the 21.3-m seine, but the relationship was weaker than for the otter trawl. Adult southern flounder spend most of the year in bays and estuaries and then migrate to deeper offshore waters to spawn during fall and winter (Gilbert 1986). Eggs and newly hatched larvae float at or near the surface and are carried to inshore estuarine nursery areas by wind and currents. Juveniles remain in the estuary until they reach sexual maturity and do not move into offshore waters until just prior to spawning (Gilbert 1986).

Larval southern flounder (<25 mm [1 in] SL) begin moving into the St. Johns River estuary from January through April (MacDonald et al. 2009). Unfortunately, not enough flounder < 25 mm [1 in] SL were collected to analyze this size class as an individual pseudospecies, so we could not look specifically at freshwater inflow effects on drift larval recruitment into the estuary. Southern flounder are euryhaline and were captured by the 6.1-m otter trawl in the river main stem and backwaters in nearly equal numbers throughout the entire length of the estuary. Lowest catches occurred in the first 18 km (11mi) upstream of the ocean. Other studies have shown young southern flounder prefer low salinity habitats (Gilbert 1986; Paperno and Brodie 2004; Rogers et al. 1984). Young flounder larvae feed on plankton and switch to larger invertebrates and fish as they grow (Gilbert 1986). The reduced abundance of small southern flounder in response to reduced freshwater inflows observed in this study could be related to a number of factors. They include reduced recruitment of YOY from offshore spawning grounds as a result of lower springtime flow, increased predation on YOY from marine and estuarine predators that move farther upriver as inflows decline, or a reduction in overall estuarine productivity with lower inflows that results in reduced food resources and lower YOY survival.

Striped Mullet

In our original analyses of the FIM data from 2001 through 2008, abundance of small striped mullet (< 40 mm [1.6 in] SL) in the 21.3-m seine was positively correlated with freshwater inflow. Looking at the effects of water withdrawals under the near-term Full1995PN Scenario using the 2001 to 2008 data set suggested the modeled median relative annual abundance of small mullet would decline by more than 10% below the baseline condition (Base1995NN) as a result of full withdrawals. The addition of two more years of data supported this earlier analysis. Based on the entire 2001 to 2010 data set, the modeled decline in relative median annual abundance of striped mullet (31 to 45 mm [1.2 to 1.8 in] SL) was more than 8% under the

Full1995PN Scenario. In addition, the strength of the relationship between inflow and relative abundance remained consistent. In the 2001 to 2008 dataset, freshwater inflow explained > 51% of the variability in annual relative abundance of striped mullet < 40 mm [1.6 in] SL. In the new 2001 to 2010 data set, 52% of the variability in relative annual abundance of 31 to 45 mm (1.2 to 1.8 in) SL striped mullet was explained by inflow and a PRESS r^2 statistic of 0.33 suggests the relationship was not heavily influenced by outliers. MacDonald et al. (2007) found similar correlations between freshwater inflow and relative abundance of pre-juvenile striped mullet (31 to 50 mm [1.2 to 2.0 in] SL in the Little Manatee River estuary.

Striped mullet are found worldwide in inshore and fresh waters from tropical to temperate marine-associated environments and may be the most widespread and abundant of all inshore fishes (Odum 1970). Striped mullet are economically important because they support one of the largest commercial fisheries in Florida. From 1982 through 2007, total striped mullet landings in Florida fluctuated between 544,311 to 2.3 million kg (1.2 to 5.0 million lbs) annually (FWC-FMRI 2008) The commercial harvest of striped mullet in the St. Johns River exceeds that for any other species, with annual yields typically greater than 45,359 kg (100,000 lbs) (FWC-FMRI 2009). Not included in this number are striped mullet from the St. Johns River caught by recreational anglers for food or bait. Because striped mullet are not only economically important but also are an important forage fish that plays a key role in estuarine energy flow (Collins and Stender 1989; Mahmoudi 2000), a decline in their abundance could have far reaching consequences both within and outside of the St. Johns River estuary.

Striped mullet spawning in northeast Florida occurs on the continental shelf (also known as the South Atlantic Bight [SAB]) over depths up to 1,524 m (5,000 ft) from October through February (Collins and Stender 1989; Mahmoudi 2000). Most spawning occurs in November through January with peak spawning during December (Anderson 1958; Greeley et al. 1987). Striped mullet broadcast buoyant eggs that generally hatch within a couple of days, depending on water temperature (Kuo et al. 1973). Larvae (2.7 mm [0.1 in] SL) begin feeding 5 to 8 days after hatching (Kuo et al. 1973) and reach the pre-juvenile stage (approximately 9 mm (0.4 in) SL) around 24 to 28 days post hatching (Martin and Drewery 1978). Young striped mullet begin moving toward the coast at this time (Anderson 1958; Collins and Stender 1989; Powles 1981). They may cue on organic molecules in the water column coming from coastal areas (Kristensen 1963). Pre-juvenile striped mullet begin entering estuaries in the winter in dense schools where they disperse to shallow water nursery areas (Martin and Drewery 1978). Approximately 10 to 20 mm (0.4 to 0.8 in) SL is the size at which pre-juvenile striped mullet begin to appear in the St. Johns River estuary (MacDonald et al. 2009; Tagatz 1968). Pre-juvenile striped mullet are planktivorous feeders until they reach a size of approximately 44 mm (1.7 in) SL, when they become juveniles, and begin feeding on detritus, attached algae, and bottom sediments (Blaber and Blaber 1980; Martin and Drewery 1978). This generally occurs within 90 days from hatching (Martin and Drewery 1978).

With the two additional years of FIM data, we were able to look specifically at how the abundance of newly recruited striped mullet (< 30 mm [1.2 in] SL) varied in response to freshwater inflow. We found no significant relationship between relative abundance of this size class and inflow ($p > 0.05$). This suggests that water withdrawals may have little effect on the ability of young striped mullet to access the estuary. A significant positive relationship between abundance of the next size group (31 to 45 mm [1.2 to 1.8 in] SL) of striped mullet and freshwater inflow, however, suggests inflow may influence survival as mullet transition from planktivorous to a detritivorous and/or omnivorous feeding strategy. The lag time associated with the response of 31 to 45 mm (1.2 to 1.8 in) SL striped mullet to inflow was long (210 days). This long lag time suggests that hydrologic factors affecting the estuary prior to and while larvae are recruiting to the estuary influences later survival. A possible causal mechanism could be that a reduction in overall estuarine productivity with lower freshwater inflows during the winter results in reduced food resources which leads to higher mortality. Under the Full1995PN Scenario, average winter freshwater inflows were reduced approximately 7% from the baseline condition (Base1995NN). Increased pressure by marine or estuarine predators could also play a role (Peebles 2002b). For example, the relative monthly abundance of juvenile spotted seatrout (51 to 100 mm [2.0 to 3.9 in] SL) increased substantially in the months prior to young striped mullet recruiting to the estuary in response to decreased freshwater inflow. Striped mullet are reported to be a major component of the spotted seatrout diet (Tabb 1966). Under the potential long-term 2030 Scenarios (excluding FwOR2030PS), the modeled abundance of 31to 45 mm [1.2 to 1.8 in] SL striped mullet increased. Under FwOR2030PS, modeled relative abundance of this same size-class declined by approximately 3% due to Ocklawaha River withdrawals reducing winter freshwater inflow below the Base1995NN Scenario.

While freshwater inflow explained more than 52% of the variability in the relative annual abundance of 31 to 45 mm (1.2 to 1.8 in) SL striped mullet in the St. Johns River estuary, inflow was not found to influence the relative abundance of the next larger size group (46 to 75 mm [1.8 to 3.0 in] SL). Striped mullet in the 31 to 45 mm [1.2 to 1.8 in] SL size class were only collected in the higher salinity lower reaches of the estuary in FIM zones 1 through 4 whereas striped mullet in the next size-class (46 to 75 mm [1.8 to 3.0 in] SL) were collected throughout the estuary (FIM zones 1through 8). As mullet grow, they are able to tolerate a wider range of salinities. Nordlie et al. (1982) showed the ability of young striped mullet to tolerate lower salinity increased with body size and that 20 to 29 mm (0.8 to 1.1 in) SL striped mullet could not tolerate freshwater until they grew larger than 40 mm (1.6 in) SL. Thus, striped mullet in the 46 to 75 mm (1.8 to 3.0 in) SL size class are likely to be able use the large low salinity habitats of the St. Johns River estuary, whereas striped mullet in 31 to 45 mm (1.2 to 1.8 in) SL size-class are restricted to lower estuarine reaches. The ability to use the freshwater reaches of the estuary may enhance feeding opportunities, reduce competition, and reduce predation risk from marine and estuarine predators (Miller et al. 1985; Peebles 2002b). A lack of flow response by the 46 to 75 mm (1.8 to 3.0 in) SL size-class of striped mullet suggests that factors other than freshwater inflow regulates the relative abundance of this size-class after they are able to disperse into lower salinity habitats. Other possible explanations for the lack of response to freshwater inflow includes possible gear avoidance after striped mullet reach a size >46 mm (1.8 in) SL, or dilution effects on catch rates due to dispersal throughout the estuary that mask true relative abundance trends.

Important questions about overall population effects of a potential decline in abundance of the 31 to 45 mm (1.2 to 1.8 in) SL size class striped mullet under the near-term withdrawal scenarios warrant consideration. For example, how does the abundance of 31 to 45 mm (1.2 to 1.8 in) SL striped mullet in the St. Johns River estuary during the recruitment period relate to the abundance of individuals of the next size class or to the number of individuals that eventually recruit to the adult population? Striped mullet are extremely abundant in estuaries and coastal areas all along the southeast Atlantic coast, including those adjacent to or near the mouth of the St. Johns River. How do these external populations and their progeny from other coastal areas affect the abundance of striped mullet within the St. Johns River through potential immigration into the estuary later in the year? Conversely, how does emigration of striped mullet that use the St. Johns River estuary throughout their recruitment period affect abundance of other coastal populations? Unfortunately, data to answer these important questions are unavailable and cannot be easily attained. It is the opinion of FWC biologists, however, that striped mullet that recruit within the St. Johns River estuary likely exert a major influence on the overall abundance of striped mullet along the northeast Florida coast and possibly throughout the SAB (T. MacDonald, FWC, pers. comm. 2010).

Atlantic Croaker and Spot

Atlantic croaker and spot are two of the dominant fishes in the St. Johns River estuary and in other estuaries of the southeastern United States. Their life history has been intensively studied (Hansen 1969; Kobylinski and Sheridan 1979). Young croaker and spot (< 25 mm [1 in] SL) recruit into the St. Johns River estuary during the winter and early spring (MacDonald et al. 2009). YOY croaker begin appearing in the estuary in November, with peak recruitment occurring during January and February. Spot spawn slightly later, with recruitment to the estuary starting in January and peaking in February and March. Similar seasonal recruitment patterns are reported for other Florida estuaries (Hansen 1969; Livingston et al. 1997; MacDonald et al. 2006; Paperno and Brodie 2004). There were no significant relationships ($p > 0.05$) between freshwater inflows to the St. Johns River estuary and relative abundance of new recruits (< 25 mm [1 in] SL) for either species. For Atlantic croaker, only larger individuals (131 to 170 mm [5.2 to 6.7 in] SL) in FIM zone 1 exhibited any response to inflow, with relative abundance in this zone predicted to increase with decreasing inflows. Although a relative abundance response to freshwater inflow was not found, small (26 to 85 mm [1.0 to 3.3 in] SL) croaker exhibited significant distribution responses ($p < 0.05$), with their predicted center-of-abundance moving upstream with decreasing inflows. Freshwater inflow does not appear to affect relative abundance of croaker (< 130 mm [5.1 in] SL) in the St. Johns River estuary. In contrast, in both the North Inlet estuary in South Carolina (Allen and Barker 1990) and the Apalachicola River estuary in Florida (Livingston 1997), croaker were found to be most abundant following extreme drought years, suggesting that extremes in freshwater inflows may play an important role in determining croaker abundance in these estuaries. In the absence of extreme conditions, however, croaker abundance was little influenced by average interannual variability in inflow (Allen and Barker 1990; Kobylinski and Sheridan 1979).

Spot exhibited both increasing and decreasing trends with decreasing freshwater inflows. Relative monthly abundance of spot in the 41 to 60 mm [1.6 to 2.4 in] SL size-class in the 6.1-m otter trawl decreased with decreasing inflow, while both relative monthly and annual relative abundance of spot in the 61 to 120 mm (2.4 to 4.7 in) SL size-class in the 183-m seine increased. In the 21.3-m seine, however, no relationship between inflows and the abundance of 0 to 75 mm (0 to 3.0 in) SL spot was detected. Spot < 70 mm (2.8 in) SL in the St. Johns River estuary were approximately four times more abundant in nearshore habitats sampled by the seines than in open water otter trawl samples (MacDonald et al. 2009). Similar affinities of spot for nearshore habitat have been noted for other estuaries (Kobylinski and Sheridan 1979). Given that small spot < 60 mm (2.4 in) SL in nearshore habitats did not exhibit an abundance response to inflows, relative abundance responses to water withdrawals of spot of this size-class determined from the 6.1-m otter trawls should be interpreted cautiously. On the other hand, data from the 183-m seine suggests the relative abundance of 60 to 120 mm (2.4 to 4.7 in) SL spot will increase with decreasing flows. The 183-m seine samples nearshore habitats where spot are most abundant and thus may be more reflective of a true abundance response. Abundance of spot in the lower Alafia River (Matheson Jr. et al. 2004) exhibited a polynomial relationship to freshwater inflows, suggesting an optimal flow regime, whereas the relative abundance of spot < 50 mm (2.0 in) SL in the Hillsborough River also increased with decreasing inflows. In the North Inlet estuary in South Carolina (Allen and Barker 1990) and the Apalachicola River estuary in Florida (Livingston 1997), spot, like croaker, were also found to be most abundant following extreme drought years. In the absence of extreme conditions, however, spot abundance also appeared little influenced by the average interannual variability of inflows (Allen and Barker 1990; Kobylinski and Sheridan 1979).

Pinfish

Similar to spot and croaker, pinfish (36 to 160 mm [1.4 to 6.3 in] SL) in the St. Johns River estuary had modeled increases with increasing inflows. Pinfish are an abundant omnivore that may play an important role in estuarine food web dynamics and have some recreational and commercial value (Hansen 1969). Pinfish are nearshore spawners whose young recruit into the St. Johns River estuary from December through May (MacDonald et al. 2009). Peak recruitment occurs in February. Pinfish were most abundant in nearshore habitats sampled by seines in FIM zones 1 through 4. The relative abundance of small pinfish (< 35 mm [1.4 in] SL) was not significantly influenced by freshwater inflow; however, the relative abundance of pinfish in the 36 to 160 mm (1.4 to 6.3 in) SL size class was negatively related to inflow in both the 21.3-m and 183-m seine catch. This suggests that the relative abundance of the 36 to 160 mm (1.4 to 6.3 in) SL size-class of pinfish will increase with reduced freshwater inflows. An increase in relative abundance is predicted to also be accompanied by a slight upstream shift in the center-of-abundance of larger pinfish (130 to 160 mm [5.1 to 6.3 in] SL). The relative abundance of pinfish (36 to 70 mm [1.4 to 2.8 in] SL) in the Weeki Watchee, lower Alafia, and Hillsborough rivers was also reported to be negatively correlated with freshwater inflow (MacDonald et al. 2006; Matheson Jr. et al. 2004; Matheson et al. 2005).

Spotted Seatrout and Atlantic Weakfish

The modeled relative abundance of juvenile spotted seatrout (31 to 110 mm [0.1 to 4.3 in] SL), an important estuarine recreational species, was predicted to increase with decreasing freshwater inflow. The modeled relative abundance changes were large. Under the Full1995PN Scenario, relative abundance of this size-class in the 21.3-m seine increased >36% over the Base1995NN Scenario. The predicted relative abundance of a similar species, Atlantic weakfish (41 to 75 mm [1.6 to 3.0 in] SL) in the 6.1-m otter trawl also increased with decreasing freshwater inflow. Spotted seatrout is one of the few estuarine species whose YOY and adults are both adapted to living in the estuary (Tabb 1966). Small spotted seatrout (10 to 25 mm [(0.4 to 1.0 in] SL) first began appearing in 21.2-m seine catches in May and were caught through September. Small seatrout were most abundant in backwater areas within 25 km (15 mi) of the ocean (MacDonald et al. 2009). Increasing relative abundance of small spotted seatrout (< 44 mm [1.7 in] SL) with decreasing freshwater inflow has also been reported for the lower Alafia River (Matheson et al. 2004). However, in the lower Alafia River the relative abundance of 45 to 100 mm (1.8 to 3.6 in)

SL spotted seatrout exhibited an opposing response and declined with decreasing inflow. In our analysis the abundance of larger seatrout (201 to 325 mm [7.9 to 12.8 in] SL) in FIM zones 1 and 2 was found to decrease significantly with decreasing freshwater inflows, although the predicted decline under the Full1995PN Scenario was only 3%.

<u>Bay Anchovies, Silversides, and Hogchoker</u>

Bay anchovies and silversides were the most abundant fishes collected in the St. Johns River estuary (MacDonald et al. 2009). Both are of great importance to the estuarine food web due to their trophic position, abundance, and small size (Pattillo et al. 1997). The abundance of both species is predicted to increase with decreasing freshwater inflow. Similar responses of bay anchovy to decreasing freshwater inflows were documented for the lower Alafia River (Matheson et al. 2004). Our modeled decreases in hogchoker abundance with decreasing inflow and upstream of the center-of abundance, is similar to predicted hogchoker responses to lower freshwater inflows in the lower Alafia, Alafia, Peace, and Hillsborough rivers (MacDonald et al. 2006; Matheson et al. 2004; Peebles 2002a; Peebles 2002b).

5.2.6 Summary of Effects on Estuarine Fish Distribution and Abundance

The results of our analysis indicate that any additional permanent water withdrawals from the St. Johns River will affect fish distribution and relative abundance in the lower basin estuary. Withdrawal effects will be diverse, with a high percentage of species likely being influenced at some stage of their life cycle. Our analyses indicate that species responses would fit into three groups: (1) species that would decline (e.g., white catfish), (2) species that would increase (e.g., spotted seatrout), and (3) species that would be largely unaffected (e.g., red drum). Although changes in salinity likely play an overarching role in the response of the fish community to water withdrawals, the complete suite of causative factors is likely to be complex and species specific, or even pseudospecies specific (Kimmerer 2002). Although correlation does not determine causation, the number of significant pseudospecies responses to variations in freshwater inflows found in this study are simply too great to be a function of chance. Moreover, these results are supported by studies of freshwater inflow effects on fishes in several other Florida estuaries.

Although the fish community of the LSJRB estuary appears extremely sensitive to variations in freshwater inflow, it is difficult to place a value on the effects predicted for the proposed water withdrawal scenarios. Our analyses indicate that additional withdrawals would cause a slight community shift to favor more salt-tolerant species and a small upstream movement of their centers-of-abundance. Although an argument can be made that an increase in marine or estuarine species offsets a decline in freshwater species, another argument can be made that any change from the baseline condition (Base1995NN) caused by water withdrawals is an undesirable effect. The primary support for the latter argument is that the potential long-term ecological effects of short-term multispecies shifts in relative abundance and distribution cannot be determined. For example, juvenile spot and croaker in the Apalachicola estuary were most abundant during and shortly after an extreme drought (Livingston 1997). However, numbers progressively declined in the following years. This decline was attributed to reduced nutrient loading during the drought that was not reflected by the fish community until months or even years later (Livingston 1997; Livingston et al. 1997).

The estuarine marshes, intertidal creeks, backwaters, and tributaries between downtown Jacksonville (river km 40 [river mi 25]) and the Atlantic Ocean constitute extremely valuable nursery habitat for many estuarine and marine offshore spawning species. These areas provide important structural habitat that interacts with a varying salinity regime to support fish survival, growth, and recruitment. Proposed water withdrawals in themselves do not appear to greatly affect the nursery value of these habitats; however, there is a small effect as evidenced by a slight upstream shift in the centers-of-abundance of several species along with predicted changes in abundance. Salinity increases in nursery habitats, beyond those caused by water withdrawals, could potentially have severe consequences on the habitat value of these areas. Anthropogenic actions that may result in future salinity increases in estuarine marsh, intertidal creek, backwater, and tributary habitats (e.g., channel dredging) will act cumulatively with water withdrawal effects and warrant careful consideration as to potential effects.

One potential effect of surface water withdrawals that we were not able to evaluate, concerns effects of reduced freshwater flows from the St. Johns River on the SAB. The SAB is an immense freshwater and saltwater mixing zone off the Atlantic coast (Pomeroy et al. 1993). Due to the inflows of southeastern coastal rivers, including the St. Johns River, waters of the SAB have slightly lower salinity and relatively higher nutrient concentrations than waters of the outer continental shelf. This results in high phytoplankton production in the SAB, which serves as important nursery habitat for a number of offshore spawning species (Cowen et al. 1993; Epifanio and Garvine 2001). The relative influence of St. Johns River flows on the water quality and biology of the SAB is not well known, but it is likely significant given it is the southernmost river to provide freshwater to the SAB. For a more thorough discussion of the SAB and its potential interactions with the St. Johns River, see Appendix 12.H.

Our analyses predict the potential effects of various scenarios on fishes (at the pseudospecies level) and are useful for weighing risks and benefits of water withdrawals. The lowest risk is clearly associated with the smallest deviation from the Base1995NN Scenario. If the desired goal is one of minimizing net change from existing conditions, then the withdrawal scenario with the least effect on inflows becomes the most desirable.

5.3 Summary Evaluation of the Ecological Effects of Potential Water Withdrawals on Fishes

5.4 Scoring Approach

Potential water withdrawal effects on fishes of the St. Johns River were evaluated by river segment (Figure 5–1) and were simplified by comparing effects at an assemblage level. See Chapter 2 (Comprehensive Integrated Assessment) for a more detailed discussion of the river segments. Potential withdrawal effects were quantified with respect to three metrics: (1) strength, (2) persistence, and (3) diversity. The strength of an effect (e.g., predicted changes in abundance) considers both intensity (magnitude of projected change) and spatial (aerial extent of change) components. Persistence relates to the ability of species to recover from perturbation caused by water withdrawals. Diversity relates to the total number of species within the

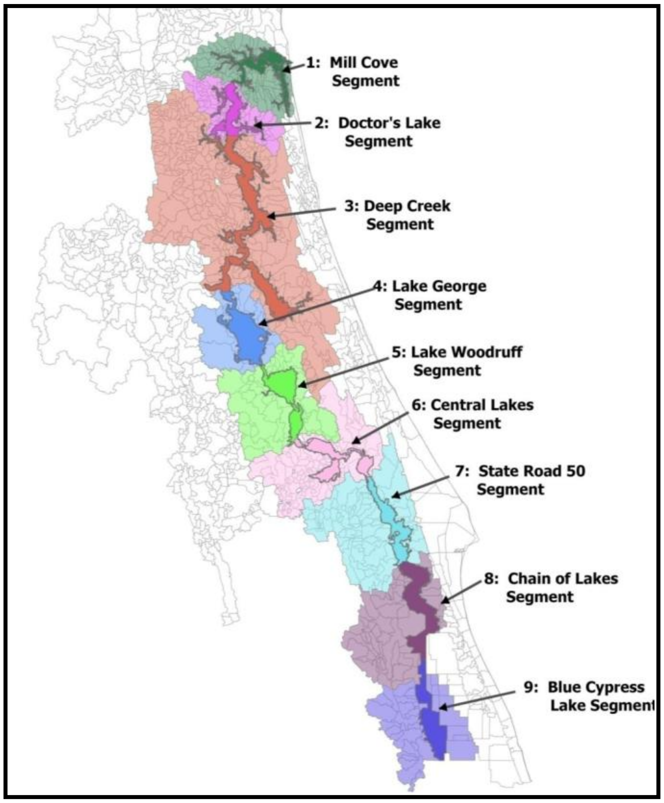

Figure 5–1. River segments developed for WSIS. See Chapter 2 (Comprehensive Integrated Assessment) for a more detailed discussion of the river segments.

assemblage affected by withdrawals. For each withdrawal scenario, individual fish assemblages were assigned a value between 1 and 3 for each metric, with 1 being the least effect due to withdrawals and 3 being the greatest effect due to withdrawals. Strength, persistence, and diversity scores for each assemblage were placed in a matrix to qualify potential cumulative effects of each withdrawal scenario as extreme, major, moderate, minor, or negligible.

To compare species responses between withdrawal scenarios and quantify changes that can be attributable to withdrawals, it is important to be able separate withdrawal effects from augmentation effects. Augmentation effects occur when a scenario increases inflow to the river over the base condition (Base1995NN) during all or part of the year. For example, predicted water levels in River Segment 7 (see Figure 5–1) are higher throughout the period modeled under the Full2030PS Scenario than under the Base1995NN Scenario due to the USJRBP and projected 2030 land use changes adding more water to the river than is removed by withdrawals. Consequently, any predicted changes in fish relative abundance between Full2030PS and Base1995N Scenarios in River Segment 7 reflect augmentation and therefore, would be characterized as negligible regardless of the magnitude of the predicted abundance change.

Separation of withdrawal and augmentation effects becomes more complicated when augmentation occurs during only part of the year. For example, net freshwater inflow to the estuary was higher under the Full2030PS Scenario than under the Base1995NN Scenario but median monthly inflow was lower during the months of December through February (see Figure 4–7). To separate withdrawal from augmentation effects we first established the direction of a predicted change (e.g., an increase or decrease in relative abundance of a pseudospecies) due solely to withdrawals by comparing the worst-case Full1995NN Scenario to Base1995NN. Freshwater inflows to the estuary were all lowest under the Full1995NN Scenario so we assume the direction of any predicted change in relative abundance from the Base1995NN Scenario was due entirely to withdrawals. Then for each scenario we only generated effect scores using those pseudospecies that had predicted relative abundance changes that were in the same direction as those predicted by comparing the Full1995NN and Base1995NN Scenarios. If predicted changes were in the opposite direction then the changes were due to augmentation and were classified as negligible regardless of the magnitude of the predicted change.

For example, monthly relative abundance of redear sunfish (0 to 125 mm [0 to 4.9 in] SL) in FIM zone 4 (River Segment 2) decreased 37% when comparing the Full1995NN Scenario to the Base1995NN Scenario (see Table 4–5). Thus, a reduction in relative abundance of this size-class of redear sunfish in this FIM zone is a predicted withdrawal effect. However, when comparing the Full2030PS Scenario to the Base1995NN Scenario, the predicted relative abundance of 0 to 125 mm (0 to 4.9 in) SL redear sunfish in FIM zone 4 increased 39%. Although a 39% increase is a relatively large change, the withdrawal effects of the Full2030PS Scenario on this pseudospecies was deemed negligible because the change in relative abundance was in the opposite direction as that indicated by comparing the Full1995NN and Base1995NN Scenarios and reflects augmentation.

In contrast, monthly relative abundance of small southern flounder (0 to 50 mm [0 to 2.0 in] SL) was predicted to decrease 19% in FIM zones 1-5 (River Segments 2 and 3) under the Full1995NN Scenario as compared to the Base1995NN Scenario (see Table 4–5). Thus, a reduction in relative abundance of this size-class of southern flounder in these FIM zones is a predicted withdrawal effect. When comparing the Full2030PS Scenario to the Base1995NN Scenario, the predicted relative abundance of (0 to 50 mm [0 to 2.0 in] SL) southern flounder declined 6%. Because this change in relative abundance was in the same direction (relative abundance decreased) as the change in abundance indicated by comparing the Full1995NN and Base1995NN Scenarios, we attributed this response to withdrawal effects under the Full2030PS Scenario.

Strength Scores for Estuary Effects (River Segments 1 to 3)

To calculate effects scores for fishes in the estuarine analyses, we used only abundance and not changes in center-of-abundance to calculate effect scores. We did not use changes in pseudospecies center-of-abundance because all predicted changes, regardless of scenario, were small (< 2.9 km [1.8 mi]).

Assemblage strength of response was determined by comparing response scores for all the individual pseudospecies in that assemblage. Strength scores for each individual pseudospecies were calculated using assigned values for both the intensity and spatial extent of the withdrawal effect. For pseudospecies with significant abundance to freshwater inflow regressions, intensity of effect was assigned a value from 0 to 3 based on the predicted percent change in median relative abundance between individual withdrawal scenarios and the Base1995NN Scenario (Table 5–1). Spatial extent scores ranged between 1 and 3, and were also assigned based upon the number of FIM sampling zones or river segments in which the changes in median abundance were predicted to occur (Table 5–1). Strength scores were not calculated for pseudospecies that had a relative abundance response that could be attributed to augmentation.

Strength scores for each pseudospecies incorporating both the intensity and spatial extent values were calculated using Equation 5–1:

$$Strength\ Score\ = \frac{(2\ X\ Intensity\ Score) + Spatial\ Extent\ Score}{3} \qquad \text{[Eq. 5–1]}$$

Strength scores were rounded to the nearest whole number. Strength scores were weighted in favor of intensity over spatial extent to prevent small abundance changes occurring over wide spatial scales from potentially inflating the final strength score.

Table 5–1. Categories used to assign intensity and spatial extent values for calculating overall strength of effect scores for individual pseudospecies. Intensity values reflect the predicted percent change in median relative abundance between individual withdrawal scenarios and Base1995NN. Spatial extent values reflect the number of FIM zones or river segments in which the abundance changes were predicted to occur.

Intensity		**Spatial Extent (Area Affected)**	
Median Relative Abundance Change from Base1995NN (%)	**Score**	**# of FIM Zones or River Segments in which Abundance Change is Predicted to Occur**	**Score**
< 2%	0	1 to 2 FIM zones	1
2% to 10%	1	3 FIM zones or 1 river segment	2
10% to 20%	2	≥ 4 FIM zones or ≥ 2 river segments	3
> 20%	3	If complete range of target taxa occurrence within FIM zones 1 to 4	3

To derive overall assemblage strength of response scores, we ranked pseudospecies or species within each assemblage as either having high or low importance (Table 5–2). High importance was assigned to species known to be important prey items (e.g., striped mullet), to species that have high recreational or commercial value (e.g., white catfish, spotted seatrout), or to species that were extremely abundant in FIM collections (e.g., bay anchovy, silversides). Final assemblage strength of response scores were weighted to favor species with high importance.

For example, if any pseudospecies designated as having high importance in an assemblage had a final strength of effect score of 3, then the entire assemblage to which that pseudospecies was assigned was given a strength score of 3. If two or more species in an assemblage designated as having low importance had strength of effect scores of 3, then the entire assemblage was also assigned a strength score of 3. If only one species designated as having low importance in an assemblage had a strength of effect score of 3, then the entire assemblage was assigned a strength score of 2. If the highest strength score of any pseudospecies in an assemblage was a 2, and either one important species or two or more less important species had a scores of 2, then the assemblage was assigned a strength of effect score of 2. If none of these criteria were met, then the assemblage was assigned a strength of effect score of 1. Species that exhibited an augmentation response were also given a strength of effect score of 1.

Table 5–2. Species (represented by pseudospecies) collected by FIM sampling designated as having high importance.

Assemblage	Species
Open Water/Riverine Large Fishes	Channel catfish (*Ictalurus punctatus*), white catfish (*Ictalurus catus*)
Open Water Small Forage Fishes	Bay anchovy (*Anchoa mitchilli*), silverside spp. (*Menidia* spp.), Striped mullet (*Mugil cephalus*) <44 mm
Large Sunfishes	Bluegill (*Lepomis macrochirus*), Redear sunfish (*Lepomis microlophus),* Redbreast sunfish (*Lepomis auritus*)
Estuarine Marshes Fishes	None
Estuarine Benthic Fishes	Southern flounder (*Paralichthyes lethostigma*)
Sciaenid Fishes	Atlantic croaker (*Micropogonias undulatus*), Spotted seatrout (*Cynoscion nebulosus*), Atlantic weakfish (*Cynoscion regalis*), Spot (*Leiostomus xanthurus*)
Estuarine Invertebrates	Blue crab (*Callinectes sapidus*
Marine Fishes	None

Strength Scores for Freshwater Effects (River Segments 4 to 8)

Because quantitative models were generally unavailable, strength effect scores for freshwater assemblages upstream of the estuary (River Segments 4 to 8) were not as straight forward to calculate as for the estuary assemblages, except for the Littoral Zone, Marsh and Floodplain Small Fishes Assemblage. This assemblage had a predictive model relating relative abundance to water levels, so strength scores could be using Equation 5–1. For the other freshwater assemblages, excluding ichthyoplankton considered under entrainment and impingement, strength scores were more subjective and score assignments were based on published responses of a relatively few species to fluctuations in hydrology. Nonetheless, given the relatively small changes in water level and flows that occurred as a result of water withdrawals, and the relatively large changes in these variables that are necessary to elicit a measurable population response (see Section 4.1.2 Effects to Freshwater Fishes Assemblages), we conclude that the strength of withdrawal effects on most freshwater fish assemblages will be small. Thus, except where noted, the strength scores for all freshwater fish assemblages for each of the withdrawal scenarios, except for the Littoral Zone, Marsh and Floodplain Small Fishes Assemblage were assigned a value of 1.

For analyses of ichthyoplankton impingement and entrainment, preliminary results suggest withdrawal effects on most species and assemblages will be minor (strength value of 1) due to the small percent of larvae in the proximity of an intake that may be potentially entrained, and the normal high natural mortality of egg and larval stages. We currently lack a predictive model of how entrainment losses will affect adult populations; however, where potential entrainment of river herring larvae may occur, potential strength effects were given a score of 3. Any potential loss of river herring eggs and larvae is considered a major detrimental effect that must be reduced to the greatest extent possible.

Persistence of Effect Scores (All River Segments)

Persistence relates to the ability of a species to rebound from possible deleterious effects of a disturbance (e.g., intense drought) and how that rebound is influenced by water withdrawals. Most fish species in the St. Johns River are highly adapted to a widely fluctuating environment. Evidence for this is provided by the fact that only one species—native striped bass—has been formally documented as being extirpated from the system in the last half century, despite ever-increasing anthropogenic stress on the river. Because of an apparent community tolerance to a widely fluctuating environment, it is not anticipated that withdrawals will have a measurable effect on the ability of fishes in the St. Johns River to repopulate after declines that may follow a disturbance. Major disturbances themselves (e.g., droughts) will also not increase as a result of withdrawals. Thus, for all withdrawal scenarios all assemblages were assigned a persistence value of 1. Whereas shifting salinity regimes in the estuary could cause a disconnect between important static and dynamic habitats of some important pseudospecies, data analyzed in this study suggest that withdrawals alone will not result in such a disconnect.

For consideration of entrainment and impingement, it is unknown how entrainment of river herring larvae may potentially affect year class strength and ultimately the abundance of adults that will return to the river to spawn in future years. Given the importance of river herrings (see Section 5.1.2 Entrainment Effects on Ichthyoplankton), and the lack of information on potential site-specific contributions of larval abundance to year class strength, we assigned a conservative persistence score of 2 to the entrainment and impingement category for all withdrawal scenarios.

Diversity Scores for Estuary Effects (River Segments 1 to 3)

Of the 57 fish species abundant enough in the FIM sampling to qualify for analyses, 47 (82%) exhibited a statistically significant abundance response to freshwater inflows at a pseudospecies level. Thus, effects of withdrawals for the estuary are judged to be highly diverse, especially for those scenarios with a persistent withdrawal effect (scenarios Full1995NN, Full1995PN, Half1995PN). All estuarine assemblages were given a diversity of effects score of 3 for these withdrawal scenarios. Diversity effects decreased for withdrawal scenarios that had an augmentation effect. Because only three pseudospecies experienced abundance changes under the Full2030PS Scenario in a direction consistent with changes observed comparing the Full1995NN and the Base1995NN Scenarios, diversity effects on this scenario were assigned a value of 1. A diversity value of 1 was also assigned to the Half2030PS Scenario. Under the FwOR2030PS Scenario, all pseudospecies changes in abundance were consistent with those observed comparing the Full1995NN and Base1995NN Scenarios, so this scenario was assigned a diversity score of 3.

Diversity Scores for Freshwater Effects (River Segments 4 to 8)

For freshwater communities upstream of the estuary (River Segments 4 to 8), diversity of effects scores were assigned either a 2 or 3 for withdrawal scenarios that resulted in a decline in either water levels or flows from the Base1995NN Scenario. Scenarios that resulted in water augmentation to the system were given a diversity score of 1. Because any withdrawals that reduce floodplain inundation below that under the Base1995NN Scenario would affect all species within the Littoral Zone, Marsh, and Small Fishes Assemblage, a diversity score of 3 was given to this assemblage for any withdrawal scenario that had this effect. Because many freshwater species—particularly economically important ones (e.g., largemouth bass, bluegill)—respond to changes in hydrology (e.g., fluctuations in year class strength, changes in growth rates), scenarios that caused a decline in water level or flow were given a diversity score of 2 for all other freshwater fish assemblages. Finally, at least 18 species of freshwater fishes in the St. Johns River (29% of the total freshwater taxa) will have eggs or larvae that may be susceptible to entrainment; thus, a diversity score of 2 was assigned for this category.

5.4.1 Summary of Ecological Effects Scores

Strength, persistence, and diversity scores were put into a three-step ordinal scale matrix to qualify potential effects as extreme, major, moderate, minor, or negligible (Table 5–3)**.**

5.5 Evaluation of Uncertainty of Analyses

5.5.1 Scoring Approach

The evaluation of uncertainty for a predicted level of effect on an assemblage is based upon an assessment of strengths of the models used to determine the predicted effect, supporting evidence from other studies for the predicted effect, and our understanding of the causal mechanisms responsible for the predicted effect.

Model Strength Scores

Model strength scores assigned to the representative estuarine assemblages were based on those pseudospecies that exhibited a statistically significant abundance response to inflow or that most strongly influenced the calculated level of effect. Model strength for each pseudospecies was calculated to provide a score between 1 (poorest model fit) and 3 (best model fit) based on the PRESS r^2 of the response regressions (Table 5–4). For estuarine pseudospecies that exhibited no statistical abundance response to freshwater inflow ($p>0.05$), model strength was also assigned a value of 3. This high score reflects strong statistical evidence that no relationships between relative abundance and freshwater inflow for these pseudospecies exists. For the freshwater fish

Table 5–3. A three-step ordinal scale for assessing the cumulative of effect of strength, persistence, and diversity scores.

Strength , Persistence	Diversity 3	2	1
3,3	3,3,3	3,3,2	3,3,1
3,2	3,2,3	3,2,2	3,2,1
2,3	2,3,3	2,3,2	2,3,1
2,2	2,2,3	2,2,2	2,2,1
3,1	3,1,3	3,1,2	3,1,1
2,1	2,1,3	2,1,2	2,1,1
1,3	1,3,3,	1,3,2	1,3,1
1,2	1,2,3	1,2,2	1,2,1
1,1	1,1,3	1,1,2	1,1,1

Level of Effect
Negligible
Minor
Moderate
Major
Extreme

assemblages upstream of the estuary where quantitative models were lacking, model strength was assigned a value of 1.

Supporting Evidence Scores

Many studies have investigated relationships between freshwater fish abundance and variations in water level and flow; however, the majority of studies have focused on effects to important recreational species, such as sunfishes and largemouth bass. Studies relating effects on non-recreational species are rare. For estuarine and saltwater species, the FIM program has published several applicable studies specifically relating to freshwater inflow to their distribution and relative abundance in Florida estuaries. Supporting evidence scores for individual fish assemblages were assigned a value from 1 (lowest support) to 3 (highest support) based on the number of studies that supported the withdrawal effects predicted in this study (Table 5–5).

Understanding of Causal Mechanisms Scores

Due to the inherent complexity of aquatic systems, most causal mechanisms responsible for observed effects of hydrologic variability on the population dynamics of freshwater fishes are poorly understood. Although several hypotheses have been proposed to explain observed

Table 5–4. Scores assigned for model strength for the uncertainty analysis

PRESS r^2	Score for Model Strength
< 0.15	1
≥ 0.15 and < 0.4	2
≥ 0.4	3

Note:
PRESS r^2 = predicted error of sum of squares coefficient of determination

Table 5–5. Scores assigned for supporting evidence for the uncertainty analysis

Number of Other Studies Supporting Effects Predicted in this Study	Score for Supporting Evidence
0	1
1 to 5	2
> 5	3

responses, rigorous scientific data that unequivocally prove that these hypotheses are correct are almost always lacking. Because complex ecosystems are not reproducible in a laboratory setting, this is a common conundrum for most fisheries-related field studies.

In estuaries, causal relationships between the responses of marine and estuarine fishes to varying freshwater inflows are also poorly understood, although consistent responses (e.g., decreased estuarine fish productivity in response to decreasing freshwater inflow) have been reported for estuaries throughout the world. Given the poor understanding of overall causal mechanisms, we generally gave a causal score of 1 for all estuarine fish responses to freshwater inflows. Causal scores for freshwater fish responses to varying water levels and flows were also generally given a score of 1, except for a few instances where we believed a higher score was warranted. These higher scores will be discussed in detail in the conclusions. For withdrawal scenarios that have consistent augmentation, we assigned a causal score of 3.

5.5.2 Summary of Uncertainty Scores

For the individual fish assemblages for which a predicted effect was quantified, model strength, supporting evidence, and understanding of causal mechanism scores were also put into a three-step ordinal matrix to qualify overall uncertainty levels: as very low, low, medium, high, or very high (Table 5–6). All withdrawal scenarios that had consistent augmentation had very low uncertainty assigned to the predicted effect. Uncertainty estimates for the fish assemblages also had to take into consideration HSPF hydrologic model uncertainty (Chapter 3. Watershed Hydrology). If HSPF hydrologic model uncertainty was higher than our uncertainty prediction for fish assemblage response within a given river segment, then HSPF hydrologic model uncertainty for that river segment was used as the uncertainty for the predicted effect.

Table 5–6. A three-step ordinal scale for assessing the cumulative of effect of predictive model strength, supporting evidence, and understanding of causal mechanisms on uncertainty.

Predictive Model, Supporting Evidence	Understanding 3	Understanding 2	Understanding 1
3,3	(3,3,3) *	(3,3,2) **	(3,3,1) **
3,2	(3,2,3) **	(3,2,2) **	(3,2,1) ***
3,1	(3,1,3) **	(3,1,2) ***	(3,1,1) ***
2,3	(2,3,3) ***	(2,3,2) ***	(2,3,1) ***
2,2	(2,2,3) ***	(2,2,2) ***	(2,2,1) ****
2,1	(2,1,3) ***	(2,1,2) ****	(2,1,1) ****
1,3	(1,3,3) ****	(1,3,2) ****	(1,3,1) ****
1,2	(1,2,3) ****	(1,2,2) ****	(1,2,1) *****
1,1	(1,1,3) ****	(1,1,2) *****	(1,1,1) *****

Uncertainty
* Very Low
** Low
*** Medium
**** High
***** Very High

5.6 Results of the Evaluation of Ecological Effects and Uncertainty by Scenario

5.6.1 Full1995NN Scenario

A summary of effects for the worst-case scenario (Full1995NN) is presented in Table 5–7. Because of the potential for substantial entrainment of river herring eggs and larvae at the SR 50 and SR 46 sites, major effects for the entrainment and impingement category were predicted for River Segments 6 and 7 and minor effects were predicted for River Segment 5. The entrainment and impingement analysis is still ongoing so ours analysis presented here is considered to be abbreviated. Because potential entrainment and impingement effects are not affected by augmentation, this category remains the same for all withdrawal scenarios evaluated (Table 5–7, Table 5–8, Table 5–9, Table 5–10, Table 5–11, and Table 5–12). Implementation of intake designs that eliminate potential entrainment would reduce the effect levels to negligible or minor (see Section 5.1.2 Entrainment Effects on Ichthyoplankton) .

Moderate effects to the Open Water/Riverine Large Fishes Assemblage in River Segments 2 and 3 for the Full1995NN Scenario (Table 5–7) reflect predicted declines in the relative abundance of white and channel catfish pseudospecies in the upper reaches of the estuary due to withdrawals (see Table 4–5). Only minor effects to this assemblage was predicted for freshwater river segments. Only minor effects to the Open Water Small Forage Fishes Assemblage and the Marsh and Floodplain Large Fishes Assemblage were predicted for the Full1995NN Scenario. Moderate effects for the Large Sunfishes Assemblage in River Segment 2 reflect predicted declines in the abundance of small bluegill and redear sunfish in this segment due to withdrawals. Moderate effects to the Littoral Zone, Marsh, and Floodplain Small Fishes Assemblage were predicted for River Segments 6 through 8 under the Full1995NN Scenario due to reduced floodplain inundation in these segments. Median relative abundance of this assemblage is predicted to decline between 5% and 10%.

5.6.2 Full1995PN and Full1995PS Scenarios

Predicted overall effects from adding the USJRBP (Full1995PN) to the full withdrawal scenario do not differ from the Full1995NN Scenario (Table 5–7). Although increased low flows from the project reduce the predicted production of small fishes on the floodplain, the magnitude of loss still warrants only a moderate effect. The USJRBP had little effect on freshwater inflows to the estuary, resulting in no change in the predicted effect of withdrawals on estuarine fish communities. It is important to reiterate that in our analysis the Full1995PN Scenario is synonymous with the Full1995PS Scenario for the estuary (River Segments 1 to 3). Freshwater inflows to the estuary did not change because of sea level rise and, therefore, no changes were predicted in the estuarine fish community response. The only difference between the Full1995PN and Full1995PS Scenarios in predicted effects was for the Littoral Zone, Marsh, and Floodplain Small Fishes Assemblage in River Segment 6. Withdrawal effects would be minor instead of moderate under the Full1995PS Scenario, because of the increased floodplain inundation in this reach caused by sea level rise. Sea level rise did not affect water levels in River Segments 7 and 8.

5.6.3 Half1995PN Scenario

Reducing water withdrawals by 50% (Half1995PN Scenario) changed predicted effects on the Littoral Zone, Marsh and Floodplain Small Fishes Assemblage from moderate to minor (Table 5–9). Surprisingly, even the Half1995PN Scenario still has a moderate effect on all estuarine fish communities. Under the Half1995PN Scenario, changes in relative abundance of most estuarine pseudospecies were approximately half of those changes observed under the Full1995PN Scenario (see Table 4–5).

5.6.4 Full2030PS Scenario

Under the Full2030PS Scenario, only four estuarine pseudospecies exhibit predicted relative abundance changes that are in the same direction as those under the Full1995NN Scenario. Thus, withdrawal effects on species diversity under the Full2030PS Scenario are low (Table 5–10). Predicted declines in the abundance of channel catfish and southern flounder (see Table 4–5) result in minor instead of negligible predicted effects for their respective assemblages. Minor effects predicted for the freshwater fish assemblages in River Segments 5 through 7 reflect the unquantified entrainment loss effects on these assemblages. Uncertainty is high, however, as we believe these effects are more likely negligible.

5.6.5 Half2030PS Scenario

The Half2030PS Scenario has the least effect on fishes of any water withdrawal scenario evaluated (Table 5–11). For the freshwater fish assemblages, effects were minor, as they were for the Full2030PS Scenario because of increased inflow to the river.

5.6.6 FwOR2030PS Scenario

The FwOR2030PS Scenario had predicted effects similar to the Full1995NN Scenario for all estuarine fish assemblages (Table 5–12). For some estuarine pseudospecies, predicted declines were even greater under FwOR2030PS Scenario (see Table 4–5). Predicted effects of the FwOR2030PS Scenario to the freshwater regions were the same as for Full2030PS Scenario, because Ocklawaha River water withdrawals do not affect the river upstream of River Segment 3.

Table 5–7. Level of effect and uncertainty for fishes of the St. Johns River for the Full1995NN Scenario. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fishes						Estuarine Fishes						
River Segment	**Entrainment/ Impingement**	**Open Water Riverine Large Fishes**	**Open Water Small Forage Fishes**	**Large Sunfishes**	**Marsh and Floodplain Large Fishes**	**Littoral Zone, Marsh, and Floodplain Small Fishes**	**Open-Water Small Estuarine Fishes**	**Estuarine Marsh Fishes**	**Estuarine Benthic Fishes**	**Sciaenid Fishes**	**Estuarine Invertebrates**	**Marine Fishes**
1		****	****	****		****	**	***	****	**	****	**
2		**	***	**		***	**	***	****	**	****	**
3		**	***	**		***	**		****	**	****	**
4		**	**	**	****	***						
5	***	**	****	****	****	***						
6	*	*****	****	****	****	***						
7	*	*****	****	****	****	***						
8		**	****	****	****	***						

Level of Effect	Uncertainty
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

Table 5–8. Level of effect and uncertainty for fishes of the St. Johns River for the Full1995PN and Full1995PS Scenarios. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fish Assemblages						Estuarine Fish Assemblages					
River Segment	**Ichthyoplankton Entrainment/ Impingement**	**Open Water/ Riverine Large Fishes**	**Open Water Small Forage Fishes**	**Large Sunfishes**	**Marsh and Floodplain Large Fishes**	**Littoral Zone, Marsh, and Floodplain Small Fishes**	**Open Water Small Estuarine Fishes**	**Estuarine Marsh Fishes**	**Estuarine Benthic Fishes**	**Sciaenid Fishes**	**Estuarine Invertebrates**	**Marine Fishes**
1		****	****	****		****	**	***	****	**	****	**
2		**	***	**		***	**	***	****	**	****	**
3		**	***	**		***	**		****	**	****	**
4		***	**	***	****	***						
5	***	***	****	****	****	***						
6	*	*****	****	****	****	***						
7	*	*****	****	****	****	***						
8		***	****	****	****	***						

Level of Effect	Uncertainty
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

Table 5–9. Level of effect and uncertainty for fishes of the St. Johns River for the Half1995PN and Half1995PS Scenarios. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fish Assemblages						Estuarine Fish Assemblages					
River Segment	Ichthyoplankton Entrainment/ Impingement	Open Water/ Riverine Large Fishes	Open Water Small Forage Fishes	Large Sunfishes	Marsh and Floodplain Large Fishes	Littoral Zone, Marsh, and Floodplain Small Fishes	Open Water Small Estuarine Fishes	Estuarine Marsh Fishes	Estuarine Benthic Fishes	Sciaenid Fishes	Estuarine Invertebrates	Marine Fishes
1		****	****	****		****	**	***	****	**	****	**
2		**	***	**		***	**	*	****	**	****	**
3		**	***	**		***	**		****	**	****	**
4		**	**	**	****	***						
5	***	**	****	****	****	***						
6	*	*****	****	****	****	***						
7	*	*****	****	****	****	***						
8		**	****	****	****	***						

Level of Effect	Uncertainty
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

Table 5–10. Level of effect and uncertainty for fishes of the St. Johns River for the Full2030PS Scenario. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fish Assemblages						Estuarine Fish Assemblages						
River Segment	**Ichthyoplankton Entrainment/ Impingement**	**Open Water/ Riverine Large Fishes**	**Open Water Small Forage Fishes**	**Large Sunfishes**	**Marsh and Floodplain Large Fishes**	**Littoral Zone, Marsh, and Floodplain Small Fishes**	**Open Water Small Estuarine Fishes**	**Estuarine Marsh Fishes**	**Estuarine Benthic Fishes**	**Sciaenid Fishes**	**Estuarine Invertebrates**	**Marine Fishes**
1		***	**	**		**	**	**	****	**	**	**
2		**	***	***		**	**	**	****	**	**	**
3		**	***	***		**	**		****	**	**	**
4		*	*	*	*	*						
5	***	**	****	****	****	****						
6	*	*****	****	****	****	****						
7	*	*****	****	****	****	****						
8		**	**	**	**	**						

Level of Effect	Uncertainty
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

Table 5–11. Level of effect and uncertainty for fishes of the St. Johns River for the Half2030PS Scenario. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fish Assemblages						Estuarine Fish Assemblages					
River Segment	**Ichthyoplankton Entrainment/ Impingement**	**Open Water/ Riverine Large Fishes**	**Open Water Small Forage Fishes**	**Large Sunfishes**	**Marsh and Floodplain Large Fishes**	**Littoral Zone, Marsh, and Floodplain Small Fishes**	**Open Water Small Estuarine Fishes**	**Estuarine Marsh Fishes**	**Estuarine Benthic Fishes**	**Sciaenid Fishes**	**Estuarine Invertebrates**	**Marine Fishes**
1		**	**	**		**	**	**	**	**	**	**
2		**	**	**		**	**	**	**	**	**	**
3		**	**	**		**	**		**	**	**	**
4		*	*	*	*	*						
5	***	*	****	****	****	****						
6	*	*****	****	****	****	****						
7	*	*****	****	****	****	****						
8		**	**	**	**	**						

Level of Effect	Uncertainty
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

Table 5–12. Level of effect and uncertainty for fishes of the St. Johns River for the FwOR2030PS Scenario. See Table 3–2 and Table 4–4 for fish assemblage descriptions.

	Freshwater Fishes						Estuarine Fishes					
River Region	**Ichthyoplankton Entrainment/ Impingement**	**Open Water/ Riverine Large Fishes**	**Open Water Small Forage Fishes**	**Large Sunfishes**	**Marsh and Floodplain Large Fishes**	**Littoral Zone, Marsh, and Floodplain Small Fishes**	**Open Water Small Estuarine Fishes**	**Estuarine Marsh Fishes**	**Estuarine Benthic Fishes**	**Sciaenid Fishes**	**Estuarine Invertebrates**	**Marine Fishes**
1		****	****	****		****	**	**	****	**	****	**
2		**	***	**		***	**	**	****	**	****	**
3		**	***	***		***	**		****	**	****	**
4		*	*	*	*	*						
5	***	**	****	****	****	****						
6	*	*****	****	****	****	****						
7	*	*****	****	****	****	****						
8		**	**	**	**	**						

Level of Effect	**Uncertainty**
Negligible	* Very low
Minor	** Low
Moderate	*** Medium
Major	**** High
Extreme	***** Very high

Cross-hatching indicates abbreviated analysis.

6 CONCLUSIONS

6.1 Ranking of Scenarios by Levels of Effects

Ranking of the predicted withdrawal effects on fish assemblages of the St. Johns River indicate that the least overall effect will occur under the Half2030PS scenario (see Table 5–11 and Table 6–1). The next least significant predicted effects are seen under the Full2030PS scenario. Although the FwOR2030PS scenario will have negligible effects on the freshwater segments of the river (equivalent to Full2030PS), the effects of this scenario on the estuary will exceed predicted effects under Half1995PN. The scenarios with the greatest predicted effects on all river fish assemblages are Full1995PN followed by Full1995NN.

Our results suggest that under near-term conditions, the Half1995PN withdrawal scenario will have the least effect on St. Johns River fish assemblages. Even under this withdrawal scenario, however, there will be a moderate loss of small fish production on the floodplain between Lakes Poinsett and Monroe. Predicted effects to the estuary include a moderate reduction in relative abundance of some important freshwater pseudospecies concurrent with an increase in the relative abundance of estuarine and marine pseudospecies. Wide interannual variability in freshwater inflows to the river, however, may make these community shifts difficult to detect. Increasing withdrawals to the full level of 155 mgd (6.8 m^3 s^{-1}) will further exacerbate these effects. Future 2030 land use changes that increase water levels and flows in the river will negate any water withdrawal effects all along the river, with the exception of the FwOr2030PS scenario, which will have moderate effects on the estuary

Table 6–1. Ranking of the five water withdrawal scenarios from least (1) to greatest (6) predicted effects on freshwater and estuarine fish assemblages in the St. Johns River.

Scenario	Ranking of Predicted Effects to Freshwater Fish Assemblages (River Segments 4 to 8)	Ranking of Predicted Effects to Estuarine Fish Assemblages (River Segments 1 to 3)
Half2030PS	1	1
Full2030PS	2.5	2
Half1995PN	4	3
FwOR2030PS	2.5	4
Full1995PN	5	5
Full1995NN	6	6

6.2 Summary Conclusions and Recommendations for Reducing Water Withdrawal Effects on Fishes

Based on our analyses of the effects of potential water withdrawals on fishes in the St. Johns River, we make the following summary conclusions and provide recommendations for reducing potential water withdrawal effects:

- Potential egg and larval fish entrainment due to surface water withdrawals can be minimized by appropriately locating and carefully designing intake structures. Although the analyses of potential entrainment effects is only partially completed, results analyzed to date indicate that gizzard shad, threadfin shad, sunfish (*Lepomis* spp.), and goby are the most abundant ichthyofauna in the river and the most likely to be entrained.
- Due to the high abundance of American shad eggs and larvae at SR 50, we recommend withdrawals not be taken at this location. If the SR 50 site is selected, we recommend withdrawals be curtailed during December through April. In addition, an intake located on the river near the SR 46 warrants special design considerations to minimize effects of potential entrainment of all river herring eggs and larvae. In our analysis, hickory shad and blueback herring eggs and larvae were most abundant at this site. Historical data suggest that under higher flows than we observed, American shad eggs and larvae will also be abundant at this location.
- Water withdrawals will likely not negatively effect spawning migrations or the availability of suitable spawning habitats for migratory American shad, hickory shad, or blueback herring. Suitable spawning habitat appears widely available. Protection from withdrawal effects can be provided primarily by a low flow cutoff level that stops withdrawals when discharges at SR 50 fall below 8.5 $m^3 s^{-1}$ (300 cfs).
- Water withdrawals cause relatively small drops in water levels and are unlikely to affect valuable sport fishes such as largemouth bass, bluegill, redear sunfish, and black crappie in freshwater river reaches. Abundance of these species is more likely related to the wide natural interannual variability in water levels and its effects on year -class strength formation, and the presence or absence and density of emergent vegetation and SAV.
- The Full1995PN Scenario results in an average 4% to 5% annual decline in the maximum densities of small forage fishes produced on the floodplain between Lakes Poinsett and Monroe as compared to the Base1995NN Scenario. Under the Half1995PN Scenario, reductions in maximum small fish densities compared to the Base1995NN Scenario were generally less than 2%.
- To reduce potential HSPF model induced withdrawal effects on the production of small fishes on the floodplain under the Full1995PN Scenario, we recommend in the future the District remodel the Taylor Creek withdrawals using SR 520, instead of SR 50 as the low flow cutoff location. Cutoff discharges will have to be reduced accordingly. Based on our calculations, a discharge of 8.9 $m^3 s^{-1}$ (300 cfs) at SR 50 equates to a discharge of approximately 6.8 $m^3 s^{-1}$ (240 cfs) at SR 520. In addition, the District should consider staging withdrawals so that a lower percentage the total flow at SR 520 is withdrawn during medium flow conditions (19.5 $m^3 s^{-1}$ [690 cfs] at SR 50). Under the current modeled Full1995PN Scenario, when discharges at SR 50 were between 10.2 $m^3 s^{-1}$ (360 cfs) and 17 $m^3 s^{-1}$ (600 cfs), at least 18% of the flow passing downstream of SR 520 was withdrawn for water supply. At a maximum, 28% of the flow at SR 520 was withdrawn when discharges at SR 50 were around 11.3 $m^3 s^{-1}$ (400 cfs). Less aggressive withdrawals at flows that are just starting to cause water levels to encroach on the floodplain will likely lessen the effects of these withdrawals on floodplain production of small forage fishes.

- Under the potential long-term Full2030PS Scenario, augmentation effects of 2030 land use changes offset withdrawal effects on water levels. Therefore, there will be negligible withdrawal effects on freshwater fishes under this scenario.
- Under all scenarios, water withdrawals will have a negligible effect on the spatial coverage of the various salinity habitat blocks as compared to the Base1995NN Scenario.
- Estuarine fish communities appear to be very sensitive to variability in total freshwater inflow to the estuary. Of the 57 estuarine fish species abundant enough to qualify for analyses, 47 (82%) exhibited a statistically significant abundance response at a pseudospecies (size class) level to freshwater inflow.
- Under the potential near-term Full1995PN Scenario, predicted declines in the abundance of YOY and juvenile white and channel catfish in the estuary due to lower freshwater inflows are potentially significant. Relationships between flow and reproductive success of white catfish and channel catfish upstream of the estuary, relationships between upstream and estuarine abundance of YOY, and relationships between YOY abundance in the estuary and future abundance of adults in the population are unclear.
- Relative abundance of juvenile southern flounder in the estuary was negatively affected by withdrawals under the Full1995PN Scenario. Predicted declines under this scenario were approximately 16%. Potential effects of withdrawals on the future abundance of adults in the population are unclear.
- Moderate effects to all estuary fish assemblages were predicted for water withdrawal scenarios that did not have an augmentation effect. The potential long-term Half2030PS Scenario was the withdrawal scenario that had the least effect of St. Johns River fishes as compared to the Base1995NN Scenario. The Half1995PN Scenario had the least effects on fishes of all potential short-term scenarios considered.

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8 APPENDICES