CHAPTER 2. COMPREHENSIVE INTEGRATED ASSESSMENT

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ACRONYMS, ABBREVIATIONS, AND CONVERSION FACTORS

BEBR	Bureau of Economic and Business Research
BOD	Biological oxygen demand
CE-Qual-ICM	A mechanistic water quality model
CI	Confidence Interval
DM	Deep marsh
DOC	Dissolved organic carbon
EFDC	Environmental Fluid Dynamics Code
EPA	Environmental Protection Agency
FDEP	Florida Department of Environmental Protection
FWC	Florida Fish and Wildlife Conservation Commission
HH	Hydric hammock
HS	Hardwood swamp
HSPF	Hydrologic Simulation Program–FORTRAN
IPCC	International Panel on Climate Change
MFL	Minimum flows and levels
mgd	Million gallons per day
NO_x	Nitrate and nitrite
NR	Not ranked
NRC	National Research Council
PLRG	Pollution load reduction goals
PM	Predictive model
PSU	Practical Salinity Units
RMSE	Root mean square errors
RK	River kilometer
SAV	Submersed aquatic vegetation
SE	Supporting evidence
SFWMD	South Florida Water Management District
SJRWMD	St. Johns River Water Management District
SLR	Sea Level Rise
SM	Shallow marsh
SR	State Road
SWFWMD	Southwest Florida Water Management District
SS	Shrub swamp
TMDL	Total maximum daily load
TN	Total nitrogen
TP	Total phosphorus
TS	Transitional shrub
UM	Understanding of the mechanism
USJRB	Upper St. Johns River basin
UWP	Upper wet prairie
WP	Wet prairie
WSIS	Water Supply Impact Study

1 INTRODUCTION

This chapter presents an integrated and comprehensive assessment of the potential effects of surface water withdrawals on the hydrology, hydrodynamics, and ecology of the main stem of the St. Johns River. It discusses the general methods employed by separate work groups, summarizes the findings for each work group, and presents findings derived from interdisciplinary considerations. It places the work in the context of global and regional water supply issues and discusses how the work can be applied in the development of alternative water supplies.

1.1 GLOBAL AND REGIONAL WATER SUPPLY ISSUES

The Earth is rich in water, with an estimated total volume of approximately 1.4 billion km^3 (Wetzel 2001), but freshwater is fast becoming a limiting natural resource for human populations (UNESCO 2006). Part of the reason is that only about 2.5% of earth's water is fresh water and more than 90% of this freshwater is stored in glaciers or in deep groundwater (Wetzel 2001). Still, the remaining volume of circulating freshwater, estimated to be 110,000-120,000 km^3 (UNESCO 2006; Falkenmark and Rockstrom 2010), is much greater than the estimated human demand (about 7,452 $km^3\ y^{-1}$; Hoekstra and Chapagin 2007). The annual discharge of the world's rivers (ca. 40,000 - 45,500 $km^3\ y^{-1}$; Postel et al. 1996; Oki and Kanae 2006), a better measure of the amount of water available for human use without mining groundwater reserves, also far exceeds the total human demand. Thus, the primary cause of freshwater shortages is not an insufficient volume of circulating freshwater but, rather, the uneven distribution of water in space and in time.

Total global runoff (40,700 $km^3\ y^{-1}$; Postel et al. 1996) is largely inaccessible either due to its location with respect to population centers (about 19% of global annual runoff is geographically inaccessible; Postel et al. 1996) or its occurrence during flood flows when a large fraction cannot be captured and used (about 73% is temporally inaccessible; Postel et al. 1996). Wide variation in rainfall creates deserts and rainforests, droughts and floods. When the spatial and temporal variation of water supplies is considered, only about 12,500 $km^3\ y^{-1}$ of global freshwater runoff is accessible (Postel et al. 1996). Population growth, pollution, and the need to reserve water for natural systems increase the pressures on freshwater resources (UNESCO 2006).

Compared to many regions of the globe, Florida is replete with freshwater resources including approximately 7,800 lakes (Brenner et al. 1990), 1,700 rivers and streams (Nordlie 1990), and vast areas of wetland that once covered more than half of the state (Ewel 1990). Its average annual rainfall of 135 cm y^{-1} (53 in y^{-1}) is much higher than the national average of 76 cm y^{-1} (30 in. y^{-1}) and is exceeded only by Alabama within the continental United States (Henry 1998). Florida's groundwater resources exceed that of any other state with an estimated volume greater than $3.785 \times 10^{12}\ m^3$ (1×10^{15} gallons) of freshwater: about 1/5 the volume of water in the five Great Lakes (Berndt et al. 1998).

Notwithstanding Florida's abundance of water resources, water management is still required to ensure their sustainable use for several reasons. First, although rainfall is high, the rate of evapotranspiration is also high. In fact, annual potential evapotranspiration nearly equals the average annual rainfall. It is only outflow, the difference between rainfall and evapotranspiration, that is available for aquifer recharge (infiltration) and surface water flows (runoff). Within the boundaries of the St. Johns River Water Management District (SJRWMD), over a recent 12-year period (1995–2006) when cumulative net rainfall was 1,642 cm (646.5 in.), cumulative outflow totaled only 45 cm (17.7 in.), or 2.7% of the net rainfall (Chapter 5. River Hydrodynamics Calibration). Second, the water requirement of Florida's aquatic, wetland, and estuarine ecosystems is high. Its thousands of lakes, more than 12,000 miles of waterways, more than 300 artesian springs, hundreds of thousands of acres of wetlands, and extensive coastal lagoons and estuaries require a large fraction of Florida's outflow to maintain ecosystem goods and services. Indeed, under low-flow conditions, the Southwest Florida Water Management District concluded that as much as 95% of the flow in some rivers must be reserved in order to prevent significant environmental harm (Rules of the Southwest Florida Water Management District, Chapter 40 D-8, Water levels and rates of flow). Third, the distribution of rainfall is not uniform over time or space. The temporal patterns of rainfall and human demands for water often are in opposition. The human requirement is high when rainfall has been low and when, consequently, lakes, streams, and aquifers are at low levels. Conversely, the human requirement is low when rainfall has been high and water resources are near their maxima. Indeed, during periods of high rainfall, human concerns may turn from water supply to flood protection. Water resources also are not uniform over space. Some of Florida's largest rivers and groundwater reserves are distant from the largest population centers. Fourth, the human requirement for water is large. The total average flow of 28 of Florida's major rivers is approximately 2300 $m^3 s^{-1}$ (52,496 mgd; Nordlie 1990) while, in 2005, human freshwater withdrawals totaled 301 $m^3 s^{-1}$ (6,873 MGD; Marella 2009), 13 % of the average flow carried by all these rivers. Fifth, it is not feasible to capture a large fraction of the water flowing in rivers and streams during peak flows. Consequently, this water has flowed to the sea by the time irrigation requirements are again increasing. These factors are salient reasons that water management is required to protect and use Florida's water resources.

1.2 Water Management in Florida

Florida has been a national leader in water management since passage of the Water Resources Act (373 F.S.) in 1972. The State Water Resource Plan (373.012-373.200, F.S.) describes Florida's water policy. The overarching goals of Florida's water policy are to realize the full beneficial use of water resources and to ensure their sustainability (373.016 (1), (2), F.S.). It directs the Florida Department of Environmental Protection (FDEP) and the five water management districts created by the act to "promote the availability of sufficient water for all existing and future reasonable-beneficial uses and natural systems." Juxtaposed with its mandate to promote water use, the act requires the preservation of water quality, natural resources, fish, and wildlife. The Water Resources Act provides an important regulatory tool for preserving natural systems: the statutory requirement and authority for water management districts to set minimum flows and levels (MFLs) for surface and ground waters (373.042, F.S). MFLs define the limits at which further water withdrawals would cause significant harm to the water resources or ecology of an area. Another important regulatory tool is the ability to reserve from permitted use the water needed for the protection of fish and wildlife, public health, and public safety (Water Resource Implementation Rule; Ch. 62-40, F.A.C.). Thus, water withdrawals in Florida can be constrained in order to protect water quality and natural systems (Figure 1-1). Protection of natural systems requires an understanding of their water requirements and this presents a considerable scientific challenge. Indeed, the need to determine the water requirements of natural systems is one of the primary reasons that scientific research must be an integral part of sound water management.

Florida's State Water Resource Plan requires that each district develop a District Water Management Plan that includes a water supply assessment and a water supply plan. The St. Johns River Water Management District (SJRWMD) has completed a water supply assessment (SJRWMD 2006a) and a water supply plan (SJRWMD 2006b). These planning efforts indicated that growth in water demand from 1995 to 2025 could not be met solely by groundwater in central portions of SJRWMD. Doing so would create a reasonable likelihood that MFLs would be violated and that there would be unacceptable impacts to groundwater quality, native vegetation, lakes, and springs.

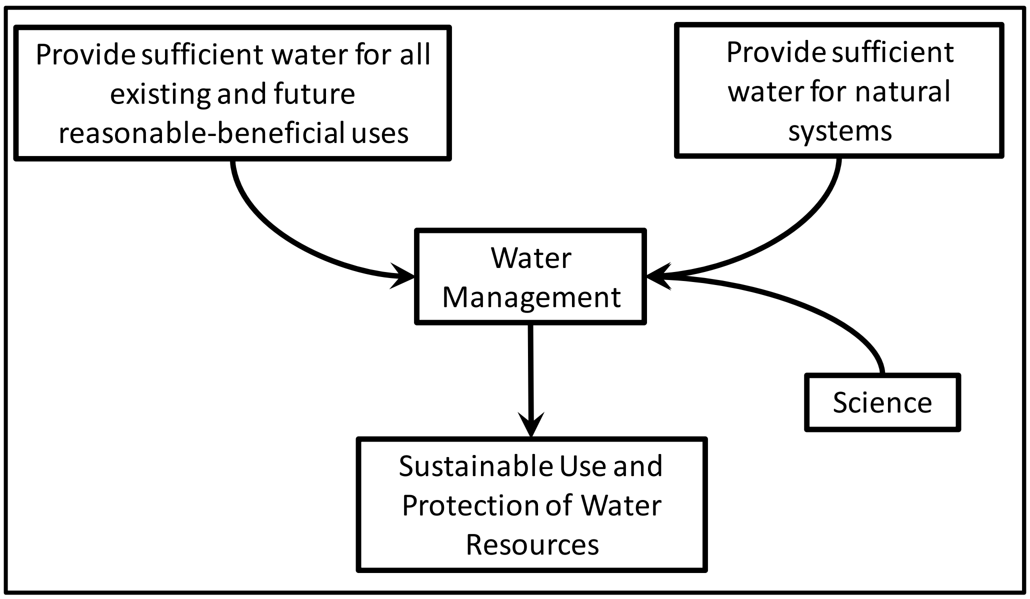

Figure 1-1. The State Water Resource Plan (373.012-373.200 F.S.) directs the water management districts to provide sufficient water for all reasonable-beneficial uses and to provide sufficient water for natural systems. Scientific research is necessary to determine the water requirements of natural systems. The goal of state policy is to provide for the sustainable use and protection of water resources.

1.3 The Need for Alternative Water Sources

The South Florida Water Management District (SFWMD), the Southwest Florida Water Management District (SWFWMD), and SJRWMD jointly recognized the potential for harm to water resources in central Florida associated with continued reliance on groundwater to meet the growing human water needs. Consequently, they produced the Recommended Action Plan for the Central Florida Coordination Area (SFWMD, SWFWMD, SJRWMD, 2006). (The Central Florida Coordination Area encompasses the water utility service areas in central Florida where the boundaries of the three districts meet.) The Action Plan capped groundwater use at the 2013 demand in order to prevent harm to water resources and natural systems of the region. The three districts agreed that alternative water supplies (AWS) would be needed to meet water demands above the 2013 level.

SJRWMD's Water Supply Plan (SJRWMD 2006a) identified many potential AWS sources. Among these sources were surface waters from the Ocklawaha River and the St. Johns River. Hall (2005), based on hydrologic modeling, estimated that an annual average of 4.69 $m^3 s^{-1}$ (107 mgd) could be withdrawn from either Rodman Reservoir or the Ocklawaha river channel without causing unacceptable environmental harm to the lower Ocklawaha River. In order to estimate the volume of water potentially available from the St. Johns River, SJRWMD used MFLs adopted for three locations: the St. Johns River at State Road 44 (SR44) near DeLand (Mace 2006), the St. Johns River at SR 50 (Mace 2007a), and Lake Monroe (Mace 2007b). Hydrologic modeling indicated that a constant withdrawal of 6.79 $m^3 s^{-1}$ (155 mgd) would not cause violations of the MFLs at DeLand (Robison 2004).

The SJRWMD Governing Board subsequently determined that a comprehensive scientific analysis was necessary to ensure that withdrawal of surface water from the Ocklawaha and St. Johns rivers would not cause unacceptable environmental effects in the St. Johns River. Consequently, the board approved this project: The Water Supply Impact Study (WSIS). Further, to ensure the scientific integrity of the work, the board approved a multi-year contract with the National Academies providing for external peer review of the work by the National Research Council (NRC) (Table 1-1). An overview of the St. Johns River Water Management District is provided in Appendix 2-A.

Table 1-1. Members of the NRC Committee to Review the St. Johns River Water Supply Impact Study and NRC staff.

Peer Review Committee Members	Affiliation
Patrick L. Brezonik, Chair	University of Minnesota, Minneapolis
M. Siobhan Fennessy	Kenyon College, Gambier, Ohio
Ben R. Hodges	University of Texas, Austin
James R. Karr	University of Washington, Seattle
Mark S. Peterson	University of Southern Mississippi, Ocean Springs
James L. Pinckney	University of South Carolina, Columbia
Jorge I. Restrepo	Florida Atlantic University, Boca Raton
Roland C. Steiner	Washington Suburban Sanitary Commission, Laurel Maryland
J. Court Stevenson	University of Maryland, Cambridge
NRC Staff	Position
Laura J. Ehlers	Study Director
Stephanie E. Johnson	Interim Study Director (Feb. 2009 – June 2009)
Michael J. Stoever	Project Assistant

1.4 THE WATER SUPPLY IMPACT STUDY (WSIS)

The goal of the WSIS was to provide a comprehensive and scientifically-rigorous analysis of the potential environmental effects to the St. Johns River associated with surface water withdrawals as high as 11.48 $m^3 s^{-1}$ (262 mgd). This was a scientifically complex endeavor. Of all the environmental factors controlling the ecological characteristics of freshwater ecosystems, the water budget is salient. The water budget controls variations in flow rates, depth, water retention time, and water flow velocity. These hydrologic characteristics, in turn, are primary factors influencing water quality, primary producer community structure and function, and the species composition of the flora and fauna. The water budget also is fundamental for estuaries. The inflow of freshwater to the estuary carries nutrients essential to estuarine productivity and the freshwater dilutes salts in inflowing marine waters creating the salinity gradient that is essential to the biological character of estuaries. For a river such as the St. Johns, reduction of the water budget has the potential to affect virtually every major component of the ecosystem downstream of the reduction.

In order to address the diversity of potential environmental effects, project personnel were organized into eight work groups: hydrologic and hydrodynamic modeling, biogeochemistry, plankton, benthos, submersed aquatic vegetation, fish, wetlands and wetland wildlife (Table 1-2). These groups cover the complete riverine ecosystem from the mouth to the headwaters, from the channel to the upland border of the floodplain, and from bottom habitats through the water column. Each work group was composed of a group leader, who was a SJRWMD senior scientist, and one or more other SJRWMD scientists. In addition, each work group was assisted by at least one non-SJRWMD scientist with outstanding expertise in the requisite discipline.

As stated above, the NRC of the National Academies provided peer review.

Table 1-2. WSIS structure and personnel

Ed Lowe	Lead Scientist						
Larry Battoe	Lead Scientist						
Mike Cullum	Lead Engineer						
Pete Sucsy, Dale Smith	Lead Engineers						
Tom Bartol	Project Manager						
Ima Bujak	Asst. Project Manager						
WORK GROUPS							
Hydrology and Hydrodynamics	**Biogeochemistry**	**Plankton**	**Benthic Macroinvertebrates**	**Submersed Aquatic Vegetation**	**Fish**	**Wetland Vegetation**	**Floodplain** **Wildlife**
Peter Sucsy, Lead	Lawrence Keenan*, Lead	Mike Coveney, Lead	Rob Mattson*, Lead	Dean Dobberfuhl*, Lead	Steven Miller, Lead	Palmer Kinser*, Lead	Donna Curtis*, Lead
Getachew Belaineh	Ed Lowe*	John Hendrickson*	Palmer Kinser*	Rob Mattson*	Ron Brockmeyer	Aisa Ceric	
David Christian	Edmond Dunne	Rolland Fulton	Jane Mace*	Lori Morris	Wendy Tweedale	Sandra Fox	(this group utilized
Kijin Park	Angelique Bochnak	Erich Marzolf	Jodi Slater*	Robert Chamberlain	Sue Connors	Donna Curtis*	the results of the
Joseph Stewart	Jian Di*	Larry Battoe	Chuck Jacoby*	Chuck Jacoby*	Chuck Jacoby*	Fay Baird	other study groups)
Ed Carter	Bill VanSickle*	Dean Dobberfuhl	Price Robison	Sonny Hall*	Jan Miller	Peter Sucsy*	
Tim Cera	Robert Freese	Jian Di*		Jodi Slater*	Lori McCloud	Lawrence Keenan*	
Tom Jobes	Cliff Neubauer*	Sonny Hall*			Roxanne Conrow	Jane Mace*	
Shaw Huang	Ima Bujak	Cliff Neubauer*			Walt Godwin	Jodi Slater*	
Dale Smith	John Hendrickson*				Jodi Slater*	Marc Minno	
Yanfeng Zhang	Palmer Kinser*				Jane Mace*	Chris Ware	
Yanbin Jia					Ed Lowe*	Steve Winkler	
Maria Mao						Kim Ponzio	
Marc Adkins						Bill VanSickle*	

Robert Freeman							
Matt Hafner							
David Clapp							
Joseph Amoah							
OUTSIDE EXPERTS							
James Martin	K. Ramesh Reddy	Hans Paerl	Ken Cummins	Ken Moore	Jonathan Shenker	William Wise	Ted Hoehn
Louis Motz	Alan Wright	Ed Phlips	Rich Merritt	Bob Virnstein*	Tim McDonald	Clay Montague	Jamie Feddersen
Patrick Tara			Paul Montagna		Dennis Helsel		Steve Johnson
Matt Goodrich			Bob Virnstein*		Shahrohk Rouhani		
Shahrohk Rouhani							
Steven Peene							
A.J. Mehta							
* = served on more than on workgroup							

The WSIS began in January 2008 and was completed in February 2012. Phase I began in January 2008 and ended with completion of an interim report in December 2008. The purpose of the interim report was to outline preliminary results and, more importantly, to describe the methods to be used for completion of the study. Another purpose for the interim report was to provide a foundation for initial review comments by the NRC peer review panel. Phase II began in January 2009 and was completed with the completion of the final report in February 2012. The final report replaces, rather than supplements, the interim report, which will remain as a draft.

In October 2008 and 2009, the staff and expert consultants met in a symposium to exchange information among the study groups and to discuss accomplishments during Phase I. The symposia were broadcast live on the Internet and were open to the public. Public comment was accepted at each meeting and at an email address for a week after each meeting.

Concurrent with the project, beginning in January 2009, the NRC has provided peer review. The NRC panel has reviewed project materials and has met approximately every four months. The NRC panel's final report was published in December 2011.

2 GENERAL METHODS OF THE WATER SUPPLY IMPACT STUDY

2.1 OVERARCHING METHOD

The overarching method of this study was to use hydrologic, hydrodynamic, and hydroecological models to simulate (over at least 10 years) the deviation from a baseline condition, caused by a water withdrawal (assuming rainfall rates as previously recorded). The modeled deviations for hydrologic and hydrodynamic drivers (H&H drivers) became forcings for hydroecological (HE) models used to simulate deviations in the state of ecological attributes (the potential effects) (Figure 2-1).

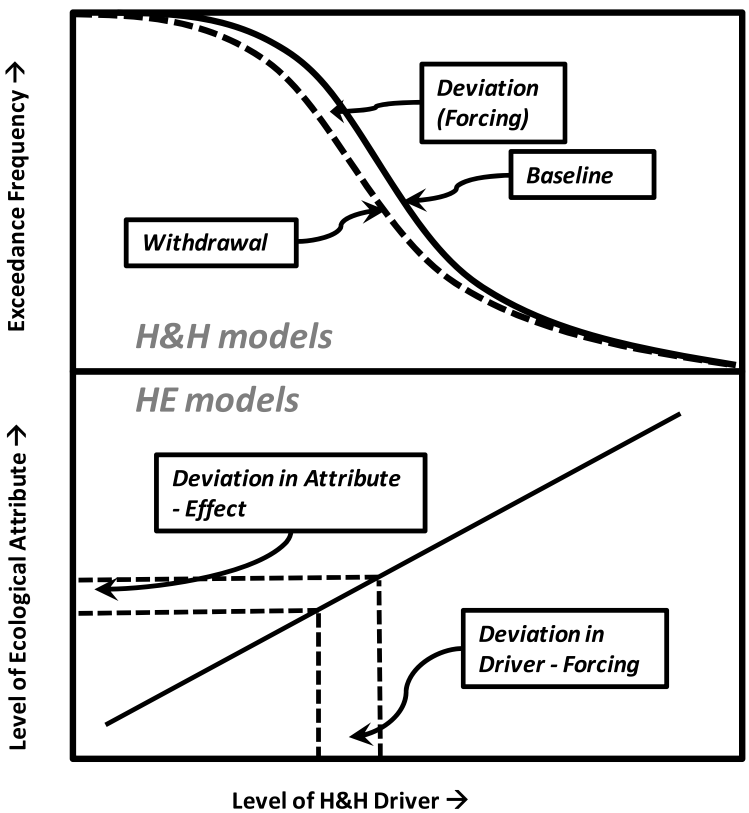

Figure 2-1. The overarching method of the WSIS. Using hydrologic and hydrodynamic (H&H) models, the deviation from a baseline condition caused by a water withdrawal was simulated over at least 10 years (top panel). The deviation (baseline minus withdrawal) in the level of an H&H driver became the forcing used in hydroecological (HE) models that related the state of ecological attributes to the level of H&H drivers (bottom panel). The effect of the forcing was the modeled deviation in the state of one or more ecological attributes, the effect(s).

2.2 MODELED WITHDRAWALS

Using an hydrologic model, the WSIS examined the potential hydrologic effects of water withdrawals at levels predicted not to cause violations of the MFLs adopted for the three previously described locations (SR 44, SR 50, and Lake Monroe). In the hydrologic model, river water was withdrawn at various rates from four locations (Figure 2-2):

1.) Up to 55 mgd from a point just north of SR 520 (near Lake Poinsett)

2.) Up to 50 mgd from the St. Johns River at SR 46 near the mouth of Lake Jesup

3.) Up to 50 mgd from a canal on the west side of the river just north of I-4 (Yankee Lake)

4.) Up to 107 mgd from the lower Ocklawaha River (upstream end of Rodman Reservoir).

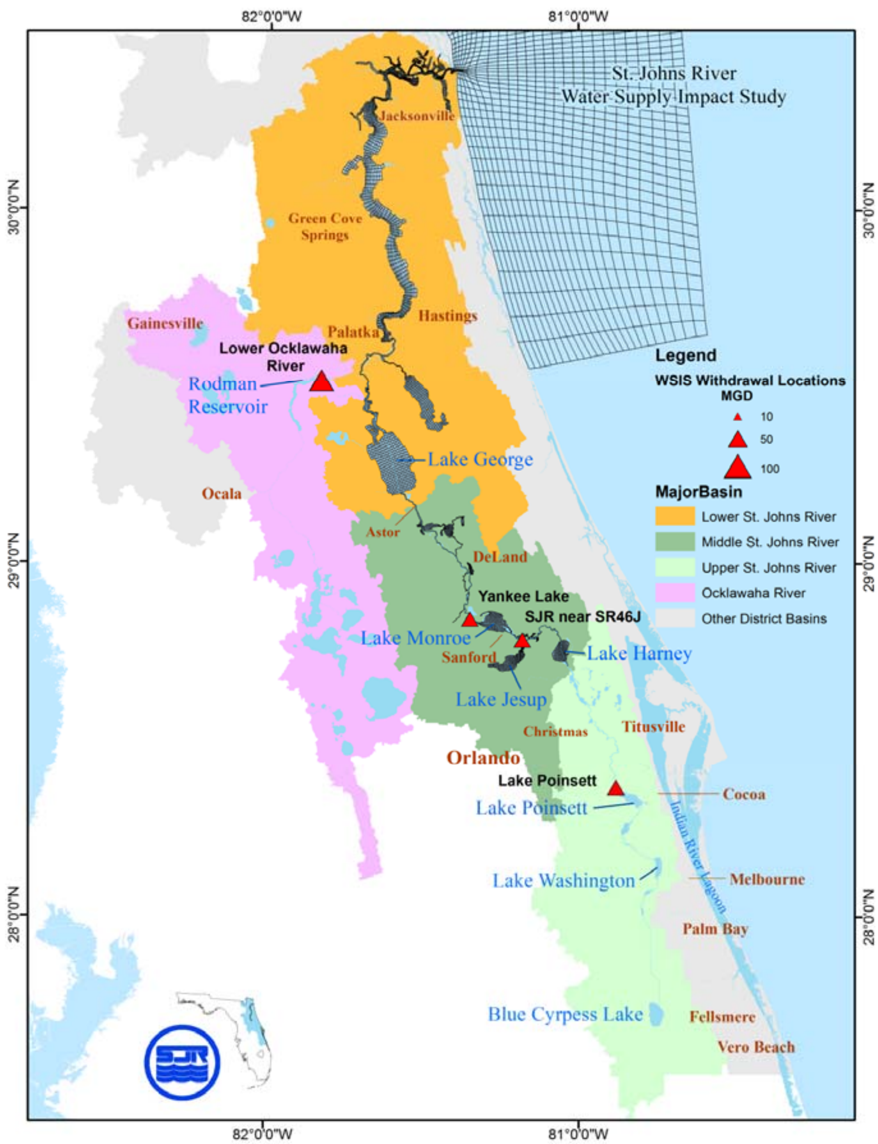

Figure 2-2. Modeled water withdrawal points

In order to prevent modeled violations of MFLs, the modeled rate of withdrawal was not constant at the Lake Poinsett withdrawal point. At discharge rates ≤ 8.50 $m^3 s^{-1}$ (≤194 mgd) at SR 50 (RK 343), withdrawals near Lake Poinsett ceased; withdrawals increased from 0.28 – 3.68 m^3

s^{-1} (6.39 – 84.0 mgd) as discharge increased from 8.50 – 18.12 $m^3 s^{-1}$ (194- 414 mgd; Chapter 3. Watershed Hydrology). Modeled water withdrawals at the other three sites were constant.

2.3 HYDROLOGIC MODELING OF SURFACE WATER RUNOFF

The first step in evaluation of the potential hydrologic and hydrodynamic effects of water withdrawals was simulation of surface water runoff from the St. Johns River watersheds using the Hydrologic Simulation Program – Fortran (HSPF) hydrologic model (Chapter 3. Watershed Hydrology). The HSPF hydrologic models supplied surface water discharge from over 90 watersheds to simulate in-stream flows and levels for the upper St. Johns River and to establish boundary conditions for a downstream hydrodynamic model. Development of the HSPF hydrologic models along with description of the physical characteristics of the watersheds are described in detail in Chapter 3. Watershed Hydrology.

The HSPF hydrologic models were calibrated using observed discharge and water levels from 1995-2006. Watershed conditions for 1995 were used to define the baseline condition. SJRWMD selected these timeframes for several reasons. A primary reason was that SJRWMDs first water supply planning year (baseline year) was 1995. An additional justification was that SJRWMDs Upper St. Johns River Basin (USJRB) Project did not have major improvements constructed between 1995 and 2006, which allowed for some stability in the hydrologic models' representations of this important project for model calibration. In addition, by 1995, SJRWMDs data collection program had matured to the point that we considered the data from 1995-2006 to be of sufficient quality for an accurate calibration. The HSPF output compared favorably to observed flows and water levels at gauged locations. A common measure of the performance of a hydrologic model is the Nash-Sutcliffe statistic. Using Nash-Sutcliffe, the calibration performance results show that 95% of the calibrated HSPF hydrologic models were rated "satisfactory" to "very good" (Nash-Sutcliffe statistic of 0.5 to 1.0).

The HSPF hydrologic models were used to evaluate the potential hydrologic effects of altered watershed conditions between the 1995 baseline condition and a projected 2030 condition – the planning horizon for water supply planning (SJRWMD 2006). Land use changes for the 2030 watershed condition were developed from parcel-level 2030 population projections of the Bureau of Business and Economic Research (BBER 2009).

2.4 HYDROLOGIC MODELING OF WATER WITHDRAWAL IN THE UPPER ST. JOHNS RIVER

The HSPF hydrologic models were used to simulate the effects of water withdrawals on flows and water levels within the upper St. Johns River extending from the headwaters to the upstream boundary of the EFDC (the Environmental Fluid Dynamics Code) hydrodynamic model at Lake Harney (Figure 2-3) (Chapter 3. Watershed Hydrology). Water withdrawals from the upper St. Johns River at the Lake Poinsett location were simulated using the HSPF hydrologic models. Although surface water withdrawals removed water from the river, two model conditions added water to the river. These conditions were (1) increased runoff from increased impervious area associated with land development from 1995 to 2030 land use and (2) increased discharge resulting from USJRB restoration projects that are expected to be completed by 2030. The USJRB restoration projects redivert flows presently directed into the Indian River Lagoon back to the St. Johns River. The USJRB projects, with anticipated completion dates are:

- Phase I C-1 Re-diversion Project (2011)
- Fellsmere Water Management Area (2015)
- Three Forks Marsh Conservation Area (2012).

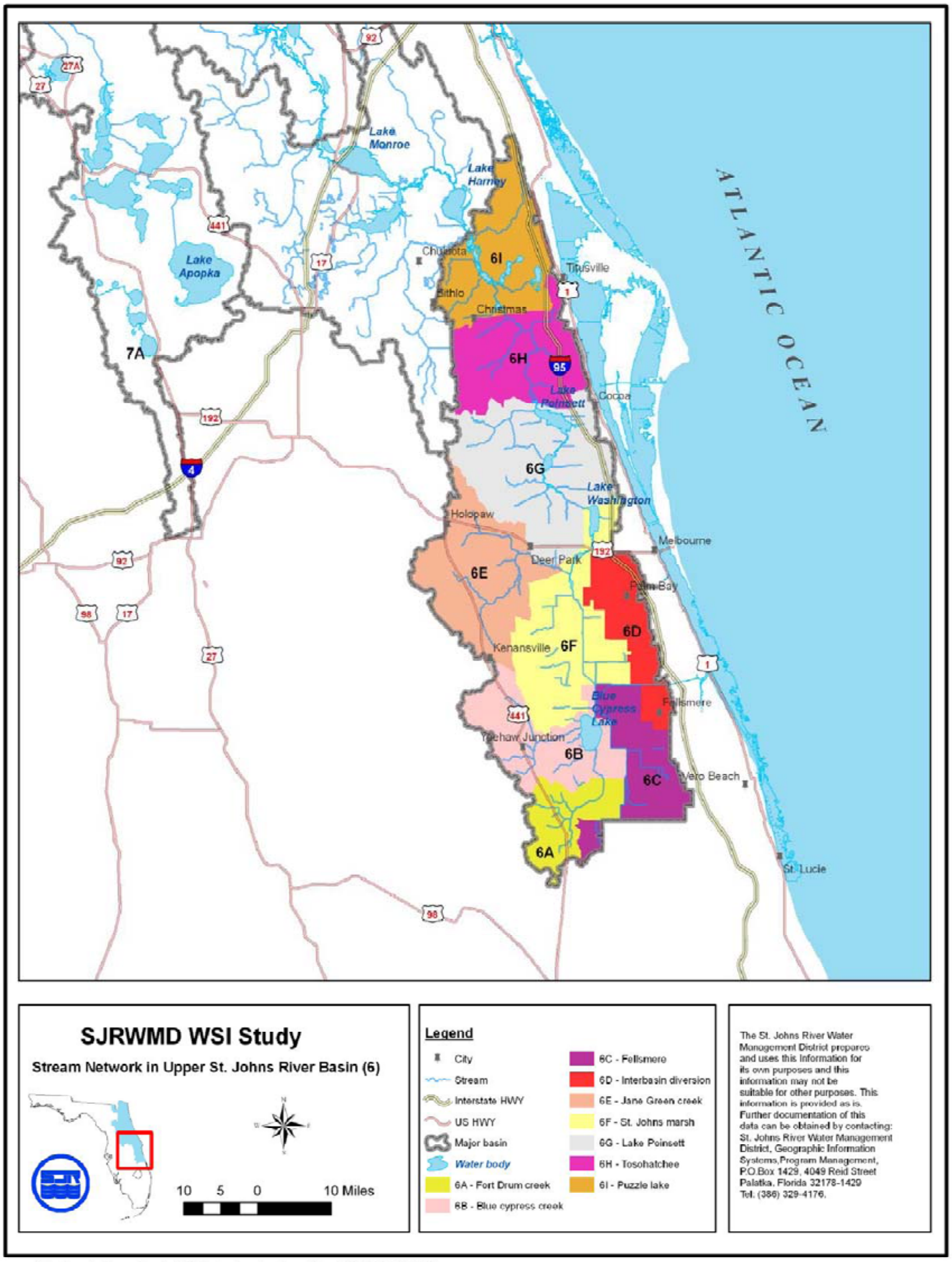

Figure 2-3. Stream network in Upper St. Johns River. In this river reach, hydrologic effects of water withdrawals at Lake Poinsett were analyzed using HSPF hydrologic models.

2.5 HYDRODYNAMIC MODELING OF WATER WITHDRAWALS IN THE MIDDLE AND LOWER ST. JOHNS RIVER

In the middle and lower St. Johns River and in Lake George, the effects of water withdrawals on water level, discharge, salinity, and residence time were analyzed using a finite-difference, three-dimensional EFDC hydrodynamic model. Chapter 5. River Hydrodynamics Calibration contains detailed information on the structure, set-up, and calibration of the EFDC hydrodynamic model. Hydrodynamic modeling was required in these areas to address the complex mixing patterns in the large lakes, the complexities introduced by tidal effects and stratification, and the need to simulate residence time (using modeled water age) and salinity. Using the EFDC hydrodynamic model it was feasible to simulate the following features:

- 3D wind-driven flow structure within the large flow-through lakes
- Influence of low frequency ocean water level variability on river stage
- Effects of sea level rise upstream through the middle St. Johns River
- Advective-diffusion of salinity and water age.

The EFDC hydrodynamic model captures these features by simulating the combined effects of ocean tide, tributary, spring, wastewater, and groundwater discharges, rainfall, evaporation, and wind. The EFDC hydrodynamic model calculates water level, velocity, discharge, salinity, and water age at each interior model cell at 30-s time intervals over the model simulation period of 1996 – 2005. Model cells are contained within a model grid (Figure 2-4) that defines the extent of the model domain. Vertical resolution is provided by division of each horizontal model cell into six equally-spaced vertical layers.

The effect of water withdrawals on river hydrodynamics was analyzed using the EFDC hydrodynamic model by simulating constant water withdrawals at the Yankee Lake and near SR 46 at Lake Jesup (SR 46J) locations. The effect from water withdrawals at the Lake Poinsett location were analyzed within the EFDC hydrodynamic model by a one-way coupling of the HSPF hydrologic model and EFDC hydrodynamic model above Lake Harney. A similar one-way coupling at the exit of Rodman Reservoir was used to study the hydrodynamic effects of water withdrawals from the lower Ocklawaha River on the lower St. Johns River.

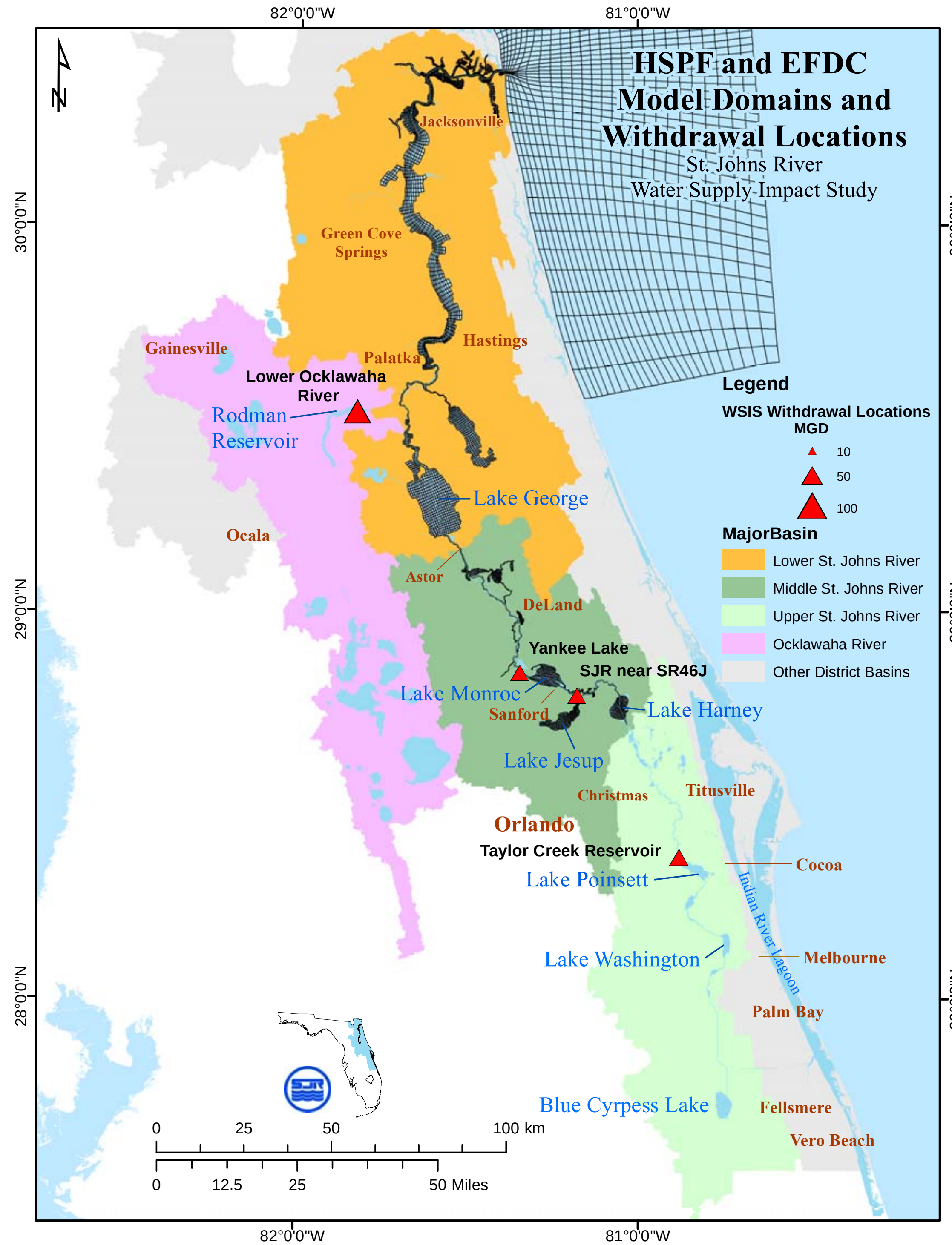

Figure 2-4. Model domains and model grids used in the EFDC hydrodynamic model for the middle and lower basins of the St. Johns River.

2.6 River Segments

The St. Johns River is not ecologically uniform because its geomorphology, hydrology, and hydrodynamic characteristics vary along it length. In response to the geographic variation in these and other environmental factors, biological communities also change from the headwaters to the mouth. In recognition of this variation, we divided the river into nine segments based on geomorphology, hydrology, hydrodynamics, water quality, soils, and floodplain communities (Figure 2-5). The potential for environmental effects associated with hydrologic and hydrodynamic (H&H) drivers differs among the segments. Consequently, our environmental assessments were not uniform among river segments. These river segments were fundamental to our analysis. A description of each river segment is in Appendix 2.

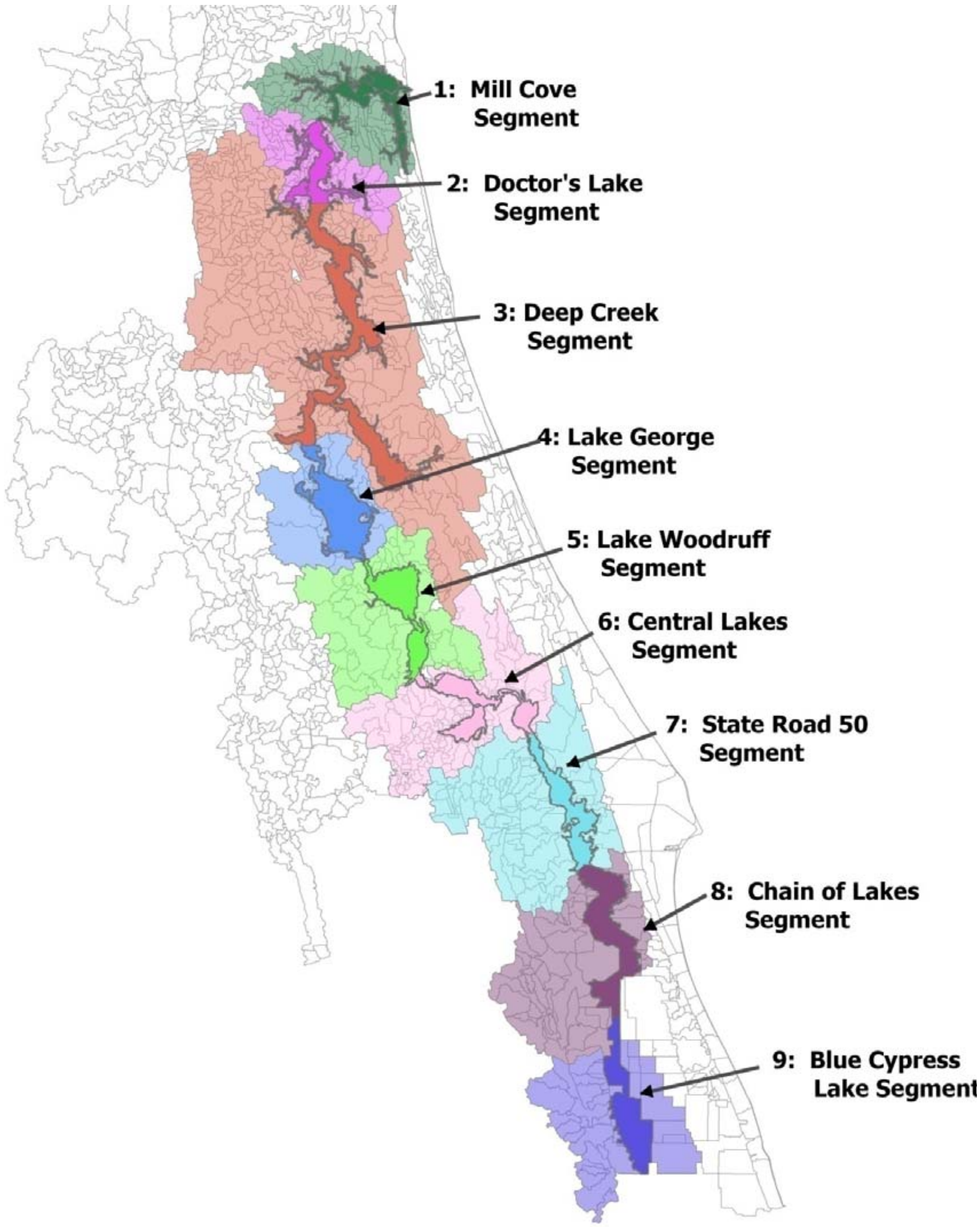

Figure 2-5. The river segments developed for the WSIS. Segment 9 and the southern portion of segment 8 (south of the Lake Washington weir) are hydrologically isolated from the withdrawal sites. Consequently, they were not used in the analysis.

2.7 Conceptual Models

A considerable body of scientific literature indicates that variation in hydrologic variables can affect the status of an array of physical, chemical, and biological ecosystem state variables through a complex chain of causation (e.g. Poff and Zimmerman 2010). As a means of elucidating salient hypotheses regarding the interaction between H&H drivers and ecological attributes, the ecological workgroups developed a suite of simplified conceptual models, similar in form to path diagrams (sensu Shipley 2002). A general conceptual model and workgroup-specific models are in Appendix 3. Each ecological chapter (Chapters 7 through 13) also includes a conceptual model along with a discussion).

A simplified general model illustrates how additional surface water withdrawals can affect a suite of fundamental H&H drivers: (1) salinity, (2) residence time (or water age in the EFDC hydrodynamic model), (3) water level, and (4) loadings of particulate and dissolved constituents (Figure 2-6). (Flow velocity can also be affected, but this aspect was not examined due to a paucity of empirical data.) The ecological significance of a water withdrawal regime hinges on the degree to which it alters the status of these H&H drivers and on its potential for direct effects on the biota through entrainment and impingement of planktonic and nektonic organisms.

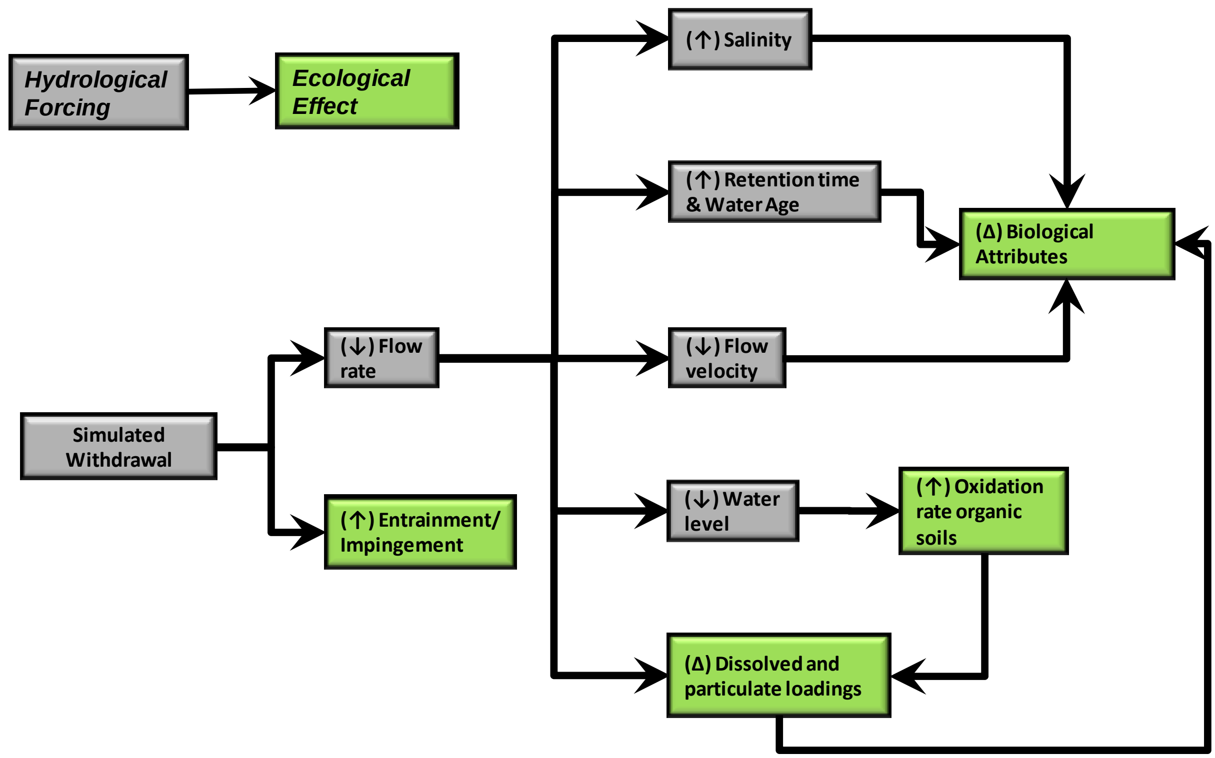

Figure 2-6. Simplified general conceptual model of the potential effects of a surface water withdrawal on H&H drivers for ecological attributes. There was insufficient information to evaluate the potential effects of changes in flow velocity.

Each workgroup developed a more detailed conceptual model following a standardized format (Figure 2-7). These conceptual models illustrated the key attributes to be evaluated, their drivers, and the other attributes that they, in turn, affect. We used the term “key effect” to denote an effect on a key attribute. Each workgroup used empirical and/or mechanistic hydroecological (HE) models to assess key effects, i.e., the potential effects of H&H drivers on key attributes. Not all hypothesized causative links were evaluated. In many cases, an HE model linked two points in a causal chain without modeling intermediate steps in the chain. However, according to the tenets of path analysis (Shipley, 2002), the effects of intermediate links in a causal chain will be captured by the correlation between endpoints.

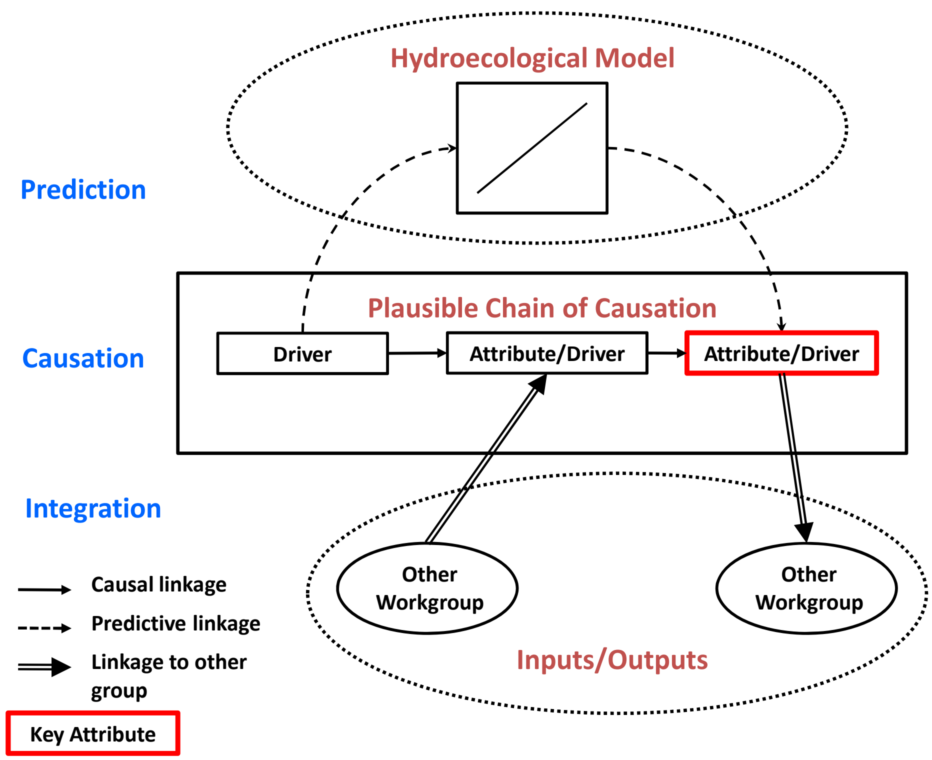

Figure 2-7. Format for work group conceptual models. The effects of linkages in the models were quantified either by a mechanistic or empirical model. In many cases, intermediate causative links were not modeled.

2.8 Hydroecological Models

For analysis of potential ecological effects, each workgroup developed hydroecological models (HE models) that related the status of key ecological attributes to H&H drivers (Table 2-1). The HE models were wholly empirical for all metrics related to key attributes except those related to phytoplankton bloom intensity, likelihood, and duration. In the phytoplankton models, the H&H driver, water age, was derived from the EFDC hydrodynamic model. Water age, a model output, was related to empirical data on phytoplankton composition and density to produce semi-empirical HE models. In addition, a mechanistic water quality model (CE-QUAL-ICM) was used to provide corroboration for the semi-empirical HE models.

We used the output of H&H models as input to the HE models to evaluate the potential changes (i.e., deviations from the baseline condition) in ecosystem attributes associated with H&H drivers for the WSIS scenarios. The HE models used to quantify, or qualify, key effects are indicated in the conceptual models for each work group (Appendix 2.C.).

Table 2-1. Primary ecological attributes, H&H drivers, and hydroecological models used for each attribute.

Ecological Attribute	H&H Driver	Type of Ecological Model	Source Data for Model Development	Assessment of Goodness of Fit
Oxidative Release of DOC and Nutrients	Hydroperiod	Mechanistic model of areal release using empirical estimates of increase in release rates with increased days of exposure – simulation period was 11/1/75-10/31/08.	Site-specific empirical data on release rates (2/9/08 – 6/22/09); digital elevation model	Reasonable consistency among separate release rate studies; findings congruent with low soil C:N ratios
DOC and nutrient loads to the river from oxidative release	Evapotranspiration and Rainfall	Empirical Model of Reduction in Concentration with Time	Literature data for uptake rate coefficients	Reasonable correspondence and values from long term nearby site
Decline in Dissolved Oxygen concentrations due to increase loads of DOC	Flow	Statistical model (multiple regression)	Site-specific empirical data	R^2 of 0.415
Phytoplankton bloom magnitude and duration and frequency that bloom thresholds were exceeded	Various measures of water age (residence time)	Statistical Models (Multiple regressions) – simulation period was 1996 - 2005	Site-specific empirical data (1979 – 2008) and modeled water age (1996 – 2005)	High adjusted R^2 (0.80 – 0.97); models and coefficients significant ($p < 0.05$); residuals normally distributed
SAV stress	Salinity level and duration	Qualitative stress model – simulation period was 1996 - 2005	Literature, site-specific empirical data on SAV (1998-2008), site-specific experimental data (1997-2010)	Good correspondence with field observations, mesocosm studies, and field transplant studies
Wetland Vegetation	Water Elevation	GIS-based model of hydrologic change combined with species/community distribution vs. hydroperiod model; simulation period was 1996 - 2005	Site-specific vegetation transect data (collected 2004-2006) and scientific literature	Good correspondence with site-specific transect data showing community transition points; Good confidence in hydrologic model, in digital elevation model, and in baseline vegetation community composition and distribution

Wetland Vegetation	Salinity	Salinity level - community threshold model; simulation period was 1996-2005	Site-specific soil salinity gradient/vegetation data (collected July and August, 2010); scientific literature	Good correspondence between soil salinities and Ortega River high salinity (95th percentile); R^2 = 0.9; site-specific community thresholds supported by literature
Freshwater Benthic Macroinvertebrates	Water Elevation	Qualitative and weak quantitative models of Species/community distributions/abundances vs. water level/hydroperiod and habitat; simulation period was 1996-2005	Site-specific data (2009) and scientific literature	Good correspondence between site-specific field observations and the scientific literature
Estuarine Benthic Macroinvertebrates	Salinity level and duration; areal extent of salinity zones	Non-linear and linear regression; qualitative model of static and dynamic habitat	Site-specific empirical data (1973 – 1999 and 2000) and literature	$0.26 \leq R^2 \leq 0.55$; good correspondence with field observations
Fish	Hydroperiod	Linear regression relating abundance of small floodplain fish to hydroperiod – simulation period was 1975 - 2008	Scientific Literature	Good correspondence in a similar ecosystem with similar species – the Everglades
Fish	Freshwater Flows to the Estuary	Linear regressions of distribution/abundance vs. discharge – simulation period was 1995 - 2005	Site-specific data – Jan. 1 2001 to Dec. 31 2010	$0.25 < R^2$ and the value of the PRESS r^2 statistic
Floodplain Wildlife	Hydroperiod and salinity	Qualitative assessment (see models used by the Wetland Vegetation, Benthic Macroinvertebrate, and Fish assessments)	Scientific literature; MFL data	Good confidence in hydrologic model, in digital elevation model, and in baseline vegetation community composition and distribution

Note:
C = Carbon
N = Nitrogen
DO = Dissolved oxygen
DOC = Dissolved organic carbon
SAV = Submersed aquatic vegetation

2.9 WSIS Scenarios

Through hydrologic and hydrodynamic modeling coupled with the hydroecological models, we evaluated the potential effects of water withdrawals in the context of past and future water shed conditions and sea levels. To examine the interactions of differing levels of water use and differing assumptions regarding watershed conditions and sea levels, 15 different withdrawal scenarios were modeled (Table 2-2; see Chapter 6. River Hydrodynamics Results for detailed discussion). Each withdrawal scenario consisted of a magnitude of additional water withdrawal [77.5 mgd (3.395 m^3 s^{-1}) or “Half”; 155 mgd (6.791 m^3 s^{-1}) or “Full”; or 262 mgd (11.479 m^3 s^{-1}) or “FullOR”], a land use condition (1995 or 2030), and a sea level condition (1995 level (N) and 1995 level plus 14 cm(S)). The scenarios also included two conditions for the Upper St. Johns River Basin Project: 1995 status (N) and completed (P). Withdrawal scenarios were identified using a nomenclature that specifies these selected features. For example, the withdrawal scenario Full2030PS is a scenario with a 155 mgd water withdrawal (**Full**2030PS), 2030 land use (Full**2030**PS), USJRB restoration projects implemented (Full2030**P**S), and expected sea level for the year 2030 (Full2030P**S**).

Each of the 15 withdrawal scenarios also can be classified either as a test scenario, used only to explore the relative sensitivities of key attributes, or as a hindcast or forecast scenario, used to explore sensitivities but also to estimate more realistic potential effects. The test scenarios allowed a more complete exploration of the hydrologic sensitivities of the attributes we examined. In many cases, conclusions regarding hindcast or forecast scenarios were supported by the results of analyses for more extreme test scenarios. Moreover, in some cases, margins of safety can be inferred for conclusions for hindcast and forecast scenarios from conclusions reached for extreme test scenarios. Four scenarios lacking any additional surface water withdrawals were modeled by the H&H workgroup: Base1995NN; Base1995PN; Base2030PN; and Base2030PS. The environmental analysis used Base1995NN as the baseline condition. The environmental workgroups did not evaluate the other three non-withdrawal scenarios because they were uninfluenced by water withdrawals and because, being forecasts, there was no empirical data from which to derive baseline biological status. All withdrawal scenarios were evaluated with respect to the modeled deviations from the baseline condition (Base1995NN).

Table 2-2. The withdrawal scenarios analyzed in the WSIS for ecological effects. All withdrawal scenarios were evaluated with respect to the modeled deviations from the baseline condition (Base1995NN).

Test Scenarios	Hindcast Scenarios	Forecast Scenarios
Half1995NN	Half1995PN	Half2030PS
Full1995NN	Full1995PN	Full2030PS
FwOR1995NN		FwOR2030PS
FwOR1995PN		
Half1995PS		
Full1995PS		
FwOR1995PS		
Half2030PN		
Full2030PN		
FwOR2030PN		

2.10 Scenario Screening

Considering that there are 15 withdrawal scenarios, eight river segments, four basic H&H drivers (discharge, water level, residence time, and salinity), and seven ecological workgroups, there were at least 3,360 different combinations for evaluation (15 x 8 x 4 x 7). Many workgroups had a diverse set of key attributes that could be affected by water withdrawals, so the actual number of potential combinations was even larger. To efficiently assess the very large number of potential combinations for analysis, we employed a set of screening principles. As described above, the first and most fundamental of these principles was to exclude from environmental analysis those scenarios lacking an imposed water withdrawal, with the single exception of the baseline condition (Base1995NN). Clearly, H&H deviations from the baseline condition in the non-withdrawal scenarios could not stem from water withdrawals.

A second screening principle separated water withdrawal scenarios into two groups with respect to the deviation from the baseline condition: (1) those with "reduction" effects, denoting effects on one or more H&H drivers associated with a decrease of discharge or water level below the baseline condition; or (2) those with "augmentation" effects, denoting effects associated with an increase of discharge or water level above the baseline condition (Figure 2-8). Because augmentation effects cannot be attributed to water withdrawals, they were not evaluated for potential environmental effects.

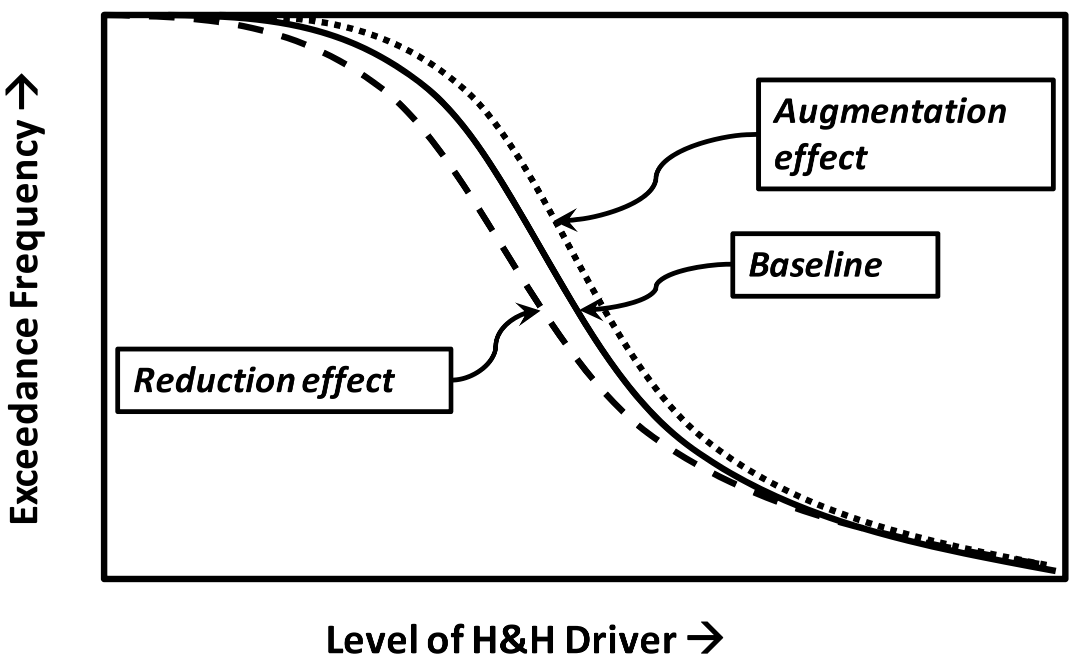

Figure 2-8. Scenarios can have reduction effects or augmentation effects. A reduction effect is any effect on an H&H driver that is linked to a reduction in water level or discharge with respect to the baseline condition (Base1995NN). Only scenarios with predicted reduction effects caused by water withdrawals were evaluated for potential environmental effects. An augmentation effect is any effect on an H&H driver that is linked to an increase water level or discharge above the baseline condition. Because augmentation effects cannot be attributed to water withdrawals, they were not evaluated for environmental effects.

A third screening principle was to rank scenarios and river segments (described below) in terms of the magnitudes of predicted H&H deviations from the baseline condition and to analyze the scenarios in sequence from the largest to smallest deviations from the baseline condition. Logic dictates that analysis could cease at the highest ranked scenario that had negligible to minor potential environmental effects because all lower-ranked scenarios would have even weaker effects. Similarly, we ranked river segments, from the most to least sensitive, and analyzed them in sequence. Again, the analysis could cease at the segment with the most extreme potential H&H effects that had only negligible or minor environmental effects.

A fourth screening principle was to rank the sensitivities of environmental state variables. We can safely assume that effects on attributes will decline down a ranking from the most to least sensitive attribute.

Finally, in some cases, we first analyzed scenarios using simplified and extreme assumptions regarding environmental drivers. A primary example is the assumption of exact equivalence of water levels in the river and its associated wetlands; an assumption employed in assessment of potential floodplain effects. Because roughness and topographic variation impedes floodplain drainage at very shallow water depth, it is clear that this is an unrealistically extreme assumption that would overestimate the potential environmental effects. However, if we found negligible or minor effects for potential scenarios using this extreme assumption, we could be confident that more realistic modeling of wetland hydrology would yield even weaker potential environmental effects.

2.11 Assessment of Ecological Significance

For each scenario, our evaluation of a key effect included an assessment of its ecological importance based on a consideration of (1) the diversity of ecosystem components affected, (2) the strength of the effect in terms of its intensity and its size, and (3) its persistence (Table 2-3; details in Appendix 2.C). Further, each work group assessed whether there was existing degradation of the attributes they evaluated and, if so, at what level of significance using the same scale of effects. In order to allow for professional discretion, it was not intended that the assessment of the levels of effects be narrowly constrained by the general guidelines. However, several groups used a three-step ordinal scale for each factor to determine the levels of effects (Table 2-4).

Table 2-3. Scale for evaluation of the level of ecological importance of an effect.

Level of Effect	Criteria
Extreme	Effect is persistent, strong, and highly diverse; significant change in natural resource values
Major	Effect is persistent, strong, but not highly diverse; significant change in natural resource values
Moderate	Effect is ephemeral or weak or is limited to minor species, no significant change in natural resource values
Minor	Effect is ephemeral and weak; no significant change in any ecosystem attribute
Negligible	No appreciable change in any ecosystem attribute

Table 2-4. Interpretation of guidelines for levels of effects using a three-step ordinal scale for each factor (strength, diversity, and persistence of the effect).

Strength, Persistence	High Diversity (3)	Medium Diversity (2)	Low Diversity (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 effect
Minor effect
Moderate effect
Major effect
Extreme effect

2.12 Assessment of Scientific Uncertainty

Uncertainty is implicit in any prediction. When dealing with predictions for complex systems, uncertainty is typically high. Some scientific uncertainty is amenable to a statistical, or frequentist, approach wherein the probability distribution about the central tendency of a prediction can be specified. In many cases, however, other uncertainties remain that are not amenable to a frequentist approach, such as the uncertainty arising from conflicting conclusions or opinions in the relevant scientific literature. It was for this reason that the International Panel on Climate Change (IPCC) relied heavily on a Bayesian, or subjective, approach for assessment of uncertainty (Moss and Schneider 1997). We adopted an approach similar to that used by the IPCC, wherein the work groups could consider (1) the strength of the evidence they obtained from observations, statistical models, and mechanistic models; (2) the amount of evidence and the degree of agreement of evidence from different sources, including other site-specific work, relevant work in the scientific literature, and expert opinion; and (3) the state of understanding of the mechanism underlying the effect. Statistical uncertainties were quantified using standard statistical techniques, but the overall uncertainty associated with major findings was characterized using a qualitative model that considered all these aspects (Table 2-5). As with the assessment of levels of effects, the guidelines for assessment of uncertainty were broadly interpreted among the workgroups. An interpretation based on a three-step ordinal scale for each factor (strength of the predictive model, strength of supporting evidence, and level of understanding of the causative mechanisms), provides additional insight into the basis for the different levels of uncertainty (Table 2-6).

Table 2-5. The scale used for assessment of scientific uncertainty.

Level of Uncertainty	Criteria
Very Low	Very strong quantitative evidence – Strong predictive model (PM), strong supporting evidence (SE), and good understanding of the mechanism (UM)
Low	Strong quantitative evidence – Strong PM and either strong SE or strong UM
Medium	Moderate quantitative evidence or strong qualitative evidence – Strong PM or strong SE and UM
High	Weak quantitative evidence or moderate qualitative evidence – Weak or no PM and at least moderate SE or UM
Very High	Weak qualitative evidence – No predictive model and weak SE and UM, i.e., weak in all three areas

Table 2-6. Interpretation of the guidelines for assessment of uncertainty using a 3-step ordinal scale for each factor: strength of the predictive model, strength of supporting evidence, and level of understanding of the causative mechanisms.

Predictive Model, Supporting Evidence	High Mechanistic Understanding (3)	Medium Mechanistic Understanding (2)	Low Mechanistic Understanding (1)
3,3	3,3,3	3,3,2	3,3,2
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 uncertainty
Low uncertainty
Medium uncertainty
High uncertainty
Very high uncertainty

3 INTEGRATED RESULTS AND DISCUSSION

3.1 DEVIATIONS IN HYDROLOGIC AND HYDRODYNAMIC DRIVERS (FORCINGS)

3.1.1 INCREASE IN DISCHARGE FOR 2030 WATERSHED CONDITIONS

HSPF hydrologic modeling indicates that both intensification of land uses and completion of the USJRB restoration projects will increase discharge of the St. Johns River. By the year 2030, increased discharge would be sufficient to reduce or eliminate the effects of water withdrawals. This is a salient finding of the WSIS. The magnitudes of the increase in river discharge for 2030 watershed conditions compared with the 1995 baseline condition are presented below.

Increase in Discharge by 2030 Land Use

The increase in developed land associated with 2030 land use, along with its resultant increase in impervious area, leads to projections of increased runoff for 2030 scenarios. To isolate the predicted effect of land development from 1995 to 2030, HSPF models' results were independently examined for the different land use conditions. Table 3-1 presents the average annual discharge from the five planning units for the two land use conditions. Modeled average river discharge for 2030 land use condition was 10.6% higher than that for 1995 land use conditions.

Table 3-1. Average annual discharge (1975-2005) for 1995 and 2030 land use conditions for five major basins of the St. Johns River. Discharge was obtained from HSPF hydrologic models.

Major Basin Number	Major Basin Name	1995 Land Use Average Discharge (mgd)	2030 Land Use Average Discharge (mgd)	% Change
6	Upper St Johns River Basin	842	932	10.7%
4	Middle St Johns River Basin	625	711	13.8%
5	Lake George River Basin	188	207	10.1%
7	Ocklawaha River Basin	759	796	4.9%
3	Lower St Johns River Basin	1412	1605	13.7%
	Total	3826	4251	10.6%

See Chapter 3. Watershed Hydrology, for a definition and description of the major basins.

Because the need for water withdrawals and intensification of land use are closely associated, it is very likely that the increases in runoff and increases in withdrawals will occur concurrently. If development occurs slower than projected, then water demand also will grow more slowly than projected. If development accelerates, then water demands will also increase proportionately.

Increase in Discharge Due to Completion of the USJRB Restoration Projects

The second source of additional water for both hindcast and forecast withdrawal scenarios was completion of several large restoration projects in the USJRB that are currently under construction. While the initial goal of the USJRB project was to provide protection of agricultural lands in the upper basin from flooding, a more recent goal is to return to the river flows that were previously diverted to the Indian River Lagoon. The projects, with anticipated completion dates are:

- Phase I C-1 Re-diversion Project (2011)
- Three Forks Marsh Conservation Area (2012)
- Fellsmere Water Management Area (2015).

A comparison of the water levels and flows between the two scenarios, both with no additional withdrawals (Base1995NN and Base1995PN) but one without and one with completion of the Upper Basin projects, indicates that both mean stage and mean flow would increase with completion of the projects (Table 3-2).

Table 3-2. Mean stage and discharge at Cocoa for the baseline condition (Base1995NN) and a scenario that includes the USJRB restoration projects (Base1995PN). Stage and discharge are simulated using the HSPF hydrologic model.

	Base1995NN	Base1995PN	Difference	Percent Difference
Mean Stage (ft, NGVD29)	11.72	11.95	+0.23	+2.0
Mean Discharge (mgd)	536	546	+10	+1.9

3.1.2 Effects of Scenario Conditions on Forcings – Water Levels and Discharge

Water Level and Discharge Effects Within the Upper St. Johns River

Water level and discharge effects in the upper St. Johns River were analyzed using the HSPF hydrologic model. Results are shown for two locations, Cocoa and Christmas, which were used for calibration of the HSPF hydrologic model. The Cocoa location (SR 520; RK 328) is very close to Lake Poinsett, the point of withdrawal from the St. Johns River to Taylor Creek Reservoir. The Christmas location (SR 50; RK 343) is located downstream of Taylor Creek Reservoir where the water withdrawal of 55 mgd is taken. Since Taylor Creek Reservoir has its own watershed, the water withdrawal from the St. Johns River at Lake Poinsett is on average less than the water withdrawal from Taylor Creek Reservoir. Operation of Taylor Creek Reservoir can also affect downstream discharge by augmenting river discharge to meet MFL requirements during dry periods and controlling flooding during wet periods.

A full withdrawal test scenario, with an average water withdrawal of 55 mgd from Taylor Creek Reservoir (Full1995NN), had reductions in average stage and discharge of 0.18 ft and 45 mgd (-8.4%), respectively, from the baseline condition (Base1995NN) at Cocoa (Table 3-3). Reductions of stage and discharge at Christmas were 0.17 ft and 54 mgd (8.1%). The Full1995NN withdrawal scenario is an unrealistic test scenario that does not include expected completion of the USJRB restoration projects and existing or future land use changes compared to the 1995 baseline condition. These differences represent changes to stage and discharge that would have occurred if the proposed water withdrawals were implemented in 1995. The Half withdrawal scenarios exhibited similar results, with a corresponding reduction in effects.

The effect of the USJRB restoration projects on stage and discharge is seen by comparing the baseline condition (Base1995PN) to water withdrawal scenarios that include the completed USJRB projects (Full1995PN and Half1995PN in Table 3-3). Average stage at Cocoa for the full-withdrawal-plus-projects scenario (Full1995PN) increased 0.04 ft. from the baseline condition, while average discharge decreased by 36 mgd (6.7%). Differences of stage and discharge at Christmas show similar effects, with an increase in average stage and a reduction in average discharge.

Table 3-3. Modeled mean stage and flow for various scenarios for sites in the Upper St. Johns River using HSPF.

USJRB Scenario	Cocoa Mean Stage (ft NGVD29)	Cocoa Mean Flow (mgd)	Change in Flow at Cocoa (%)	Christmas Mean Stage (ft NGVD29)	Christmas Mean Flow (mgd)	Change in Flow at Christmas (%)
Base1995NN	11.72	536	NA	6.11	668	NA
Half1995NN	11.62	513	-4.3%	6.03	639	-4.3%
Full1995NN	11.54	491	-8.4%	5.94	614	-8.1%
Base1995PN	11.95	546	+1.9%	6.27	676	+1.2%
Half1995PN	11.84	522	-2.6%	6.17	647	-3.1%
Full1995PN	11.76	500	-6.7%	6.09	621	-7.0%
Base2030PS	12.09	587	+9.5%	6.47	740	+10.8%
Half2030PS	11.99	564	+5.2%	6.37	710	+6.3%
Full2030PS	11.91	542	+1.1%	6.29	683	+2.2%

Note:
NGVD29 = National Geodetic Vertical Datum, 1929
mgd = Million gallons per day

Both average stage and discharge increased when projected 2030 land use changes were combined with the USJRB restoration projects, even for full withdrawal scenarios. For the Full2030PS scenario, average stage at Cocoa increased 0.19 ft and average discharge increased 6 mgd (1.1%) from the baseline condition (Base1995NN). At Christmas, this scenario increased average stage by 0.18 ft. and average discharge by 15 mgd. The projected 2030 land use used for this study is based on BEBR 2030 population growth predictions developed in 2008. At the time the population growth predictions were made, Florida had experienced unprecedented population growth from 1995-2006. Subsequent population growth during a recessionary period of 2008-2011 was considerably lower. SJRWMD has recently received 2010 population and land use information, and these data will be developed and used to update the HSPF hydrologic models for assessing the river's hydrologic response to requested surface water withdrawals in the Consumptive Use Permit program.

It was an important result of the pattern and level of water withdrawals for the upper St. Johns River that the fraction of discharge withdrawn from the river near Lake Poinsett was greatest for mid-range discharges (Figure 3-1). The largest percent reductions in river discharge occurred near the median discharge. This effect occurs because the modeled rate of water withdrawal from Lake Poinsett was reduced at low flow to meet the requirements of the MFL downstream of this location. No water withdrawals occurred when river discharge at SR 50 (RK 343) was below 194 mgd. Withdrawals then increased from 6.39 to 84.0 mgd as river discharge increased from 194 to 414 mgd.

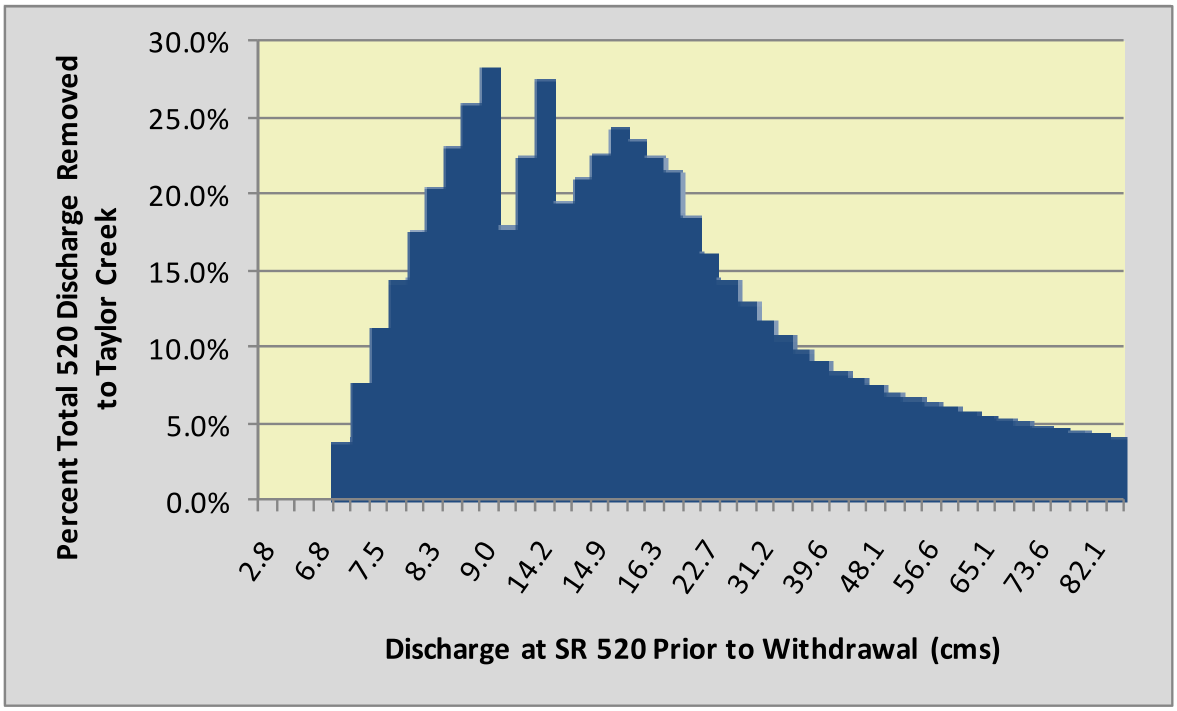

Figure 3-1. Percent of discharge at SR 520 withdrawn in the full withdrawal (Full1995PN). Median discharge for the baseline condition (Base1995NN) at this site is approximately 274 mgd (12 $m^3 s^{-1}$).

An important consequence of this withdrawal schedule is that the greatest percent reductions in river discharge occur near the median discharge. However, under this flow condition, small changes in water level will inundate or expose relatively large areas of floodplain wetlands. This effect is caused by the interaction between water level and the topography of the river and floodplain. When discharge is much below the median discharge, water levels are also low and the river is within its banks. In this condition, small variations in water levels cannot appreciably affect floodplain wetlands. For the opposite extreme, when discharge is much above the median discharge, water levels are high and the floodplain is inundated. However, under this condition, the water withdrawals represent only a small fraction of the river discharge and the effects of water withdrawals are correspondingly lessened. Thus, areas of floodplain wetlands affected by water withdrawals first increase with discharge, as the river first comes out of bank, and then decrease with discharge, as the fraction of withdrawal relative to river discharge becomes small.

Water Level and Discharge Effects Within the Lower and Middle St. Johns River and Lake George

Water level and discharge effects in the middle and lower St. Johns River and Lake George were analyzed using the EFDC hydrodynamic model. Discharge reductions have very small temporal variability in this reach first, because modeled water withdrawals from the middle St. Johns River and lower Ocklawaha River are constant in time, and second, because the effects of variable water withdrawal from the upper St. Johns River at Lake Poinsett are considerably attenuated far downstream of that withdrawal point. The remainder of this section, then, focuses on water level effects.

Forecast scenarios for this area necessarily include expected sea level rise (SLR) between 1995 and 2030. Results show that water levels in the year 2030 throughout the lower and middle St. Johns River and Lake George will increase primarily due to SLR and secondarily due to increased runoff from urbanization (Figure 3-2). Increased water levels caused by these two factors would dominate over reductions in water level due to proposed water withdrawals of 155 mgd upstream of DeLand and 107 mgd from the Ocklawaha River. At the level of the proposed water withdrawals, reduction of water levels in the lower and middle St. Johns River and Lake George is inconsequential to consideration of water withdrawal effects. Water levels throughout this reach are certain to rise to the year 2030 and beyond with or without proposed surface water withdrawals.

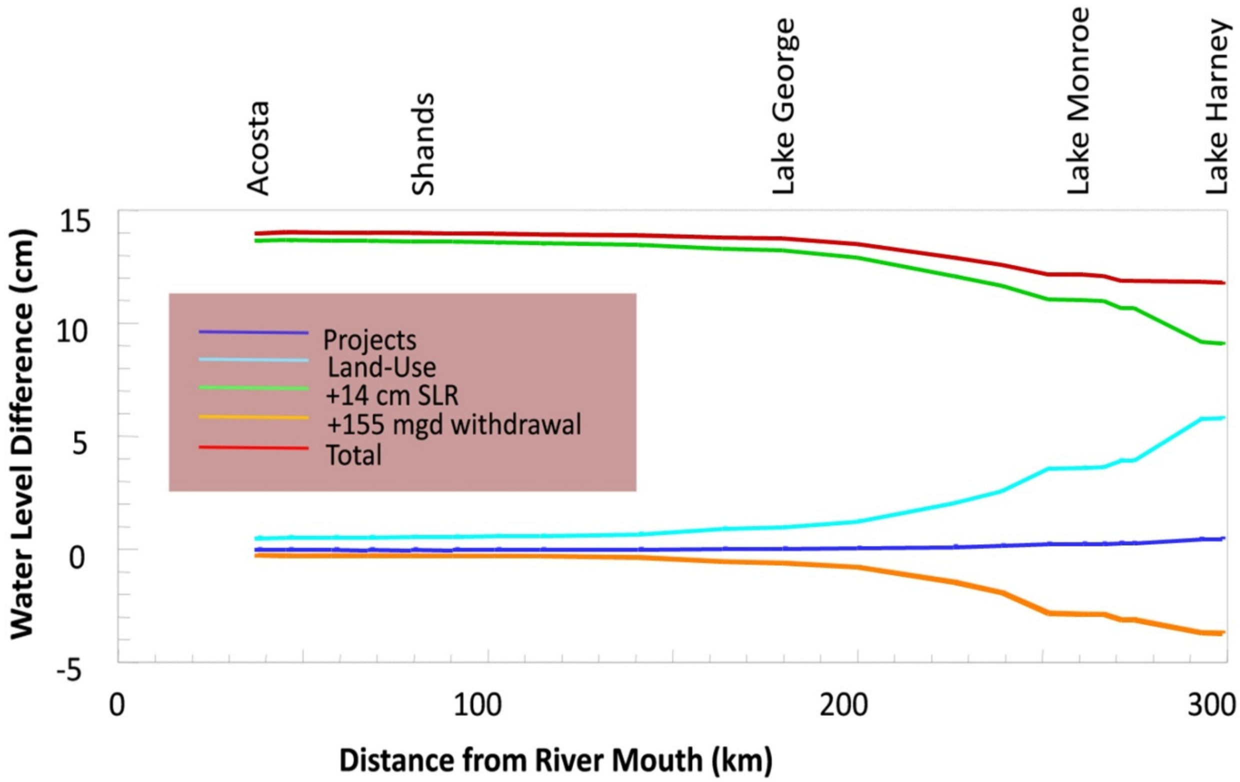

Figure 3-2. Longitudinal distribution of mean water level differences associated with the various WSIS scenarios compared with the baseline condition (Base1995NN). The first four lines show the individual effect on water level of USJRB restoration projects (Projects), 2030 land use (Land-Use), expected SLR between 1995 and 2030 (+14 cm SLR), and a 155 mgd water withdrawal (+155 mgd withdrawal). The red line is the total effect (Full2030PS) for all of the individual factors combined compared with the baseline condition (Base1995NN).

Water level changes over the lower 300 km of the river are small because this river reach is dominated by ocean water level, not discharge, when discharge is below the median discharge. Under these discharge conditions, changes in discharge have little effect on water level. Ocean influences extend far up the river because of the river's low hydraulic slope. In addition, the river's bottom slope does not drive river discharge in this reach since bottom elevations are below mean sea level to Lake Harney (Figure 3-3). Even if there were no freshwater flow into the St. Johns River at all, the main stem river and connected lakes of the lower and middle St. Johns River and Lake George would be inundated.

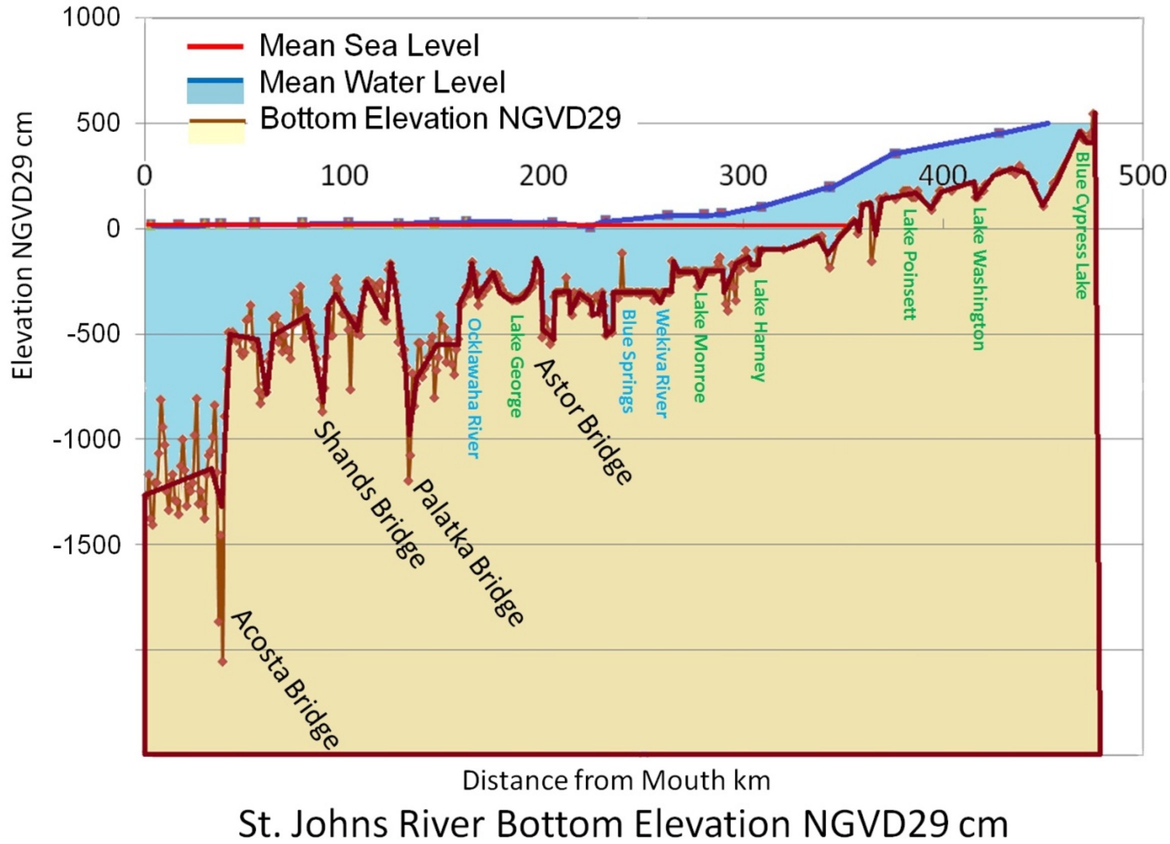

Figure 3-3. Longitudinal distribution of St. Johns River bottom elevations and mean water level.

The importance of ocean water level effects on water levels in the St. Johns River also means that the effects of SLR extend over the lower 300 km of the river, from the river mouth to Lake Harney. At present rates of SLR, water levels in the St. Johns River are estimated to increase about 14 cm between 1995 and 2030. For a high rate of SLR that considers uncertainty due to global climate change, water levels are projected to increase about 28 cm over that period. The rate of SLR in the middle St. Johns River is lower than in the lower St. Johns River because the full amount of SLR is not realized in the Middle St. Johns River Basin under high flow conditions. However, under high flow conditions increased runoff due to urbanization of watersheds also increases water levels in the middle St. Johns River. As a result, modeled water levels rise throughout the lower and middle St. Johns River over all flow conditions by the year 2030.

Among the future conditions we examined, only SLR materially affected water levels in the middle and lower reaches. An additional water withdrawal of 107 mgd from the Ocklawaha River has essentially no effect on water levels in the St. Johns River because the Ocklawaha River enters downstream of Lake George, where levels are determined by the ocean level more than discharge. Channel deepening and reuse of wastewater also have inconsequential effects on river water levels.

3.1.3 Effects of Scenario Conditions on Forcings – Salinity

From the perspective of water supply, the lower and middle St. Johns River including Lake George have essentially infinite storage capacity because the river surface is nearly at sea level and cannot be drawn down lower than sea level. Withdrawing water from a closed reservoir in excess of resupply will eventually deplete the reservoir, whereas deficit withdrawal of water from the lower or middle St. Johns River cannot deplete the river. Instead, deficit withdrawal would pull water into the lower St. Johns River from the ocean and increase salinity in the estuarine reach. Water withdrawals from the lower and middle St. Johns River also decrease flushing which could increase salinity in areas of the river upstream of the estuarine reach where salinity is dominated by groundwater inflow.

Modeled changes to salinity in areas upstream of the estuarine reach are small for all scenarios. Modeled salinity in Lake George and the middle St. Johns River typically varied less than 0.05 psu among scenarios. The oligohaline character of the middle St. Johns River is biologically important, but we found no evidence that it would appreciably change due to modeled water withdrawal or any of the other factors considered for the WSIS. The only possible change to salinity in the middle St. Johns River that is related to water withdrawal would be the return of reject water from reverse osmosis, if that technology were required for water treatment. This localized effect is discussed below and in detail in Chapter 5. River Hydrodynamics Calibration.

The estuarine reach of the river extends over the lower 80 km of river from the river mouth to Shands Bridge. The upper portions of the estuarine reach, between Buckman Bridge and Shands Bridge, are typically fresh to oligohaline, with salinity below one psu, but experience infrequent ocean incursions of higher salinity during droughts. Modeled salinity in the estuarine reach of the river increased with increased water withdrawals and SLR. It was decreased by increased discharge from 2030 land use changes and the USJRB restoration projects.

Increased discharge from 2030 land use changes and water withdrawal has the greatest effects on estuarine salinity for potential scenarios. However, because these two factors work in opposition, the salinity change was negligible when comparing forecast withdrawal scenarios (e.g. Full2030PS) with the baseline condition (Base1995NN). Mean salinity at Acosta Bridge increased only 0.04 psu between these two scenarios, while 10-yr high salinity events at Shands Bridge increased only 0.35 psu for 1- to 30-day durations.

Modeled salinity effects in the estuarine reach for a 155 mgd withdrawal were small even in isolation (Figure 3-4). The isolated effect of a 155 mgd withdrawal (Full1995NN) increased mean salinity 0.1 – 0.3 psu between Shands Bridge and Acosta Bridge compared with the baseline condition (Base1995NN). This level of increase is very small relative to the dynamic range of salinity conditions occurring in this reach of the river where salinity varies from 0.3 to over 20 psu. Because differences in mean values might obscure more important changes to high salinity events, we also examined changes to 2-, 5-, and 10-yr high salinity events over a range of durations. Salinity changes were remarkably similar for high salinity events over a wide range of both frequency and duration. Salinity levels increase 0.4 – 0.65 psu for high salinity events with frequencies ranging from 2- to 10-years and durations ranging from 1-day to 1-year. A 155 mgd withdrawal, then, produced no anomalous, large shifts in salinity.

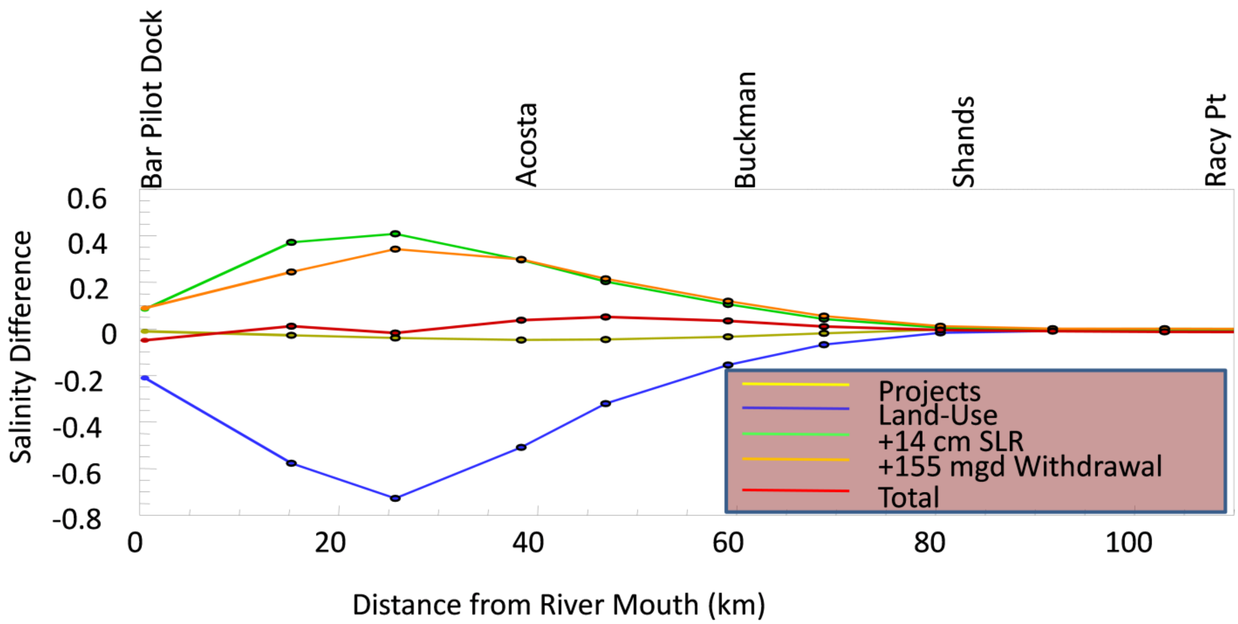

Figure 3-4. Longitudinal distribution of salinity differences associated with the various WSIS scenarios compared with the baseline condition (Base1995NN). The first four lines show the individual effect on salinity of USJRB restoration projects (Projects), 2030 land use (Land-Use), expected SLR between 1995 and 2030 (+14 cm SLR), and a 155 mgd water withdrawal (+155 mgd withdrawal). The red line is the total effect (Full2030PS) for all of the individual factors combined compared with the baseline condition (Base1995NN).

Water withdrawal from the middle St. Johns River for public water supply could require treatment to remove excessive chloride and other salts. Reverse osmosis is a common treatment that generates a waste stream of concentrated salts. The discharge of this waste stream back to the river as reject water could increase salinity near the discharge locations. A far-field analysis of the effects of reject water on salinity in the middle St. Johns River shows that discharge of reject water from a 50 mgd water withdrawal into the narrow river channel between Lake Monroe and Lake Jesup could increase salinity from a background level of 1.5 to 3 psu during drought periods. Discharge of reject water at a location below Lake Monroe (Yankee Lake) has less effect on salinity. These results indicate that the far-field effects of reject water on salinity should be considered for design and permitting of reverse osmosis plants in the middle St. Johns River.

3.1.4 Effects of Scenario Conditions on Forcings – Water Age

Water age differences between the baseline condition and hindcast and forecast scenarios are small (~5 days) relative to the large natural variation (20 to 200 days) of water age for the river. The greatest change to water age due to a 155 mgd withdrawal occurs in Lake George. Water age differences are greatest when absolute water age is high, and water age differences caused by a 155 mgd withdrawal only rarely exceed 10% of ambient water age. The 10-yr high water age event in Lake George increases 10 – 14 days for durations of 1-day to 1-year. Events that are more moderate show less of an absolute increase in water age; the 2-yr high water age for the same range of durations increases about 6 days. The percent increase of water age remains at 4 – 5% over all frequencies and durations of high events.

For forecast scenarios, the increase in water age due to a water withdrawal is offset by a decrease in water age due to increased runoff from 2030 land use. Mean water age in Lake George under these watershed conditions with a 155 mgd water withdrawal increases only 2.7 days compared with 1995 baseline conditions.

3.1.5 Effects of Scenario Conditions on Forcings – Ecological Considerations

The environmental workgroups determined which aspects of hydrology and hydrodynamic factors were most strongly associated with ecological effects. For example, the Littoral Zone workgroup determined that the strongest effects on SAV would be associated with the 7-day maximum mean salinity. These ecologically potent factors became the drivers for evaluation of effects on ecological attributes, such as the distribution of SAV.

Variation Among Scenarios

A prominent finding of our analysis is that the deviations of H&H drivers from the base condition, or forcings, were generally small for all scenarios (Table 3-4, Table 3-5, Table 3-6, and Table 3-7). Further, reduction effects were reduced or eliminated in the forecast scenarios and replaced by augmentation effects.

Scenario rankings show that, generally, the largest deviations from the baseline condition were associated with the most extreme test scenarios (FwOR1995NN and FwOR1995NS). Modeling indicates that completion of the USJRB restoration projects will significantly reduce the effects of water withdrawals on H&H drivers. Moreover, the modeled effect of projected land use changes from 1995 to 2030 was an increase in discharge that often exceeded the modeled withdrawals. Consequently, scenarios that combined these two conditions had few to no modeled reduction effects. Scenarios that lacked reduction effects were not subjected to hydroecological analysis. Importantly, many of these scenarios more closely approximate the conditions that will exist at the time when water withdrawals could occur.

Table 3-4. Ranking of reduction effects of withdrawal scenarios based on deviations (m^3/s) from the base scenario (Base1995NN) in discharge. Discharge used was the sum of discharges for the river at DeLand and for the Ocklawaha River. Unshaded row is a test scenario; shaded rows are hindcast or forecast scenarios.

Scenario	Rank in Freshwater Segments (4-8)	Rank in Estuarine Segments (1-3)	Deviation from Mean Baseline Discharge (%) Dec-May	Deviation from Mean Baseline Discharge (%) June-Nov.
Full1995NN	1	1	-5.80	-7.56
Full1995PN	2	2	-5.49	-5.88
FwOR2030PS	3.5	3	-5.88	+1.25
Half1995PN	5	4	-2.54	-2.29
Full2030PS	3.5	5	-1.27	+6.04
Half2030PS	6	6	+1.33	+10.04

Table 3-5. Ranking of reduction effects of withdrawal scenarios based on deviations (cm) from the base scenario (Base1995NN) in mean annual lake stage in Lake Poinsett (upper St. Johns River) and Lake Harney (middle St. Johns River). For these lakes, the FwOR and Full withdrawal scenarios were equivalent. NR indicates a scenario lacked reduction effects and was not ranked. Unshaded rows are test scenarios; shaded rows are hindcast or forecast scenarios.

Scenario	Rank	Deviation (Water Level, cm) Lake Poinsett	Deviation (Water Level, cm) Lake Harney
Full1995NN	1	-5	-4
Half1995NN	2	-3	-2
Full1995PN	3	1	-3
Half1995PN	4	4	-1
Full2030PN	NR	6	2
Half2030PN	NR	8	4
Full1995PS	NR	1	6
Full2030PS	NR	6	12
Half2030PS	NR	8	14

Table 3-6. Ranking of reduction effects of withdrawal scenarios based on deviations (days) from the base scenario (Base1995NN) in one-day maximum water age at Racy Point. Water age for Racy Point is representative of the lower St. Johns River and Lake George. NR indicates a scenario lacked reduction effects and was not ranked. Unshaded rows are test scenarios; shaded rows are hindcast or forecast scenarios.

Scenario	Rank	Deviation (Δ Water Age Racy Pt) (days)
FwOR1995NN	1	13
FwOR1995PN	2	12
FwOR2030PS	3	11
FwOR2030PN	4	5
Full1995NN	5	5
Full1995PN	6	4
Full2030PS	7	3
Half1995NN	8	3
Half1995PN	9	1
Half2030PS	10	0
Full2030PN	NR	decrease
Half2030PN	NR	decrease
Full1995PS	NR	decrease

Table 3-7. Ranking of reduction effects of withdrawal scenarios based on deviations (psu from the base scenario (Base1995NN) in maximum 7-day salinity near the downstream edge of present day SAV distribution. The 7-day maximum salinity for the base scenario was 15.46 psu. NR indicates a scenario lacked reduction effects and was not ranked. Unshaded rows are test scenarios; shaded rows are hindcast or future scenarios.

Scenario	Rank	Deviation (psu)
FwOR1995PS	1	0.95
FwOR1995NN	2	0.90
FwOR1995PN	3	0.81
Full1995PS	4	0.55
Full1995NN	5	0.49
Full1995PN	6	0.40
FwOR2030PS	7	0.33
Half1995PS	8	0.30
Half1995NN	9	0.23
FwOR2030PN	10	0.18
Half1995PN	11	0.14
Full2030PS	NR	-0.06
Base1995PN	NR	-0.10
Full2030PN	NR	-0.22
Half2030PS	NR	-0.31
Half2030PN	NR	-0.48

Longitudinal Variation

There was significant variation in the responsiveness (relative deviations) of H&H drivers along the river's length (Figure 3-5, Figure 3-6, and Figure 3-7). The modeled potential for salinity deviations was greatest at river kilometer (RK) 50, while deviations in residence time peaked further upstream (ca. RK 125). The relative potential for deviations in water levels peaked much further upstream, reaching a maximum at about RK 360. The potential for entrainment and impingement of ichthyoplankton peaks at about RK 300 for all fish species combined and at about RK 340-350 for American shad (Figure 3-5 and Figure 3-7). The area of the river showing the lowest relative deviations in all drivers was the reach extending from about RK 200 – 250. No area of the river, however, was completely free of deviations in drivers.

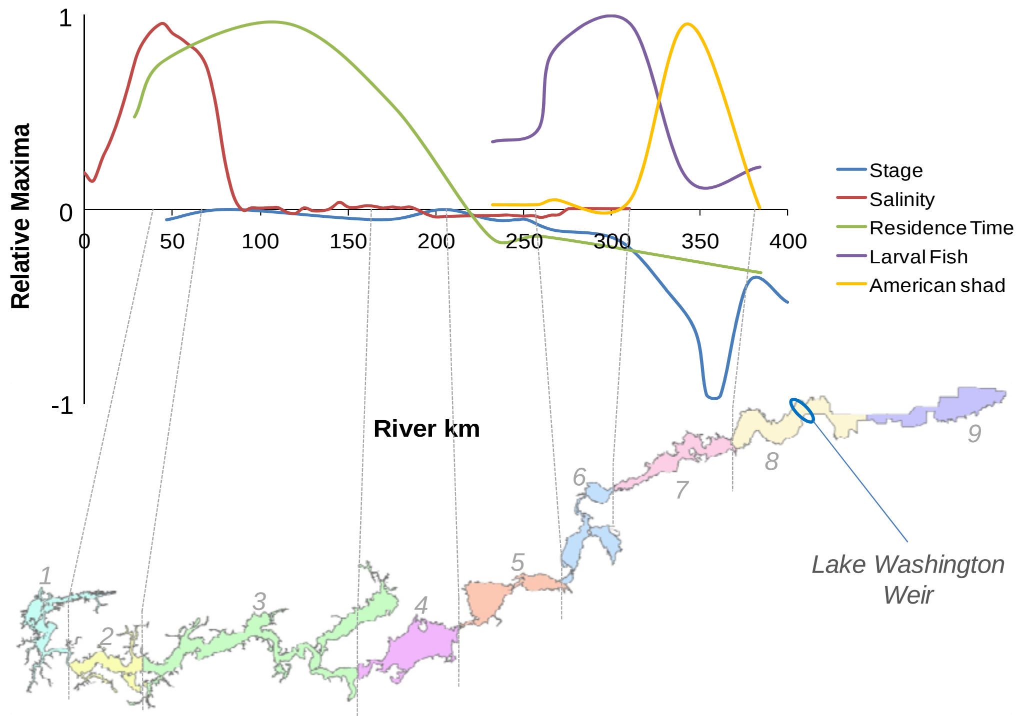

Figure 3-5. Variation in fraction of maximum deviation from the baseline condition (Base1995NN) for maximum monthly median stage, 7-d maximum salinity, and residence time (as indicated by the 90th percentile for water age in the middle and lower St. Johns River and the 90th percentile for residence time in the upper St. Johns River) and the potential for entrainment and impingement effects as indicated by the relative maximum densities of planktonic stages for all fish and for American shad.

Our analysis indicates that the potential for deviations in water levels (or stage) peaks in segment 7 near RK 360 and diminishes both upstream and downstream of that segment. Downstream of segment 7, the potential for deviations approaches zero near RK 200 where the effect of ocean level begins to dominate other factors influencing river stage at and below the median stage (see above and Chapter 6. River Hydrodynamics Results). For this reason, water level effects would not be expected to be appreciable downstream of segment 5. Upstream of segment 7, effects would cease at the Lake Washington weir because the weir hydrologically separates upstream and downstream waters at levels below the fixed-crest weir.

Material deviations in salinity would not occur upstream of the Shands Bridge (RK 80.5) or downstream of the John T. Alsop “Mainstreet” bridge (RK 38.0). Thus, it is only within this 42 km stretch of the river that there is an appreciable potential for biological effects caused by deviations in salinity. Farther upstream, ocean influence on salinity is very slight and farther downstream the salinity regime is strongly dominated by tides. Effects on residence time could be widely distributed. The modeled potential for deviations in residence time was significant over nearly 150 km of the river's length.

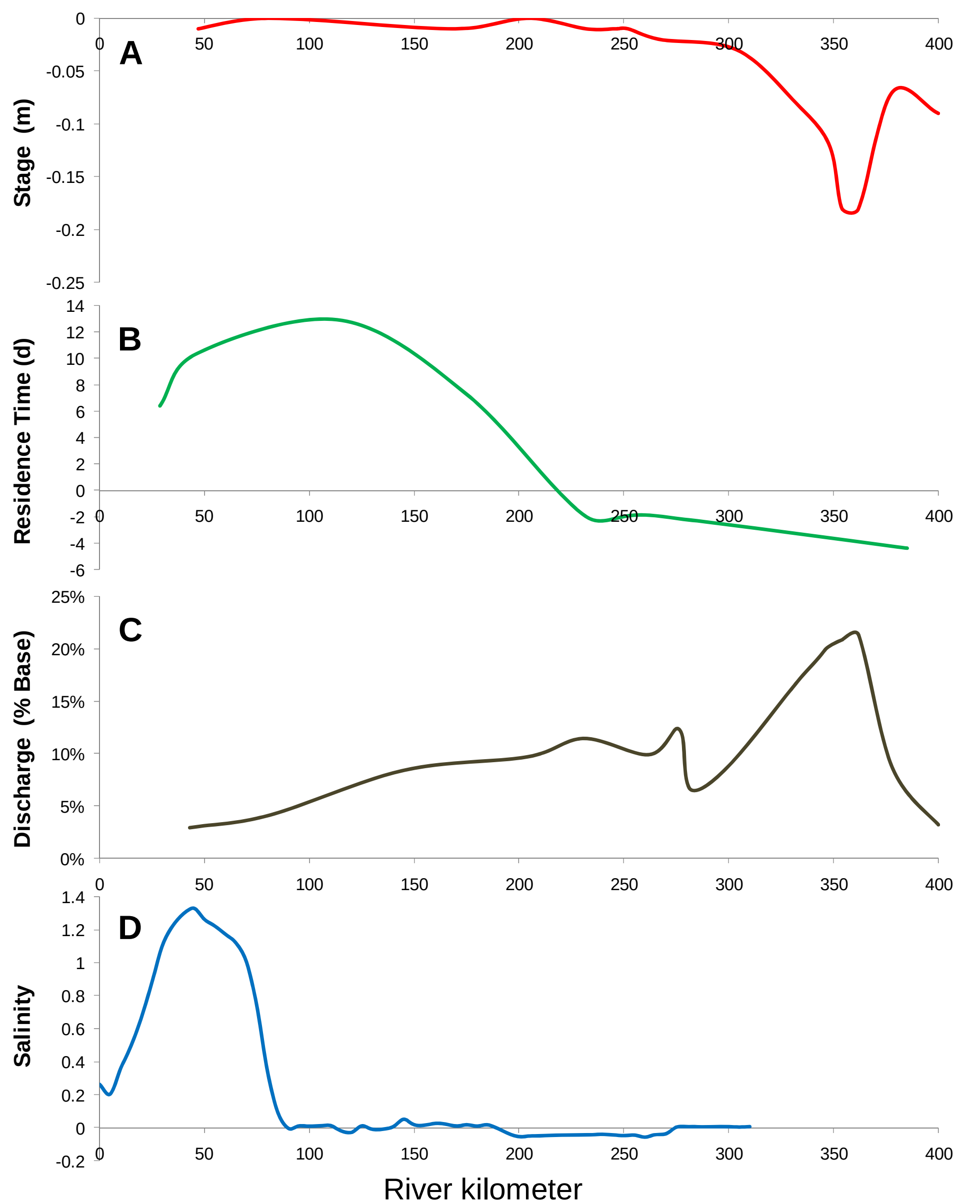

Figure 3-6. Longitudinal variation in maximum modeled deviation from the base condition (Base1995NN) for H&H drivers for extreme scenarios (FwOR1995NN, downstream of the Ocklawaha River; Full1995NN, upstream of the Ocklawaha River). Hydrologic drivers are (A) maximum monthly median June stage, (B) median 7-day residence time, (C) difference in median June discharge as a percentage of base modeled discharge, and (D) 7-day maximum salinity.

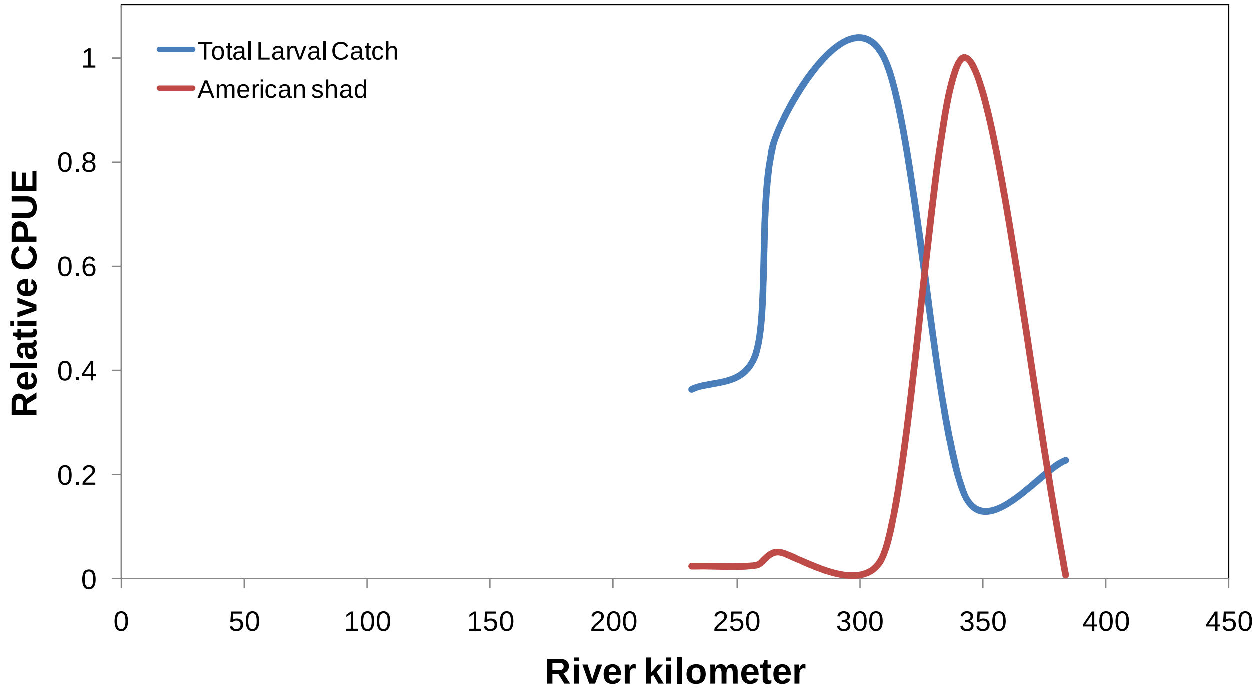

Figure 3-7. Longitudinal variation in the potential for entrainment and impingement of ichthyoplankton as indicated by relative larval fish catch per unit effort along the St. Johns River. One line represents the total larval fish catch and the other represents only American shad. Relative values were calculated from the number of larvae caught per sample transect.

Effects on Areal Extent of Salinity Zones

Modeled salinity deviations were greatest in the estuarine segments (1-3). To a great degree, the ecological effect of these deviations would depend upon whether the upstream movement of salinity isopleths caused a reduction in the areal extent of some salinity zones. In fact, there was little change in these areas, even under the most extreme test scenario (Figure 3-8; for details see Chapter 12. Fish).

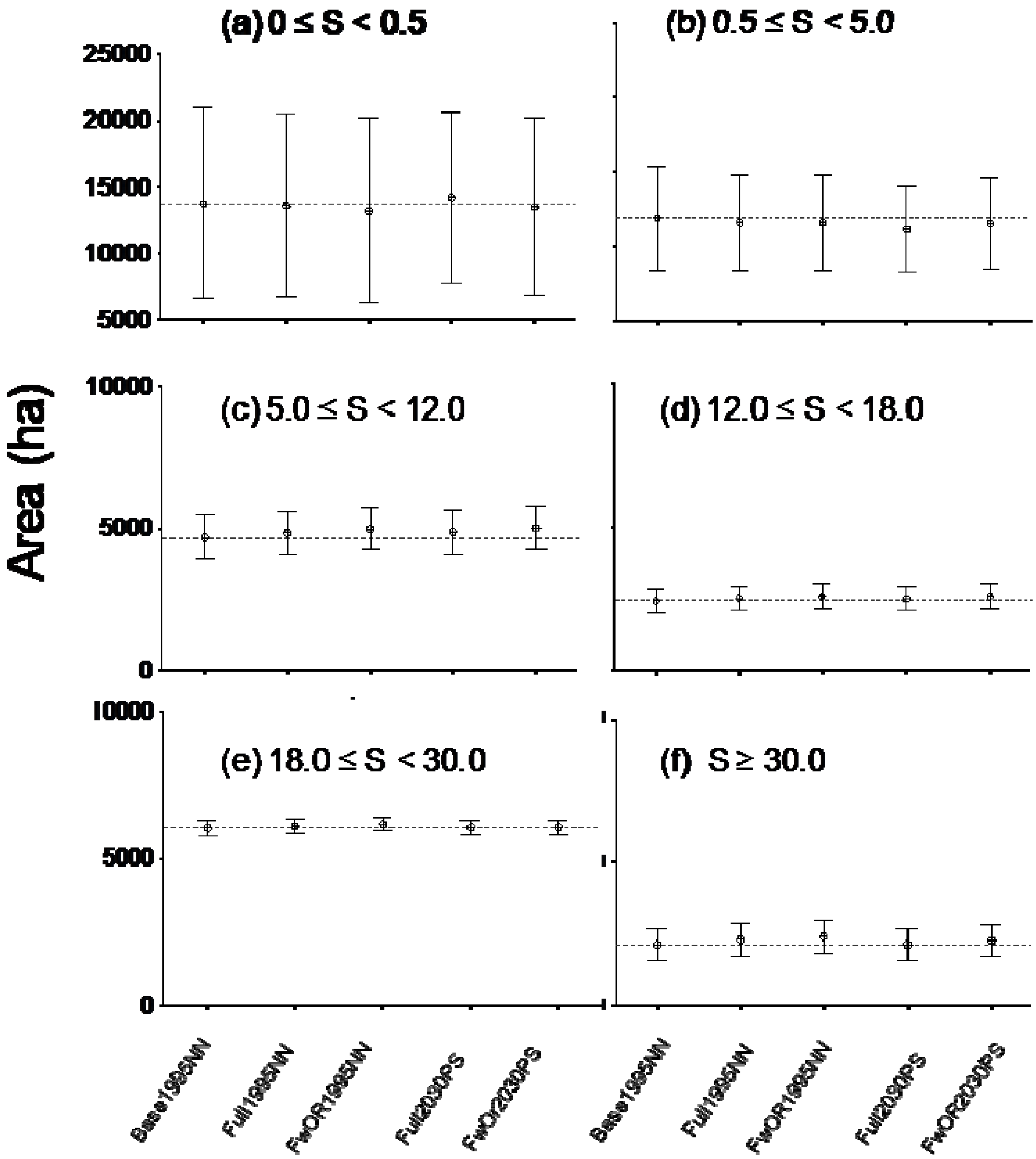

Figure 3-8. Mean annual areal coverage (ha) ±95% CI of various salinity habitat units (S, psu) in the lower St. Johns River estuary for five scenarios. Mean annual values were derived for the years 1996-2005 based on the same period of highest mean 30-day salinity as the baseline condition (Base1995NN). Salinity habitat zones are those used in the analysis of fish population data by the Florida Fish and Wildlife Conservation Commission; (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 baseline condition.

3.2 Deviations in Ecological Attributes (Effects)

The relatively small deviations in H&H drivers for hindcast and forecast scenarios indicate that expected effects on ecological attributes should not be major. This was confirmed by the results of the HE models used to calculate deviations in key ecological attributes.

3.2.1 Biogeochemical Attributes

The Biogeochemistry Workgroup developed a suite of models for simulating the deviation in loadings to the river of constituents released by oxidation of organic wetland soils. The deviations in loadings would be associated with increased durations of exposure of soils due to reduced water levels. The consequences of the change in loadings were evaluated by calculating the change in concentration of the constituent in outflowing water (details in Chapter 7. Biogeochemistry). The potential for this effect is strongest in segment 8, which has extensive organic wetland soils and a relatively large degree of deviation in water levels. In segment 8, the most extreme scenario was Full1995NN (a test scenario), with a 5 cm decrease in mean annual water level at Lake Poinsett (Table 3-5).

Experimental data and soil analyses indicate that the organic soils in segment 8 are resistant to oxidation. As a result, we found small deviations in release rates associated with deviations in days of soil exposure. The corresponding simulated releases of nitrogen and phosphorus were negligible even under the most extreme test scenario.

The workgroup also examined the potential for reduction of dissolved oxygen concentrations due to increased loadings of dissolved organic carbon. The modeled potential was quite low. In the most extreme test scenario (Full1995NN), and using high estimates of increased DOC loading, simulated deviations did not exceed -0.14 mg L^{-1} at the extreme of the range.

3.2.2 Plankton Attributes

The Plankton Workgroup examined the potential for deviations in five ecological attributes associated with phytoplankton blooms:

1.) Marine blooms (as indicated by maximum annual dinoflagellate biovolume) in segment 2

2.) Nitrogen loads (annual mass nitrogen added by nitrogen fixation) in segments 3 and 4

3.) Freshwater bloom magnitude (as indicated by the concentration of chlorophyll-a) in segments 2, 3, 4, 6, and 8

4.) Freshwater dissolved oxygen depletion (also indicated by the concentration of chlorophyll-a) in segment 2-4

5.) Freshwater bloom duration (duration of longest annual bloom) in segments 2, 3, 4, 6, and 8.

In segments that were not assessed, monitoring data were insufficient to develop hydroecological models.

The workgroup developed multiple linear regression models, using two to seven independent variables, with good fits to empirical data (adjusted R^2 of 0.80 to 0.97) (Table 2-1; details in Chapter 8. Plankton). The results of these regression models were supported by a mechanistic water quality model (CE-QUAL-ICM). Deviations in residence time indicated that the most extreme scenario was FwOR1995NN, a test scenario (Table 3-6). Under this most extreme scenario, modeled deviations in all attributes were modest (Table 3-8). For most areas, deviations did not exceed +3.6% for bloom magnitude or +0.1 for annual frequency. For FwOR2030PS, the most extreme of the hindcast and forecast scenarios, modeled effects were small and, in many cases, negative. Negative effects indicate a reduction in the potential for phytoplankton blooms, probably associated with the increase in discharge associated with 2030 watershed conditions.

Table 3-8 Modeled deviations from the baseline condition for ecological attributes associated with phytoplankton blooms for scenario base FwOR1995NN. In most cases, the deviations in bloom attributes were expressed both as the percent change in the median annual value (withdrawal minus baseline) of the attribute and as the change in annual frequency that thresholds for the attribute were exceeded. In some cases, the effect of withdrawal could be expressed only as the change in annual frequency of exceedence. See Chapter 8. Plankton for further detail.

River Segment*	Location	Algal Bloom Metric				
		Marine Algal Blooms	Change in N Load	Magnitude of Freshwater Algal Blooms	Magnitude of Freshwater Algal Blooms (DO)	Duration of Freshwater Algal Blooms
2	Mandarin Point	± 0.0				
	Doctors Lake	+ 0.09		+ 3.1% - 0.1	+ 3.1% ± 0.0	+ 3.6% ± 0.0
3	Racy Point		+ 0.1	+ 1.0% + 0.1	+1.0 % ± 0.0	+ 0.06
4	Lake George		+ 0.1	- 1.8% ± 0.0	- 1.8% ± 0.0	+ 0.04
6	Lake Monroe			+ 0.6% ± 0.0		- 2.9% ± 0.0
	Lake Jesup			- 1.7% ± 0.0		- 2.4% - 0.1
	Lake Harney			+ 7.6% + 0.1		
8	Lake Poinsett			+ 1.3% ± 0.0		

*Segments 1, 5, 7, and 9 were not assessed by the Plankton Working Group.

3.2.3 Submersed Aquatic Vegetation Attributes

The Submersed Aquatic Vegetation Workgroup assessed the potential for effects on submersed aquatic vegetation stemming from salinity stress in the estuary and from water level declines in the freshwater reaches (details in Chapter 9. Submersed Aquatic Vegetation).

With respect to salinity, they developed a stress model for *Vallisneria americana* (common name: eelgrass or tape grass), the dominant species of SAV in the estuary. The stress model related levels of stress (none, low, moderate, and extreme) to the duration of exposure to levels of salinity from 3 to 25 psu. Using an area most likely to experience salinity stress, they used deviations in 7-day maximum salinity to rank scenarios. Scenario FwOR1995PS (an extreme test scenario) had a seven-day maximum salinity of 16.41, a deviation of 0.95 from the baseline condition (Table 3-7). In this most extreme test scenario, 4% (199 ha) of the potential SAV habitat in the estuary would experience some degree of increased salinity stress (Figure 3-9).

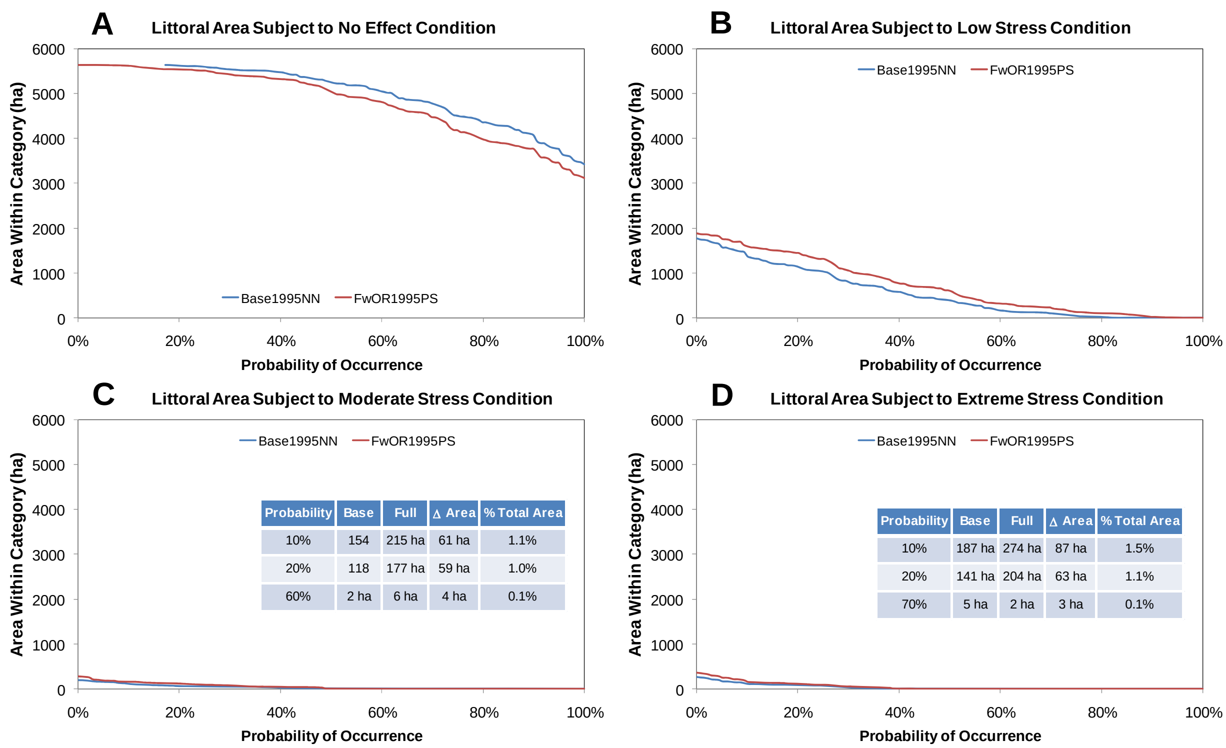

Figure 3-9. Cumulative probability of individual cell areas experiencing (A) No Effect, (B) Low Stress, (C) Moderate Stress, and (D) Extreme Stress categories. Each panel shows curves for both the base scenario (Base1995NN, blue) and full top-ranked scenario (FwOR1995PS, red). The full model run (140 littoral cells × 3652 model days) was considered for each scenario. Inset tables show various probability levels and associated differences in predicted areas under each scenario.

Most of this area (162 ha) would move from no stress to low stress. This effect would occur 90% of the time over the simulated 10-year period. At higher stress levels, the areas affected and the percentage of time the effects would intensify diminished. Eighty-seven hectares were exposed to extreme stress and 61 hectares to moderate stress for 10% of the time. Thus, the most extreme scenario caused a low risk of high levels of stress to a small area for a small percentage of the time.

The workgroup also examined the potential for decreases in water levels to reduce the depth distribution of SAV. By relating deviations in water level to bottom bathymetry of lakes Poinsett and Harney, the workgroup modeled the potential change in the depth distribution of SAV associated with simulated deviations in water levels. The Full1995NN test scenario was the most extreme scenario for this analysis (Table 3-5). Under this scenario, the bottom area with an average depth of 85 cm or less during the growing season increased in Lake Poinsett by 9% (89 ha) and decreased in Lake Harney by 2% (9 ha) (Table 3-9).

Table 3-9. Potential SAV habitat in two scenarios and the percent difference from the baseline condition (Base1995NN) for extreme test scenario Full1995NN. Calculated from the stage-area curve for Lakes Poinsett and Harney. Areas were calculated based on the assumption that the depth limit of SAV habitat was 85 cm.

Lake	Scenario		Difference
	Base1995NN	Full1995NN	
Poinsett	989 ha	1,078 ha	9%
Harney	391ha	382 ha	-2%

3.2.4 WETLAND VEGETATION ATTRIBUTES

The Wetland Vegetation Workgroup examined the potential for deviations in the areal extent and community proportionality of wetland plant communities in segments 7 and 8, where simulated deviations in water levels were greatest, and in segment 2, where freshwater wetland plant communities would be expected to experience the greatest salinity stress. In both cases, they used empirical data to develop community boundaries in terms of hydrology (water level and soil elevation were used to calculate percent exceedence of soil surface elevation; segments 7 and 8) or soil salinity (95th percentile high salinity in river; segment 2) (details in Chapter 10). In segments 7 and 8, MFL transects (Mace 2007a, b) were used to relate vegetation community boundaries to long-term hydrology. A survey of vegetation along a soil salinity gradient in the Ortega River was used to discern the salinity-based community boundaries.

In segments 7 and 8, the simulated deviations in percent exceedance could be related to surveyed transect elevations to model the movement of community boundaries. Using a GIS digital elevation model, changes in the areal extent of floodplain wetland plant communities also could be modeled. For water levels, Full1995NN (a test scenario) was the most extreme scenario.

Modeled deviations for wetland plant communities were larger than those for the preceding three ecological components (Biogeochemistry, Plankton, and SAV). Where modeled deviations in water levels were greatest (segment 8), there were material modeled changes in the proportionality of wetland communities for the most extreme test scenario (Full1995NN; Table 3-10 and Table 3-11) but there was no appreciable modeled change in total wetland area.

Table 3-10. Area of wetland types for the Lake Poinsett pool of segment 8 under the baseline condition (Base1995NN) and the most extreme test scenario (Full1995NN) and the change in area between scenarios. (Key: DM = Deep Marsh, SM = Shallow Marsh, SS = Shrub Swamp, WP = Wet Prairie, UWP = Upper Wet Prairie, TS = Transitional Swamp, HS = Hardwood Swamp, HH = Hydric Hammock.)

Vegetation type	base hectares	Base %	Full95NN hectares	Full1995NN %	Change (ha)
Open Water	101	1.2	83.4	1.0	-17.6
DM	479.1	5.8	453	5.5	-26.1
SM	4071.9	49.2	3613.6	43.7	-458.3
SS	279.7	3.4	180.2	2.2	-99.5
WP	1788.4	21.6	2450.3	29.6	661.9
UWP	172.2	2.1	264.4	3.2	92.2
TS	112.8	1.4	104.4	1.3	-8.4
HS	703.2	8.5	640.9	7.8	-62.3
HH	469.3	5.7	490.2	5.9	20.9
Berms	92.1	1.1	92.1	1.1	0
Total	8269.7	100.0	8372.5	101.2	102.8

Table 3-11. Area of wetland types for the lake Poinsett pool of segment 8 under the baseline scenario (Base1995NN) and a potential near-term scenario (Full1995PN) and the change in area between scenarios. (Key: DM = Deep Marsh, SM = Shallow Marsh, SS = Shrub Swamp, WP = Wet Prairie, UWP = Upper Wet Prairie, TS = Transitional Swamp, HS = Hardwood Swamp, HH = Hydric Hammock.)

Vegetation type	Base hectares	Base %	Full95PN hectares	Full1995PN %	Change (ha)
Open Water	101	1.2	104.3	1.3	3.3
DM	479.1	5.8	475.7	5.8	-3.4
SM	4071.9	49.2	3787	45.8	-284.9
SS	279.7	3.4	84.7	1.0	-195
WP	1788.4	21.6	2243.1	27.1	454.7
UWP	172.2	2.1	377.9	4.6	205.7
TSS	112.8	1.4	119.6	1.4	6.8
HS	703.2	8.5	567.4	6.9	-135.8
HH	469.3	5.7	464	5.6	-5.3
Berms	92.1	1.1	91.4	1.1	-0.7
Total	8269.7	100.0	8315.1	100.5	45.4

Modeled reduction effects also occurred for the other hindcast scenario (Half1995PN). No modeled reduction effects were associated with the Full2030PN forecast scenario, with the exception of very slight decreases in the elevation of minimum boundary elevations for mid-elevation communities (Figure 3-10).

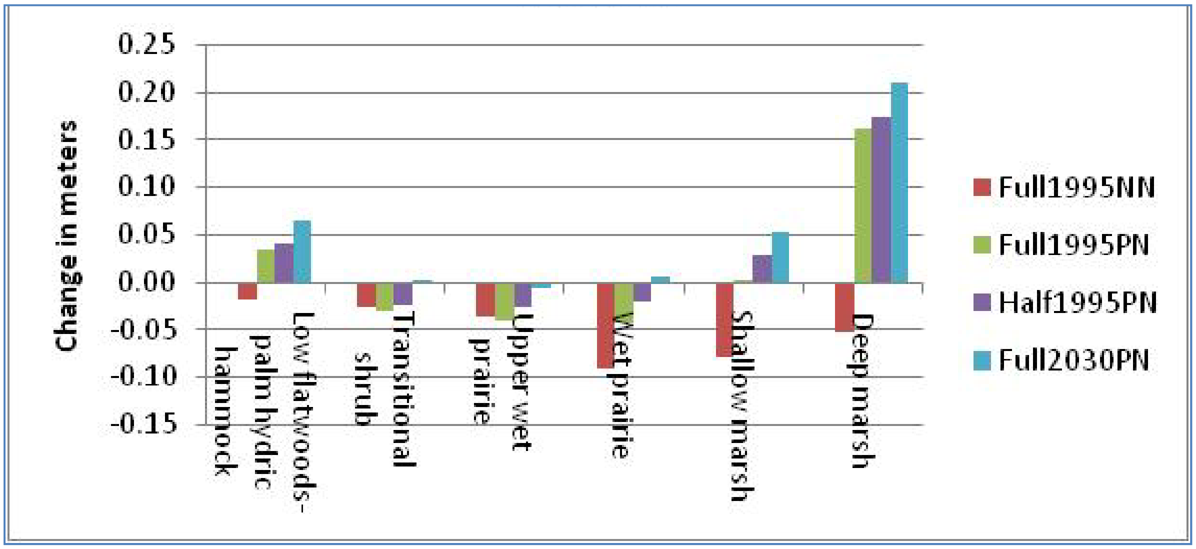

Figure 3-10. Modeled reductions in minimum community boundary elevations for the County Line transect, near Lake Poinsett, under various water withdrawal scenarios.

In river segment 8, the Full2030PN forecast scenario had reduction effects on small fractions of the areas assessed - ranging from 0.06 % to 0.90 % of the total modeled area (Table 3-12).

Table 3-12. Wetland areas affected at various levels of water level exceedance for the Full2030PN scenario in river segment 8.

FULL2030PN Exceedence (%)	Area Impacted (ha)	Reduction in Days Inundated (maximum over 10 yrs)	Area Impacted (percent of total area)	Maximum Hectare-days Impact (over 10 yrs)
5	19.4	48	0.11	932
10	11.5	46	0.06	527
15	72.9	37	0.40	2,696
20	6.2	10	0.03	62
25	54.5	34	0.30	1,853
30	0	0	0	0
35	25.1	13	0.14	327
40	140.9	46	0.77	6,479
45	164.4	41	0.90	6,739
50	157.3	57	0.86	8,966
55	45.1	36	0.25	1,623
60	0	0	0	0
65	0	0	0	0
70	0	0	0	0
75	0	0	0	0
80	0	0	0	0
85	0	0	0	0

Because the modeling of wetland vegetation effects assumed a perfect correspondence between water levels in the river and those in the wetlands, the modeled effects were overestimates of the potential effects. This is because drainage of water from the wetlands becomes progressively impeded by hydrologic roughness as water levels fall. Moreover, local topography can pool water, preventing drainage and collecting rain water.

Using the empirically determined soil salinity boundaries, the modeled deviations in salinity isopleths would cause upstream movements of the boundaries by 2.8 to 3.3 km under the most extreme test scenario (Full1995NN; Table 3-13). Modeled deviations in the boundaries were much smaller under the Full2030PS forecast scenario.

Table 3-13. Soil salinity (psu) breakpoints (plant community boundaries), breakpoint locations (river kilometers) and the modeled upstream movement (km) of breakpoints under various water withdrawal scenarios.

Soil Salinity Breakpoint	River Salinity Breakpoint	River Kilometer of Breakpoint Base1995NN	River Kilometer of Breakpoint Full95NN	River Kilometer of Breakpoint FwOR1995NN	River Kilometer of Breakpoint Full2030PS	Distance moved (km) FwOR1995NN - Base1995NN	Distance moved (km) Full95NN-Base 1995NN	Distance moved (km) Full2030PS - Base1995NN
0.47	3.21	63.45	64.87	66.28	63.66	2.83	1.42	0.21
1.53	4.13	60.28	61.87	63.38	60.46	3.1	1.59	0.18
2.44	4.93	57.58	59.26	60.88	57.68	3.3	1.68	0.1
3.41	5.77	54.90	56.59	58.24	54.83	3.34	1.69	-0.07

3.2.5 BENTHIC MACROINVERTEBRATES ATTRIBUTES

The Benthic Macroinvertebrates Workgroup examined the potential for deviations in the density and composition of benthic macroinvertebrates in the estuary and in freshwater reaches. In the estuary, non-linear regressions were used to relate density and diversity to salinity (e.g., Figure 3-11; details in Chapter 11. Benthic Macroinvertebrates).

The regression models indicate four different responses to salinity in terms of abundance: (1) a decrease in response to increasing salinity; (2) an increase in response to salinity; (3) no response; and (4) declines on either side of an optimum salinity (optimum value) (Table 3-14).

Table 3-14. Types of responses to salinity as indicated by non-linear regression models of abundance versus salinity for the St. Johns River estuary. Asterisks indicate significant regressions (salinity v. indicated taxon) ($p<0.05$). Summarized from Montagna et al., 2011.

Response to Salinity Increase	Taxon
Decrease*	Arachnida, Crustacea, Insecta, Mollusca, Oligochaeta, Chironomidae, *Apocorophium lacustre*, *Limnodrilus* sp., *Polypedilum* spp.
Increase*	Cnidaria, Holothuroidea, Nemertea, Ophiuroida, Phoronida, Urochordata
Optimum Value*	Balanidae, Corophiidae, Dreissenidae, Hydrobiidae, Mactridae, *Balanus* sp., *Cladotanytarsus* sp., *Corophium* sp., *Littoridinops* sp., *Marenzelleria viridis*, *Mulinia lateralis*, *Mytilopsis leucophaeata*, *Polydora* sp., *Rangia cuneata*
No relationship determined by the model	Polychaeta, Anthuridae, Capitellidae, Gammaridae, Nereididae, Spionidae, Tubificidae, *Chironomus* sp., *Coelotanypus* sp., *Cyathura* sp., *Gammarus* sp., *Gemma gemma*, *Glyptotendipes* sp., *Mediomastus* sp., *Melita* sp., *Rheotanytarsus* sp., *Sabellaria vulgaris*, *Streblospio benedicti*, *Tanytarsus* sp.

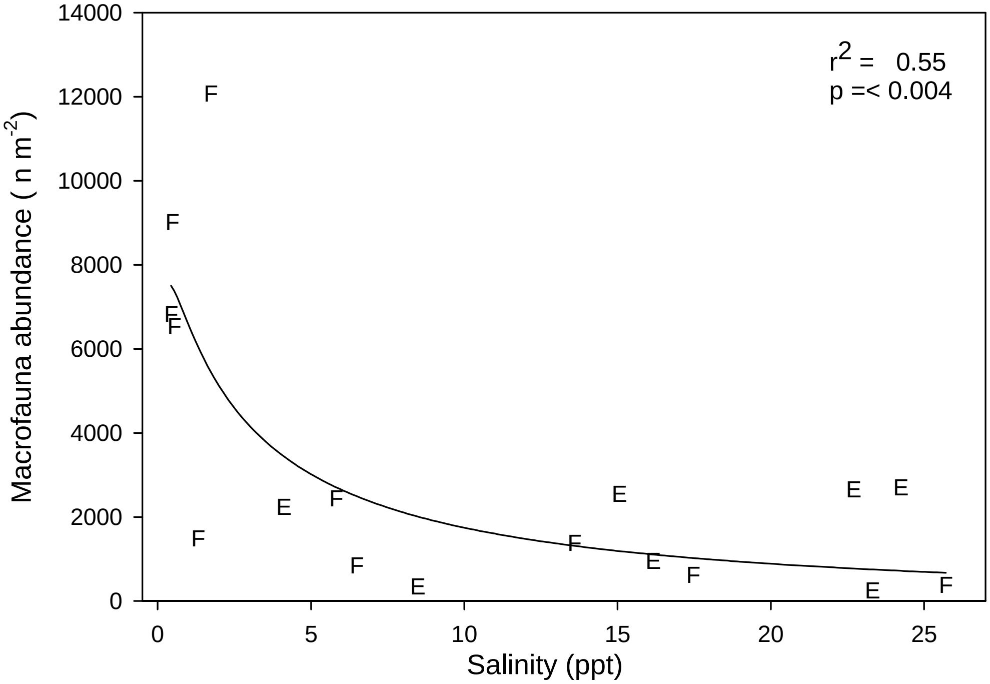

Figure 3-11. Relationship between macroinvertebrate abundance and salinity in the lower St. Johns River estuary.

The non-linear model for total abundance (Figure 3-11) yields small deviations in density (Figure 3-12) in response to changes in the areas of salinity zones (Figure 3-8). This reflects the finding that the areas of salinity zones change little in response to water withdrawals, though the positions of the zones move.

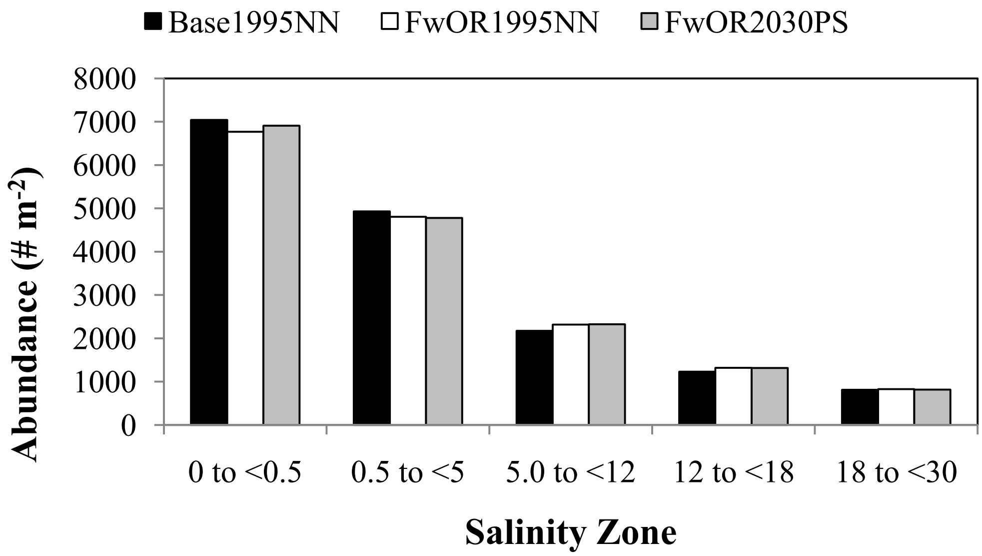

Figure 3-12. Modeled changes in the density of benthic macroinvertebrates in five salinity zones for three scenarios.

The Benthic Macroinvertebrates Workgroup used annual production rates of macroinvertebrates within four floodplain plant communities in the Kissimmee River (Koebel et al. 2005) that are also present in the floodplain of the upper St. Johns River:

- Broadleaf marsh – 6.55 g m^{-2} yr^{-1}
- *Nuphar advena* bed – 28.13 g m^{-2} yr^{-1}
- *Polygonum* marsh – 32.12 g m^{-2} yr^{-1}
- *Scripus* marsh – 203.15 g m^{-2} yr^{-1}.

The average production rate (67.49 g m^{-2} y^{-1}) for the four communities was used to estimate the deviation in floodplain production associated with water withdrawal scenarios. The deviation in production was the product of the change in m^{-2}-days of exposure and the average areal production rate in days. The change in m^{-2} days of exposure was modeled by the Biogeochemistry Workgroup (details in Chapter 7. Biogeochemistry). Under the most extreme scenario (Full1995NN, a test scenario), there was a modeled decrease in floodplain macroinvertebrate production of 516 Mg/y for the simulated period (1996-2005). This is about 14.7% of the estimated total annual production in the Lake Poinsett floodplain (3,500 Mg y^{-1}). The more realistic hindcast and forecast scenarios, such as Full1995PN and Full2030PN, had no additional days of drying, so there would be no modeled loss of benthic macroinvertebrate production.

3.2.6 Fish Attributes

The Fish Workgroup examined the potential for effects on fish species in the estuary (segments 1-3) and in freshwater areas (segments 7 and 8) (details in Chapter 12). In freshwater segments, they evaluated the potential for impingement and entrainment (I&E) of ichthyoplankton (segments 6 and 7) and for water level effects on groups of fish species.

Freshwater fish were grouped into five assemblages: (1) open water large fishes; (2) open water small forage fishes; (3) large sunfishes; (4) marsh and floodplain large fishes; and (5) littoral zone, marsh, and floodplain small fishes. For the first four groups, no material potential for effects was found even for the most extreme test scenario (Full1995NN) in the area most strongly affected (Lake Poinsett to Lake Harney; segments 7 and 8). Material modeled effects were found, however, for the fifth group.

Using a model that related densities of small fishes to flooding durations in the Everglades (DeAngelis et al. 1997), simulated declines in the density of small fish ranged from -9.3 % to -11.3 % upstream of Lake Monroe in the Full1995NN extreme test scenario (Figure 3-13). From Lake Monroe downstream, reductions in abundance ranged from -10.0 % at Pine Island to -0.2 % at Lake Woodruff. Modeled effects on small fish abundance declined markedly in scenarios with completed Upper Basin projects. The maximum modeled deviation under the Full1995PN hindcast scenario was -4.7 %. Scenarios with 2030 watershed conditions had modeled increases in small fish abundance because flood durations increased.

The potential for I&E is directly related to ichthyoplankton densities. To assess this potential, SJRWMD implemented an extensive sampling program for ichthyoplankton. This program collected 708,032 fish larvae composed primarily of 16 species. The highest densities of ichthyoplankton were collected at SR 46 (RK 310) and Lake Monroe (RK 265). Densities at these sites were 2.5 – 5 times greater than at any other site (Table 3-15; Figure 3-14). The lowest densities were found at SR 50 (RK 343) and Lake Poinsett (RK 378). However, the total catch of American shad larvae was highest at SR 50 (11,883) being more than 10 times greater than at any other site. Delineation of the peak for American shad is important because American shad is an economically important species with populations much lower than historical levels. Larval densities for all species peaked during the dry season (January – May) and larvae were nearly absent from September through November (Figure 3-14).

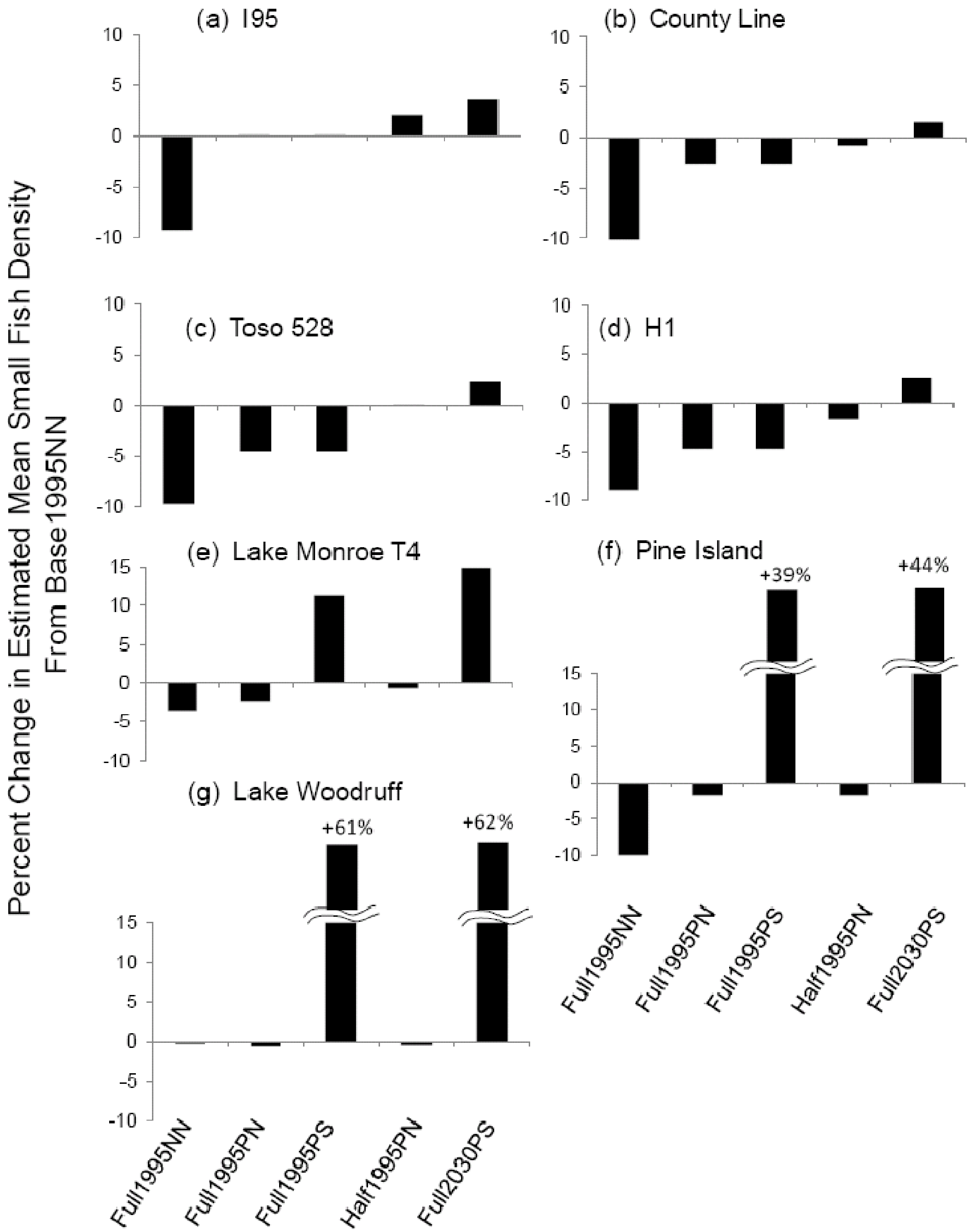

Figure 3-13. Percent deviation from the baseline condition (Base1995NN) in the modeled mean density of small fishes on the floodplain based on floodplain elevations and simulated flooding durations along seven MFL transects for five withdrawal scenarios.

Table 3-15. Number of individuals of larval fishes of various species collected at six locations in the St. Johns River from February 2008 through September 2009.

Species	Lake Poinsett	SR50	SR46	Lake Monroe	Yankee Lake	SR44
American shad	109	11,883	674	493	313	295
Blueback herring	0	52	145	70	19	11
Hickory shad	3	18	100	53	26	14
Gizzard shad	38,848	18,237	102,593	29,884	28,319	33,049
Threadfin shad	6,590	2,350	103,513	89,009	24,594	16,435
Unidentified *Dorosoma*	44	7	11,296	831	13	4,862
Channel catfish	0	3	227	1	4	1
White catfish	3	27	565	8	31	18
Tailight shiner	98	12	62	14	24	51
Tidewater silversides	5,757	1,212	3,247	2,638	1,572	643
Rough silversides	0	3	57	93	188	127
Bluegill	3,095	519	5,277	1,656	1,738	3,337
Redear sunfish	950	370	1,837	346	910	531
Black crappie	3,575	377	1,939	1,379	3,596	9,800
Clown goby	7	25	17,377	11,658	15,733	5,387
Naked goby	2	33	14,296	16,867	13,851	4,786
Unidentified gobies	0	33	9,236	931	3,067	2,913
Swamp darter	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
Avg Catch Per Transect	10,328	7,290	45,505	39,024	19,587	16,511

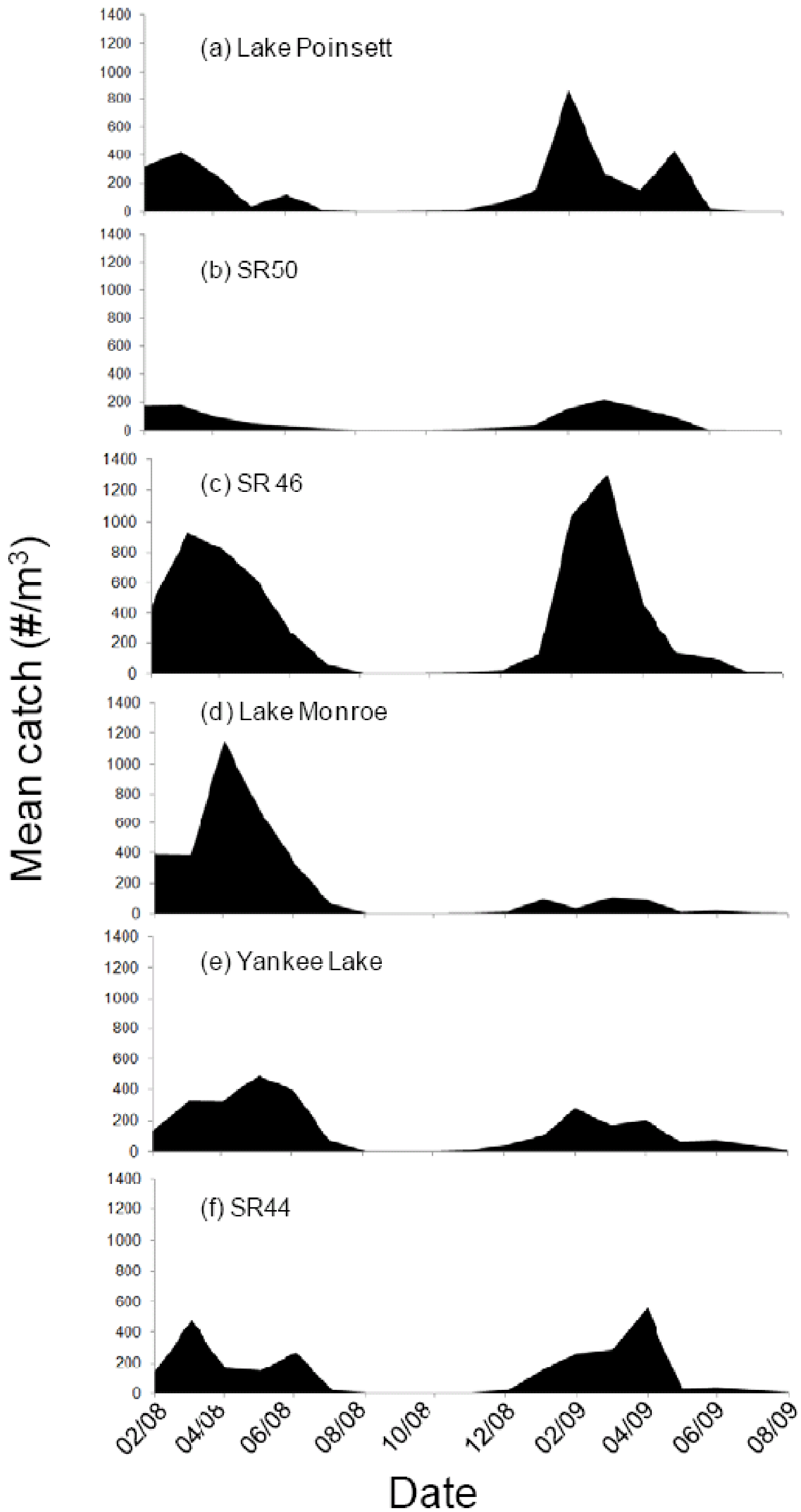

Figure 3-14. Monthly distribution of fish larvae density (number per m^3) at various locations in the St. Johns River.

In the estuary, the workgroup used regression models developed by the Florida Fish and Wildlife Commission (FWC) based on a large data set collected over a 10-year period (Jan. 1, 2001, to Dec. 31, 2010; details in Chapter 12. Fish). The sampling program consisted of 7,467 sampling events in the estuary that collected 854,233 individual fish of at least 160 species. Collected fish were grouped as pseudo-species, a designation that considered species, size, collection gear, collection zone, and recruitment period.

The regression models indicated that the abundances of 10 pseudo-species (representing 10 species) were uncorrelated with freshwater inflow (Table 3-16). Another 22 pseudo-species (21 species) had very weak correlations between abundance and freshwater inflow. Seventeen pseudo-species (16 species) had weak correlations (Table 3-17).

Table 3-16. Pseudo-species showing no or very weak correlations between abundance and freshwater inflow. Size groups are in parentheses (mm total length). Superscripts denote whether the relationship is for monthly (m) or annual (a) abundance. Lag times in days that had the best fit (highest r^2) are indicated in bold italics. + indicates a positive correlation with freshwater inflow. – indicates a negative correlation.

No abundance response to freshwater inflow

Atlantic silverside (0-75) ^{m,a}
Blackcheek tonguefish (20-110) ^{m,a}
Eastern mosquito fish (0-32) ^{m}
Ladyfish (150-525) ^{m,a}
Longnose gar (675-950) ^{m,a}
Menhaden (20-40) ^{m,a}
Seminole killifish (0-80) ^{m}
Southern kingfish (10-40) ^{m,a}
Striped mullet (0-30) ^{m,a}
White mullet (0-30) ^{m,a}

Very weak abundance response to freshwater inflow (Spearman's rho < 0.40; p < 0.05) Lag times are highest Spearman rho's .

Atlantic needlefish (100-175) ^{m} ***60*** –
Atlantic stingray (125-325) ^{m} ***90*** –
Blue crab (10-50) ^{m} ***360*** +
Blue crab (40-90) ^{m} ***30*** –
Blue crab (81-110) ^{m} ***120*** –
Bluegill (60-120) ^{m} ***360*** +
Channel catfish (101-350) ^{m} ***360*** +
Clown goby (19-28) ^{m} ***270*** –
Darter goby (0-35) ^{m} ***30*** +
Gizzard shad (201-375) ^{m} ***270*** –
Hogchoker (0-30) ^{m} ***30*** + (31-60) ^{m} ***360*** +
Hogchoker (46-75) ^{m} ***300*** +
Largemouth bass (126-250) ^{m} ***360*** +
Lined sole (10-60) ^{m} ***30*** +
Mummichog (35-60) ^{m} ***360*** –
Silver perch (10-40) ^{a} ***30*** –
Southern flounder (126-325) ^{m} **120** –
Spotted seatrout (0-30) ^{m} ***180*** –
Striped anchovy (0-45) ^{m} ***30*** –
Threadfin shad (75-125) ^{m} ***230*** –
White mullet (151-200) ^{m} ***360*** –

Table 3-17. Pseudo-species showing weak correlations between abundance and freshwater inflow. Size groups are in parentheses (mm total length). Superscripts denote whether the relationship is for monthly (m) or annual (a) abundance. Lag times in days that had the best fit (highest r2) are indicated in bold italics. + indicates a positive correlation with freshwater inflow. – indicates a negative correlation.

Weak abundance response to freshwater inflow (Spearman's rho > 0.40, but linear regression $r^2 < 0.25$; *- indicates linear regression was not significant at $p<0.05$. For pseudo-species with non-significant regression, lag times are highest Spearman's rho's.

Atlantic needlefish* (325-500) ^{a} ***30,120*** –	Silver jenny (65-90) ^{m} ***120*** –
Atlantic croaker (60-100) ^{m} ***240*** –	Silver perch (0-30*) ^{m} ***120*** – (100-130) ^{a} ***90*** +
Bay anchovy (20-35) ^{m} ***180*** –	Silver perch (131-170) ^{m} ***30*** +
Crevalle jack (50-200) ^{m} ***240*** –	Silversides (25-40) ^{m} ***120*** –
Lookdown*(30-80) ^{a} ***270,300,330*** –	Southern flounder (0-50) ^{m} ***60*** +
Pinfish (70-100) ^{m} ***360*** –	Spot (26-40) ^{m} ***30*** +
Redear sunfish (20-80) ^{m} ***90*** +	Tidewater mojarra (60-90) ^{m} ***120*** –
Red drum (36-80) ^{m} ***360*** –	White shrimp (12-27) ^{m} ***210*** +

The abundances of 61 pseudo-species were strongly correlated with freshwater inflow (Table 3-18). Ten of these pseudo-species (6 species) were freshwater fishes of which four species (bluegill, channel catfish, white catfish, and redear sunfish) had abundances positively correlated with freshwater inflow and two species (redbreast sunfish and golden shiner) had abundances negatively correlated with freshwater inflow. For estuarine or marine pseudo-species, abundances of 39 pseudo-species declined with increasing freshwater inflows while abundances of 12 pseudo-species increased with freshwater inflow. Among the more economically important species, abundances of Atlantic croaker, Atlantic weakfish, Bay anchovy, Gulf flounder, Golden shiner, and Pinfish declined with increasing freshwater inflow, while abundances of Bluegill, Channel catfish, Irish pompano, Redear sunfish, Striped mullet, and White catfish increased with increasing freshwater inflow. The correlations for several species differed by size class. Redbreast sunfish, Southern flounder, Spotted sea trout, and White mullet were noteworthy species in this group.

Table 3-18. Pseudo-species showing strong correlations between abundance and freshwater inflow. Size groups are in parentheses (mm total length). Superscripts denote whether the relationship is for monthly (m) or annual (a) abundance. Lag times in days that had the best fit (highest r2) are indicated in bold italics. + indicates a positive correlation with freshwater inflow. – indicates a negative correlation.

Strong abundance response to freshwater inflow (significant linear regression with $r^2 > 0.25$)

Atlantic croaker (131-170) ^{m} ***120*** –
Atlantic thread herring (70-110) ^{m} ***60*** –
Atlantic weakfish (41 75) ^{m} ***90*** –
Bay anchovy (36-600 ^{a} ***360*** –
Bay whiff (50-70) ^{a} ***90*** – (71-100) ^{a} ***300*** , –
Blue crab (91-170) ^{m} ***180*** – (111-180) ^{m} ***180*** –
Bluegill (20-65) ^{m} ***300*** + (20-65) ^{m} ***300*** –
Channel catfish (50-100) ^{m} ***180*** +
Channel catfish (150-275) ^{a} ***150*** +
Clown goby (29-36) ^{m} ***270*** – (37-56) ^{m} ***300*** –
Freshwater goby (0-50) ^{m} ***360*** + (30-55) ^{m} ***360*** +
Freshwater goby (30-55) ^{a} ***360*** +
Fringed flounder (50-85) ^{m} ***60*** –
Fringed flounder (61-90) ^{m} ***30*** –
Golden shiner (0-50) ^{m} ***180*** –
Gulf flounder (60-180) ^{m} ***360*** – (60-180) ^{a} ***150*** –
Gulf pipefish (0-120) ^{m} ***210*** –
Hogchoker (20-45) ^{m} ***60*** + (20-45) ^{a} ***60*** +
Irish pompano (60-110) ^{a} ***360*** +
Mummichog (0-34) ^{m} ***360*** – (0-34) ^{a} ***30*** –
Naked goby (20-35) ^{m} ***210*** – (20-35) ^{a} ***240*** –
Pigfish (80-130) ^{m} ***360*** –
Pinfish (36-70) ^{m} ***300*** – (101-130) ^{m} ***330*** –
Pinfish (101-130) ^{a} ***300*** – (131-160) ^{m} ***360*** –
Pinfish (131-160) ^{a} ***30*** , –
Rainwater killifish (0 32) ^{m} ***270*** –
Redbreast sunfish (20-110) ^{a} ***360*** +
Redbreast sunfish (131-190) ^{a} ***30*** –
Redear sunfish (0-125) ^{m} ***360*** +
Silver perch (31-55) ^{m} ***90*** – (80-100) ^{m} ***90*** –
Silversides (41-55) ^{m} ***60*** –
Southern flounder (0-50) ^{m} ***30*** +
Southern flounder (51-100) ^{m} ***210*** +
Southern flounder (126-325) ^{a} ***150*** –
Southern puffer (70-170) ^{a} ***210*** –
Spot (41-60) ^{m} ***60*** + (60-90) ^{m} ***60*** –
Spot (60-90) ^{a} ***30*** – (91-120) ^{m} ***60*** –
Spotted seatrout (31-50) ^{m} ***150*** –
Spotted seatrout(51-110) ^{m} ***300*** –
Spotted seatrout (210-325) ^{a} ***60*** +
Striped burrfish (40-110) ^{a} ***270*** –
Striped mullet (31-45) ^{a} ***210*** +
Tidewater mojarra (91-110) ^{m} ***360*** –
White catfish (25-100) ^{m} ***300*** +
White catfish (101-200) ^{m} ***360*** +
White mullet (31-80) ^{m} ***180*** + (100-130) ^{m} ***150*** –

Important deviations in abundances of fishes were associated with an extreme test scenario (Full1995NN; Table 3-19). For large, open water/riverine species, deviations ranged from -13 – -43 % (White catfish and Channel catfish, respectively) and for large sunfishes from +6 - -37% (Redbreast sunfish and Redear sunfish, respectively).

For estuarine marsh and benthic species, modeled deviations varied from -43 – 73 %, for freshwater goby (30-55 mm) and tidewater mojarra (60-180 mm), respectively (Table 3-19). Sciaenids had modeled deviations of -19 – 65 % for spot (41-60 mm) and silver perch (80-100 mm), respectively. All other marine fishes increased in abundance above the baseline condition with deviations ranging from 7 – 55 % for pinfish (36-70 mm) and striped burrfish (40-110 mm), respectively.

The direction of deviations in the Full1995NN extreme test scenario indicates the direction of reduction effects. A reversal of the direction of the deviation indicates an augmentation effect. Reduction effects were largely eliminated for the forecast scenarios, Half2030PS and

Full2030PS. For many species, reduction effects were observed, however, for the forecast scenario FwOR2030PS.

Table 3-19. Predicted deviations (%) from the base condition (Base1995NN) in median monthly and/or annual relative abundance for species with abundances strongly correlated with freshwater inflow ($P<0.05$; $r^2>0.25$). Gear types used for collection are indicated by superscripts (1, 21.3-m seine; 2, 183-m seine; 3, 6.1-m trawl. Zones are those for which the modeled deviations are appropriate.

Common Name	TL (mm)	Zones	Period Analyzed	Temporal Response	r^2	Full1995NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030PS
				Open Water/Riverine Large Fishes							
Channel catfish [3]	50-100	4-8	Sep-Jan	Monthly	0.67	-23.20%	-7.77%	-17.40%	29.94%	17.20%	-0.90%
Channel catfish [2]	150-275	4	May-Sep	Annual	0.61	-42.68%	-28.45%	-52.06%	7.03%	-21.92%	-63.82%
White catfish [3]	25-100	3-8	Sep-Mar	Monthly	0.69	-22.60%	-9.79%	-20.90%	24.46%	10.50%	-8.30%
White catfish [3]	101-200	3-8	Jan-Dec	Monthly	0.35	-12.80%	-4.91%	-11.30%	10.70%	4.60%	-5.70%
				Open Water Small Forage Fishes							
Golden shiner [1]	0-50	6-8	May-Jul	Monthly	0.27	28.40%	10.90%	25.10%	-5.16%	4.90%	26.50%
				Large Sunfishes							
Bluegill [1]	20-65	3-4	Aug-Nov	Monthly	0.52	-32.90%	-13.95%	-30.20%	33.66%	13.60%	-16.20%
Bluegill [1]	20-65	3-4	Aug-Nov	Annual	0.61	-11.29%	-4.57%	-9.91%	6.03%	0.33%	-6.74%
Redbreast sunfish [1]	20-110	3-4	Jan-Dec	Annual	0.74	-9.74%	-4.71%	-8.95%	8.34%	3.83%	-3.14%
Redbreast sunfish [2]	131-190	4	Sep-Apr	Annual	0.77	6.01%	1.30%	4.59%	-11.27%	-6.42%	-2.43%
Redear sunfish [2]	0-125	4	Nov-Jun	Monthly	0.38	-36.90%	-17.37%	-34.10%	39.32%	14.10%	-16.70%
				Open Water Small Estuarine Fishes							
Atlantic thread herring [2]	70-110	1-2	Aug-Oct	Monthly	0.33	36.90%	4.75%	25.00%	-31.71%	-16.70%	-0.20%
Bay anchovy [3]	36-60	1-4	May-Jan	Annual	0.64	7.08%	2.23%	5.34%	-4.38%	-1.28%	3.78%
Silversides [1]	41-55	1-8	Jan-Dec	Monthly	0.26	4.08%	1.36%	3.27%	-2.60%	-0.55%	2.58%

Table 3-19 (continued). Predicted deviations (%) from the baseline condition (Base1995NN) in median monthly and/or annual relative abundance for species with abundances strongly correlated with freshwater inflow ($P<0.05$; $r^2>0.25$). Gear types used for collection are indicated by superscripts (1, 21.3-m seine; 2, 183-m seine; 3, 6.1-m trawl. Zones are those for which the modeled deviations are appropriate.

Common Name	TL (mm)	Zones	Period Analyzed	Temporal Response	r^2	Full1995NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030PS
Estuarine Marsh Fishes											
Mummichog [1]	0-34	1-2	Dec-Jan	Monthly	0.45	29.70%	11.47%	25.30%	-19.90%	-9.10%	8.60%
Mummichog [1]	0-34	1-2	Dec-Jan	Annual	0.76	6.41%	4.24%	8.53%	0.41%	2.60%	7.20%
Rainwater killifish [1]	0-32	4-8	Mar-Jun	Monthly	0.71	19.00%	6.86%	15.20%	-11.74%	-5.40%	6.40%
Striped mullet [1]	31-45	1-4	Mar-Jun	Annual	0.52	-9.94%	-3.76%	-8.28%	8.93%	3.90%	-3.36%
White mullet [1]	31-80	1-2	Jun-Jul	Monthly	0.32	-17.50%	-6.31%	-15.90%	7.77%	-3.00%	-19.20%
White mullet [2]	100-130	2-4	Oct-Jan	Monthly	0.28	10.80%	3.36%	7.60%	-11.71%	-7.60%	-1.40%
Estuarine Benthic Fishes											
Bay whiff [2]	50-70	1-4	Jun-Jul	Annual	0.50	40.67%	5.56%	20.28%	-12.53%	-0.07%	21.64%
Bay whiff [2]	71-100	1-4	Jun-Sep	Annual	0.56	8.12%	3.23%	6.52%	-4.82%	-1.93%	3.99%
Clown goby [3]	29-36	4-8	Sep-Apr	Monthly	0.32	10.20%	3.35%	8.40%	-6.89%	-4.20%	2.30%
Clown goby [3]	37-56	4-8	Oct-Apr	Monthly	0.31	17.70%	5.37%	13.50%	-13.16%	-5.50%	4.70%
Naked goby [1]	20-35	1-8	Dec-Apr	Monthly	0.47	13.80%	4.66%	10.30%	-10.84%	-5.60%	2.40%
Naked goby [1]	20-35	1-4	Dec-Apr	Annual	0.583	8.29%	2.27%	5.74%	-7.25%	-3.84%	1.41%
Freshwater goby [1]	0-50	1-4	Nov-Mar	Monthly	0.29	-30.90%	-13.96%	-20.30%	32.63%	11.90%	-13.00%
Freshwater goby [3]	30-55	3-4	Dec-Mar	Monthly	0.42	-35.60%	-15.45%	-32.00%	37.82%	14.50%	-13.90%
Freshwater goby [3]	30-55	3-4	Dec-Mar	Annual	0.62	-42.59%	-25.04%	-42.13%	56.78%	19.76%	-19.26%
Fringed flounder [2]	50-85	1-3	Sep-Dec	Monthly	0.26	18.00%	5.33%	14.30%	-19.85%	-15.40%	-6.10%
Fringed flounder [3]	61-90	1-3	Aug-Dec	Monthly	0.28	13.20%	2.60%	8.90%	-16.73%	-16.10%	-11.50%
Gulf flounder [3]	60-100	1-3	Mar-Sep	Monthly	0.27	40.50%	15.00%	34.00%	-19.69%	-0.90%	12.40%
Gulf flounder [2]	60-180	1-3	Mar-Sep	Annual	0.47	27.32%	8.65%	44.27%	-11.32%	-3.49%	14.07%
Hogchoker [3]	20-45	3-6	Sep-Mar	Monthly	0.37	-4.40%	-2.83%	-4.50%	3.58%	2.20%	-0.60%
Hogchoker [3]	20-45	1-4	Sep-Mar	Annual	0.60	-8.00%	-2.07%	-5.97%	13.28%	8.35%	3.19%

Table 3-19 (continued). Predicted deviations (%) from the baseline condition (Base1995NN) in median monthly and/or annual relative abundance for species with abundances strongly correlated with freshwater inflow (P<0.05; r2>0.25). Gear types used for collection are indicated by superscripts (1, 21.3-m seine; 2, 183-m seine; 3, 6.1-m trawl. Zones are those for which the modeled deviations are appropriate.

Common Name	TL (mm)	Zones	Period Analyzed	Temporal Response	r^2	Full1995NN	Half1995PN	Full1995PN	Half2030PS	Full2030PS	FwOR2030PS
Irish pompano [2]	60-110	1-4	Oct-Dec	Annual	0.60	-10.47%	-3.96%	-9.12%	9.04%	3.21%	-4.66%
Southern flounder [3]	0-50	1-5	Feb-May	Monthly	0.35	-18.90%	-5.94%	-15.70%	2.30%	-6.10%	-22.70%
Southern flounder [3]	51-100	1-8	Apr-Sep	Monthly	0.27	-19.70%	-7.87%	-16.50%	10.10%	0.30%	-12.00%
Southern flounder [2]	126-325	1-3	Feb-Nov	Annual	0.55	3.64%	1.61%	3.06%	-2.68%	-1.16%	0.98%
Tidewater mojarra [2]	91-110	4	May-Jun	Monthly	0.46	72.80%	18.65%	48.70%	-29.30%	-5.60%	28.30%
						Sciaenid Fishes					
Atlantic croaker [2]	131-170	1	Jul-Sep	Monthly	0.33	17.00%	3.36%	13.30%	-10.80%	-3.30%	6.10%
Atlantic weakfish [3]	41-75	1-8	Jun-Nov	Monthly	0.42	15.70%	3.34%	10.30%	-15.17%	-6.90%	1.80%
Silver perch [2]	80-100	4	Sep-Oct	Monthly	0.26	64.80%	9.13%	40.00%	-42.44%	-27.40%	-5.30%
Silver perch [1]	31-55	1-6	May-Jul	Monthly	0.42	18.60%	4.49%	13.40%	-7.08%	1.20%	17.10%
Spot [3]	41-60	1-4	Apr-Jun	Monthly	0.32	-19.00%	-5.17%	-13.60%	4.52%	-2.70%	-21.30%
Spot [2]	60-90	1-4	Apr-Jun	Monthly	0.38	16.20%	4.34%	12.20%	-4.25%	2.10%	16.70%
Spot [2]	60-90	1-4	Apr-Jun	Annual	0.49	8.42%	2.26%	6.12%	-1.53%	1.23%	7.29%
Spot [2]	91-120	1-4	May-Sep	Monthly	0.32	13.30%	2.86%	9.80%	-10.46%	-4.40%	5.40%
Spotted seatrout [1]	31-50	1-6	Jun-Oct	Monthly	0.65	56.80%	10.71%	42.40%	-34.25%	-14.90%	23.10%
Spotted seatrout [1]	51-110	1-6	Jul-Dec	Monthly	0.39	47.90%	16.48%	36.90%	-27.57%	-12.50%	17.30%
Spotted seatrout [2]	201-325	1-2	Nov-Mar	Annual	0.58	-3.70%	-1.81%	-2.99%	4.72%	4.15%	1.70%
						Marine Fishes					
Gulf pipefish [1]	0-120	4-8	May-Oct	Monthly	0.65	30.50%	8.27%	23.90%	-14.70%	-3.10%	15.40%
Pigfish [2]	80-130	2-4	Jul-Sep	Monthly	0.30	41.30%	7.83%	25.80%	-32.54%	-22.60%	-0.10%
Pinfish [1]	36-70	1-5	Apr-Jul	Monthly	0.29	7.00%	2.73%	6.60%	-4.27%	-1.80%	2.30%
Pinfish [2]	101-130	1-4	Jul-Oct	Monthly	0.33	13.40%	4.95%	12.60%	-8.49%	-2.70%	6.50%
Pinfish [2]	101-130	1-4	Jul-Oct	Annual	0.57	8.03%	3.83%	8.48%	-3.87%	-1.41%	3.72%
Pinfish [2]	131-160	1-4	Aug-Oct	Monthly	0.39	10.40%	4.89%	9.70%	-7.63%	-2.80%	4.40%
Pinfish [2]	131-160	1-4	Aug-Oct	Annual	0.63	11.73%	1.11%	7.94%	-14.29%	-9.75%	-2.42%
Southern puffer [2]	70-170	1-3	Aug-Nov	Annual	0.74	16.44%	4.16%	12.30%	-9.49%	-2.99%	11.57%
Striped burrfish [2]	40-110	1-2	Jun-Oct	Annual	0.67	54.98%	32.86%	66.66%	-13.34%	2.36%	41.25%

3.2.7 FLOODPLAIN WILDLIFE

Quantitative HE models could not be developed for wildlife species. Consequently, the deviations in habitats and in food sources become qualitative forcings for effects on floodplain wildlife (Figure 3-15). Details on the qualitative relationships between floodplain wildlife and H&H drivers can be found in Chapter 13. Floodplain Wildlife.

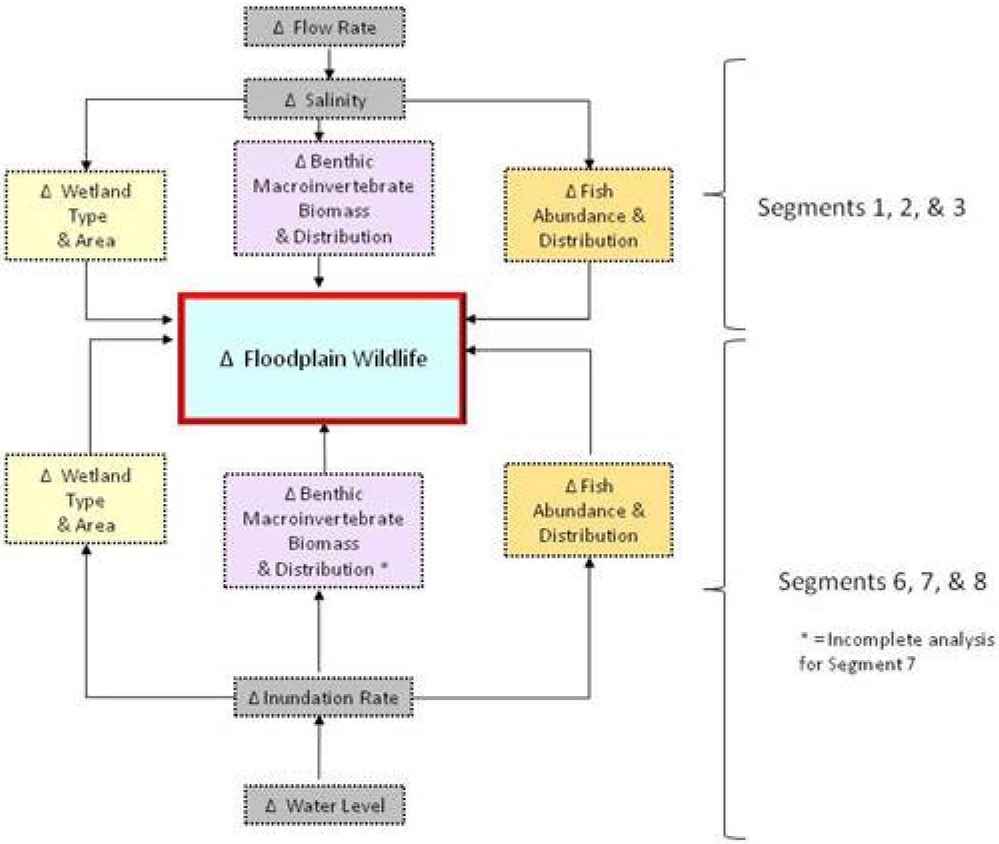

Figure 3-15. Conceptual model for potential effects on floodplain wildlife.

3.3 EVALUATION OF THE SIGNIFICANCE OF EFFECTS

3.3.1 EFFECTS BY H&H DRIVER

As described in section 2.9, each ecological workgroup evaluated the significance of potential effects on key ecological attributes based on considerations of the strength, persistence, and diversity of the modeled effects. We used five levels of effects ranging from extreme to negligible. These levels provide a means for holistic consideration of the various scenarios and their ecological effects along the river's length.

There is the potential for major effects in only two segments (6 and 7) that is attributable to the potential for entrainment and impingement (E&I) of planktonic life stages of fish. We assume that the potential for E&I can be substantially reduced with appropriate design and operation of water intake structures. All other effects ranged from negligible to moderate.

If we examine the overall pattern of the levels of effects attributable to each of the main H&H drivers for the most extreme test scenario (FwOR1995PN), we see that the levels of effects were lowest in segments 4 and 5 (Table 3-20). Modeled effects in other segments were attributable primarily to water level effects on wetland vegetation (segments 6–8) and to flow effects on fish communities (segments 1–3). In segment 2, upstream movement of salinity isopleths has the potential to cause some loss of freshwater plant communities. These species would be replaced by more salt-tolerant species. In segments 2-4, there is a potential for minor intensification of phytoplankton blooms, due to increased residence time (as indicated by modeled water age), and for minor effects on the proportionality of wetland vegetation communities along the hydrologic gradient, due to small decreases in inundation of the floodplain. At the modeled levels of withdrawals, effects from deviations in salinity and water age seem to be minor concerns compared to effects due to deviations in discharge and water levels.

The spatial pattern of deviations in H&H drivers (Figure 3-5, Figure 3-6, and Figure 3-7) was reflected in the pattern of modeled ecological effects. Appreciable salinity effects were limited to segment 2, but fish population responses to discharge deviations extended the potential for effects on the estuary to segments 1 and 3. As expected ecological effects from water level deviations were limited to segments 6–8. Ecological effects stemming from deviations in water age were no more than minor in any segment of the river for hindcast or forecast scenarios because the deviations in water age were small. As previously described, the potential for effects from entrainment and impingement was highest in segments 6 and 7.

3.3.2 Effects by Scenario

An examination of the pattern of modeled effects for hindcast and forecast scenarios clarifies the relative sensitivities of the ecological attributes to withdrawals and indicates the relative risks associated with each of these withdrawal scenarios (Table 3-21 to Table 3-25). Excluding the potential for entrainment and impingement of larval fish, which can be avoided (see discussion below), this analysis shows that no hindcast for forecast scenario had extreme or major effects. Moderate effects were limited to effects on attributes of wetland vegetation, benthic macroinvertebrates, fish, and floodplain wildlife. In all scenarios, modeled effects on attributes of biogeochemistry, plankton, and SAV were negligible.

Using the modeled deviations in mean discharge during the dry season (Table 3-4), the hindcast and forecast scenarios can be ranked from largest to smallest deviations: Full1995PN, FwOR2030PS, Half1995PN, Full2030PS, and Half2030PS. Moderate effects were limited to the first three of these scenarios. Only the extreme hindcast scenario (Full1995PN; Table 3-21), had moderate effects for segments in both the lower and upper reaches of the river. For the next two scenarios (FwOR2030PS and Half1995PN; Table 3-22 and Table 3-23), moderate effects were restricted to the estuarine segments. Effects were limited to no more than minor for the remaining forecast scenarios (Full2030PS and Half2030PS; Table 3-24 and Table 3-25).

To elucidate the relative risks associated with the hindcast and forecast scenarios, for each scenario we assessed the overall effects, across the ecological attributes, for each river segment (Table 3-26). We judged the overall level of effect for the segment to be equal to the highest level of effect for any of the ecological attributes assessed. We judged the overall uncertainty to be the lowest uncertainty associated with an attribute that had this highest level of effect.In scenarios where effects were judged negligible, uncertainty was that associated with the H&H models, which was always low or very low. We judged that this process identifies the attribute that provides the strongest basis for making decisions regarding water withdrawals.

This analysis shows that only the Full1995PN hindcast scenario had moderate modeled effects in segments upstream of the estuary. Two of the forecast scenarios (Full2030PS and Half2030PS), had no more than minor modeled effects in all segments. The overall uncertainties associated with these two scenarios spanned the range from very low to very high. As discussed below, in cases where there is high or very high uncertainty, it would be prudent to use a margin of safety in approving water withdrawals.

Excluding the potential for E&I, no hindcast or forecast scenario had more than moderate potential effects for any river segment. Indeed, with the exception of the potential for E&I, there were no major or extreme modeled effects for any of the scenarios analyzed, even including the most extreme test scenarios.

Table 3-20. Summary of overall levels of ecological effects attributable to each H&H driver by river segment and the determinative ecological attribute for non-negligible effects (Biogeochemistry, Plankton, Littoral Zone, Wetland Vegetation, Macroinvertebrate Benthos, Fish, Floodplain Wildlife). Scenario FwOR1995PN/PS (Segments 1-3), extreme test scenarios, or Full1995PN (segments 4-8; and segment 2 for Wetland Vegetation) – an extreme, hindcast scenario. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/driver was not analyzed.

River Segment	Entrainment & Impingement	Flow Rate	Salinity	Water Age	Water Level	Overall (excluding E&I)
1	NA	**F		NA		F
2	NA	**F	*V	**P		F, V
3	NA	**F	***M	**P		F
4	NA	***F	NA	*P		F
5	*F	NA	NA	NA	***V	F, V
6	*F	NA	NA	*P	***F	F
7	*F	NA	NA	NA	***F ***V	F, V
8	NA	NA	NA	**P	***F **V	F, V

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-21. Summary of levels of effects for ecological attributes for each river segment for hindcast scenario Full1995PN. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/attribute was not analyzed. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Biogeochemistry	Plankton	Submerged Aquatic Vegetation	Wetland Vegetation	Benthos	Fish	Wildlife
1	*	NA		*	**	**	*****
2	*	**	**	*	***	**	*****
3	*	**	**	*	***	**	*****
4	*	*	**	*	*****	***	NA
5	*	NA	***	***	*****	***	NA
6	*	*	***	***	****	***	****
7	**	NA	***	***	****	***	****
8	*	**	***	**	****	***	****

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-22. Summary of levels of effects for ecological attributes for each river segment for forecast scenario FwOr2030PS. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/attribute was not analyzed. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Biogeochemistry	Plankton	Submerged Aquatic Vegetation	Wetland Vegetation	Benthos	Fish	Wildlife
1	*	NA		*	**	**	*****
2	*	**	**	***	***	**	*****
3	*	**	**	*	***	**	*****
4	*	*	**	*	*****	*	NA
5	*	NA	***	***	*****	****	NA
6	*	*	***	***	****	****	****
7	**	NA	***	***	****	****	****
8	*	**	***	**	****	**	****

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-23. Summary of levels of effects for ecological attributes for each river segment for hindcast scenario Half1995PN. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/attribute was not analyzed. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Biogeochemistry	Plankton	Submerged Aquatic Vegetation	Wetland Vegetation	Benthos	Fish	Wildlife
1	*	NA		*	**	**	*****
2	*	**	**	*	***	**	*****
3	*	**	**	*	***	**	*****
4	*	*	**	*	*****	***	NA
5	*	NA	***	***	*****	***	NA
6	*	*	***	***	****	***	****
7	**	NA	***	***	****	***	****
8	*	**	***	**	****	***	****

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-24. Summary of levels of effects for ecological attributes for each river segment for forecast scenario Full2030PS. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/attribute was not analyzed. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Biogeochemistry	Plankton	Submerged Aquatic Vegetation	Wetland vegetation	Benthos	Fish	Wildlife
1	*	NA		*	**	****	*****
2	*	**	**	*	***	****	*****
3	*	**	**	*	***	****	*****
4	*	*	**	*	*****	*	NA
5	*	NA	***	***	*****	**	NA
6	*	*	***	***	****	**	****
7	**	NA	***	***	****	**	****
8	*	**	***	**	****	**	****

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-25. Summary of levels of effects for ecological attributes for each river segment for forecast scenario Half2030PS. Vertical hatching indicates an abbreviated analysis. NA indicates the segment/attribute was not analyzed. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Biogeochemistry	Plankton	Submerged Aquatic Vegetation	Wetland Vegetation	Benthos	Fish	Wildlife
1	*	NA	NA	*	**	**	*****
2	*	**	**	*	***	**	*****
3	*	**	**	*	***	**	*****
4	*	*	**	*	*****	*	NA
5	*	NA	***	***	*****	**	NA
6	*	*	***	***	****	**	****
7	**	NA	***	***	****	**	****
8	*	**	***	**	****	**	****

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

Table 3-26. Overall level of effects and uncertainty for each river segment for hindcast and forecast scenarios. The specific ecological attribute with the highest effects level and with the lowest scientific uncertainty or in the case that all effects were judged negligible, the uncertainty was that associated with the H&H models, which was always low or very low (Biogeochemistry, Plankton, Submerged Aquatic Vegetation, Wetland Vegetation, Macroinvertebrate Benthos, Fish, Floodplain Wildlife). Vertical hatching indicates an abbreviated analysis. Potential direct and indirect effects due to entrainment and impingement were excluded from this analysis.

Segment	Full1995PN	Half1995PN	FwOR2030PS	Full2030PS	Half2030PS
1	** F	** F	** F	**** F	
2	* V	** F	* **V	* V	
3	** F	** F	** F	*** M	
4	*** F	**** F			
5	*** F	*** F			
6	*** F	*** F			
7	*** F	*** F			
8	*** F	** V	** V	** V	

	Negligible effect	*	Very low uncertainty
	Minor effect	**	Low uncertainty
	Moderate effect	***	Medium uncertainty
	Major effect	****	High uncertainty
	Extreme effect	*****	Very high uncertainty

4 GENERAL DISCUSSION AND CONCLUSIONS

4.1 VALUATION OF EFFECTS

We based the levels of effects on the strength, persistence, and diversity of deviations from the baseline condition. Whether an effect is detrimental, beneficial, or neutral, therefore, is not inherent in its level. In some cases, such valuation of an effect will be quite difficult unless one adheres to the position that any deviation from the baseline condition is deleterious. For example, effects on the estuarine fish community stem from modeled changes in catch per unit effort for a wide array of pseudospecies. The modeled effect for some pseudospecies was an increase in abundance; for others, a decrease in abundance; and for still others, there was no statistically significant change in abundance. Thus, there are potential “increasers,” “decreasers,” and “non-responders.” Whether the net effect is judged as beneficial, deleterious, or neutral depends upon societal considerations, such as aesthetics, economics, regulations, and policies. We wish to point out, however, that it would not be an unreasonable initial position to consider deviations from the baseline to be detrimental (with some exceptions, such as a decrease in phytoplankton blooms or an increase in SAV). The precautionary principle would argue for such a stance given the uncertainty of the sustainability of an altered ecological condition and the potential for unforeseen, undesirable, and unintended consequences.

4.2 Scaling Potential Effects on H&H Drivers

All aquatic life in the St. Johns River system must cope with a dynamic (i.e., highly variable) abiotic environment. The potential forcings for H&H drivers must be considered in this context. To gain this perspective, we scaled the modeled deviations in H&H drivers to their levels of natural background variation. This scaling indicates that forcings for the four main H&H drivers were small in comparison to natural background variability. It seems a reasonable proposition, that changes in hydrologic drivers that are proportionately small in comparison to natural background variation will have proportionately small effects on ecological attributes, for the reason that these attributes already must cope with much larger natural variability in the relevant H&H driver.

We note that each segment of the river has historically experienced high variation in at least one of the four primary drivers: salinity, water level, discharge, and residence time. The consequence of this variability in hydrologic drivers is that ecological communities have adapted to relatively large and rapid changes in those drivers. The degree and spatial extent of variability in each driver differs along the length of the river. For example, variation in salinity has been greatest in the lower reach while variation in water level has been greatest in the upper reach.

We compared the modeled median deviations in these drivers from the baseline condition (Base1995NN) for two extreme test scenarios (Full1995NN, upstream of the Ocklawaha River; FwOR1995NN, downstream of the Ocklawaha River) to the modeled variation in each driver for the baseline condition (Base1995NN). This approach is analogous to using the coefficient of variation.

Salinity is highly variable in the lower reaches of the river (Figure 4-1). Tidal changes in salinity are frequently exacerbated by reverse flow events that drive sea water upstream. The modeled 10-year median deviation for the extreme scenario (FwOR1995NN) was 0.6 at approximately RK 45, the location where the modeled deviation in salinity was greatest. However, the median daily salinity range in the base scenario (Base1995NN) is 0.5. Hence, the 10-y median deviation is equivalent to the daily range already present in this reach. The maximum modeled daily salinity range was 11.7. Therefore, the median salinity difference is only 5% of the maximum daily range. Furthermore, the maximum instantaneous modeled salinity deviation (1.4) is only

12% of the maximum daily range under current conditions. At a weekly scale, the median salinity range was 3.1 and the maximum range was 16.9. Therefore, the 10-y median deviation is 19% and 4% of the median and maximum weekly ranges, respectively.

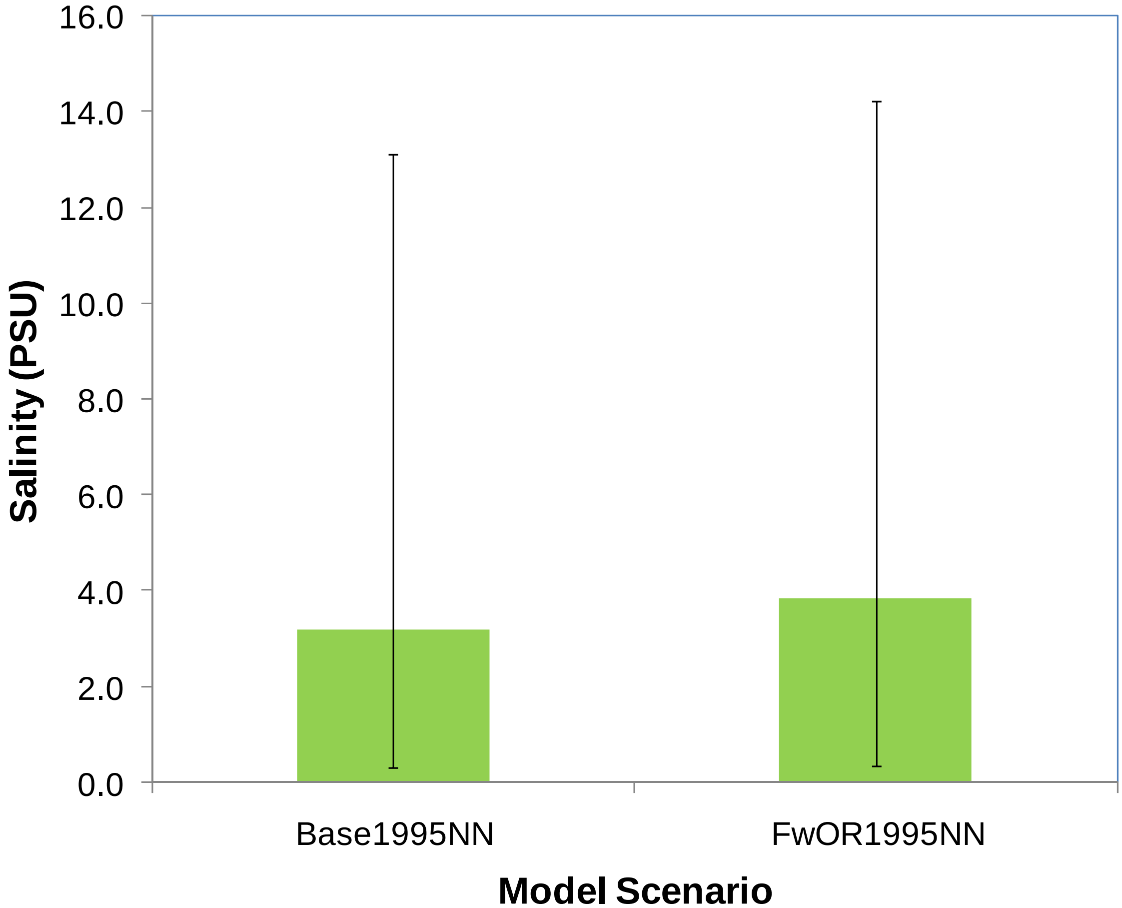

Figure 4-1. Modeled 10-y medians and ranges in salinity for the site that showed the largest deviations in salinity (river km 45) for the baseline condition (Base1995NN) and most extreme test scenario (FwOR1995NN). Error bars indicate the 10^{th} and 90^{th} percentiles of modeled average daily salinity. The modeled 10-y median deviation (difference between the two medians) was 0.6 psu.

Variable and stochastic salinity spikes characterize the estuarine reach. The communities in that river reach are regularly exposed to broad and rapid salinity changes (Chapter 11. Benthic Macroinvertebrates). In comparison, salinity changes suggested by model scenarios are relatively small. It seems unlikely that communities that experience daily to weekly salinity extremes of 12-17 psu would be greatly affected by the small salinity differences shown by model results. Water withdrawals are unlikely to change low salinity conditions but do appear to affect the frequency of higher salinity events (Figure 4-1). Although the overall differences are small compared to normal variation, it is still possible that organisms with strict salinity thresholds, particularly sessile organisms, might be affected by increases in salinity to some degree.

The effects of water withdrawals on water levels (stage) also were relatively small compared to normal background variation (Figure 4-2). The maximum deviation from the baseline condition (Base1995NN) for extreme test scenarios (Full1995NN, upstream of the Ocklawaha River; FwOR1995NN, downstream of the Ocklawaha River) occurred in June. The maximum median June stage difference of 0.12 m occurred at RK 354. The modeled withdrawal effects on water levels were evenly distributed from RK 348 to RK 362 (in segment 7) but rapidly diminished downstream. For the extreme test scenario (Full1995NN), the decreases in water levels in this area of segment 7 represent 21-23% of the maximum June range in the baseline condition. Downstream of this reach, the median difference in stage becomes a smaller proportion of the normal stage range. As with salinity, empirical data and modeled output indicates that the river has experienced variations in water level larger than the modeled deviations associated with water withdrawals, even under the most extreme test scenarios examined. Nevertheless, these small stage changes can have disproportionately larger biological or biogeochemical effects on the floodplain. Small changes in flood durations can affect processes like soil oxidation, nutrient release, and primary and secondary productivity over large areas of the floodplain. Were it not for the refractory character of floodplain organic soils, additional soil oxidation over large areas could have the potential for substantial increases in loadings of nutrients.

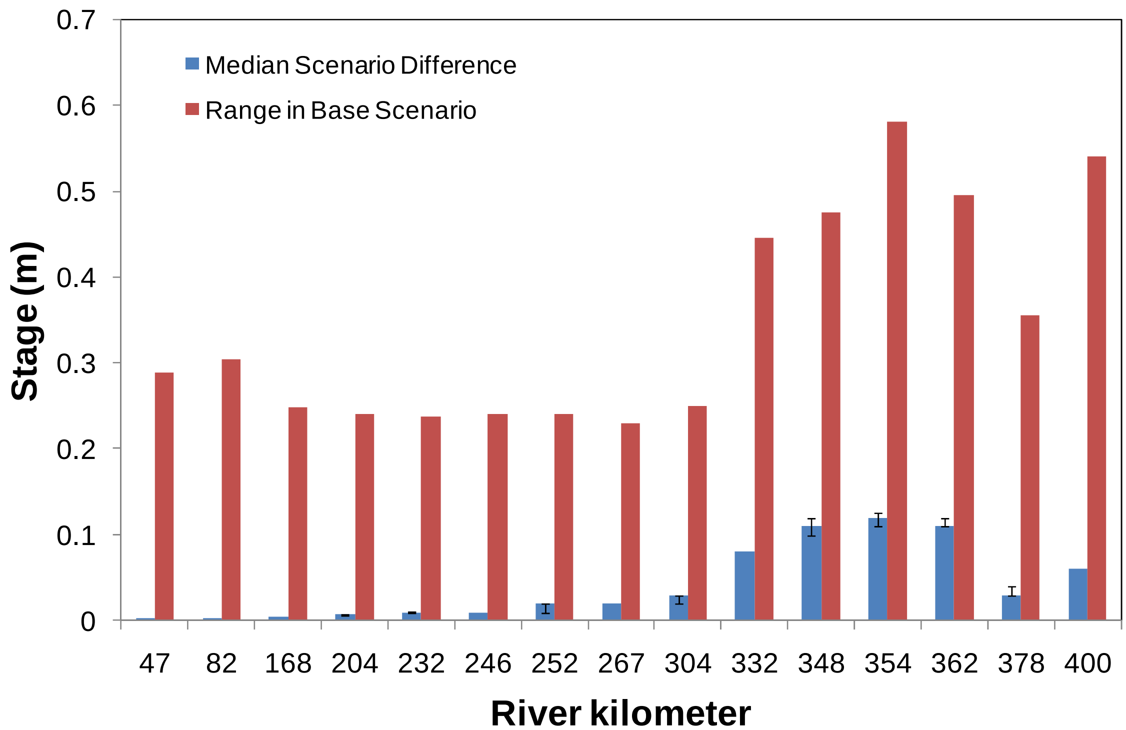

Figure 4-2. Longitudinal variation in maximum modeled range in water level (stage) during June, the month with the greatest modeled deviations from the base condition, and in modeled median deviation from the base condition during June. Deviations are between the base condition (Base1995NN) and the most extreme scenario (FwOR1995NN, downstream of the Ocklawaha River; Full1995NN, upstream of the Ocklawaha River). Error bars indicate the 95% confidence intervals of the median deviations.

As with water level, June showed the largest deviations in discharge. Discharge is also a highly variable hydrologic driver. The maximum June discharge range was 50-60 $m^3 s^{-1}$ (1141 – 1369 mgd) between river kilometers 282-400 (Figure 4-3). Downstream from river kilometer 282 the range in discharge rapidly increased to 1380 $m^3 s^{-1}$ (31498 mgd) at river kilometer 43. The median deviation from the baseline condition (Base1995NN) for the most extreme test scenario (FwOR1995NN, downstream of the Ocklawaha River; Full1995NN, upstream of the Ocklawaha River) varies from ca. 2 $m^3 s^{-1}$ (45.6 mgd) in the upper reaches of the river (RK 282-400) to ca. 10 $m^3 s^{-1}$ (228 mgd) near the mouth (RK 43; Figure 4-3). Upstream from river kilometer 145, 2 $m^3 s^{-1}$ (45.6 mgd) represents a relatively fixed 2-4% of the maximum June discharge variation. In the lower 100 km of the river, the median deviation represents less than 1% of the maximum June discharge variation. The variation in June discharge for the baseline condition is much greater than the deviations from the baseline condition for extreme test scenarios, suggesting that the ecosystem likely has the capacity to adapt to withdrawal-related discharge reductions. Moreover, these small discharge differences are unlikely to cause discernable changes in flow velocity, vertical mixing, or shear velocity.

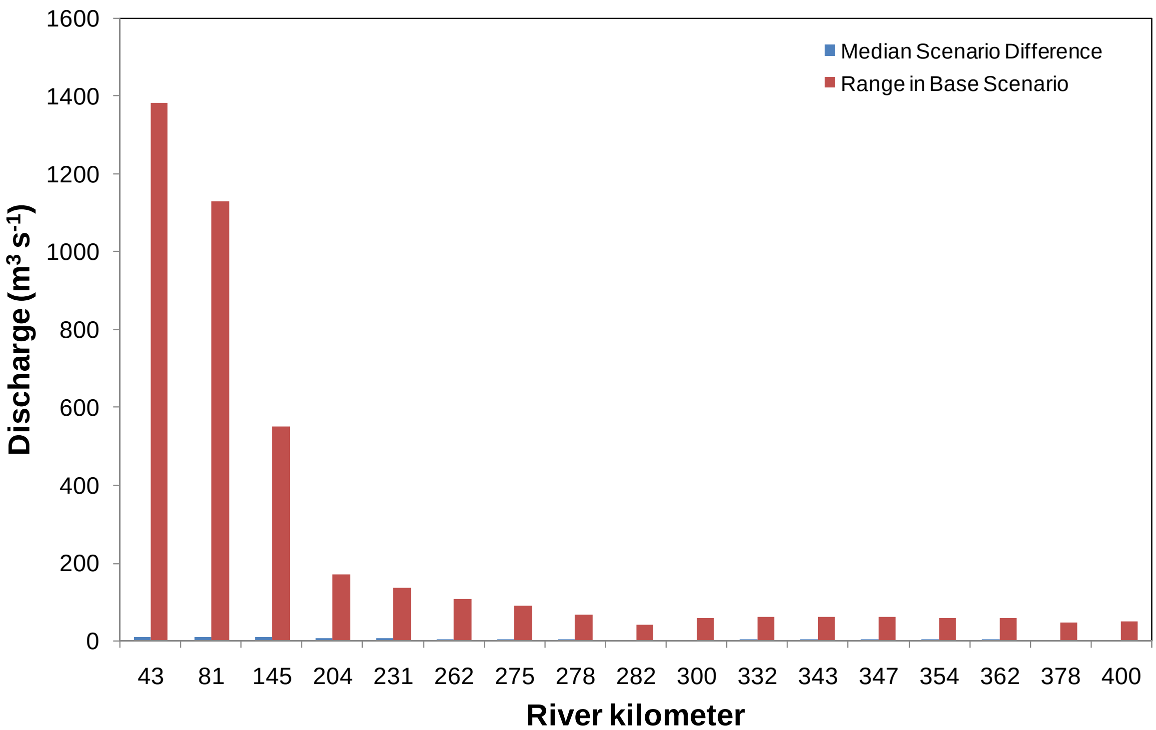

Figure 4-3. Longitudinal variation in modeled June discharge for the baseline condition (Base1995NN) and the median deviation from baseline condition June discharge for the most extreme test scenario (FwOR1995NN, downstream of the Ocklawaha River; Full1995NN, upstream of the Ocklawaha River).

Residence time showed a level of variability similar to the other hydrologic drivers (Figure 4-4). The median deviation from the baseline condition was -1.4 to 6.1 days, representing a -6 to 6% change compared to the baseline condition (Figure 4-4). Interestingly, residence time decreases slightly at RK 232 and RK 385. This modeled decrease reflects water withdrawals at locations below these points that draws water downstream and lowers residence times. Using the interquartile range, typical residence times can vary from 3 to 10 weeks. Therefore, the median change between scenarios represents 1-9% of the variation in the baseline condition. However, the disproportionate increase in the 75th percentile in the extreme test scenario suggests that the frequency of long residence time events could increase markedly in some areas of the river. The analysis of the Plankton Workgroup (Chapter 8), however, indicates that the ecological effects of these extended residence times would not significantly worsen the intensity or duration of phytoplankton blooms.

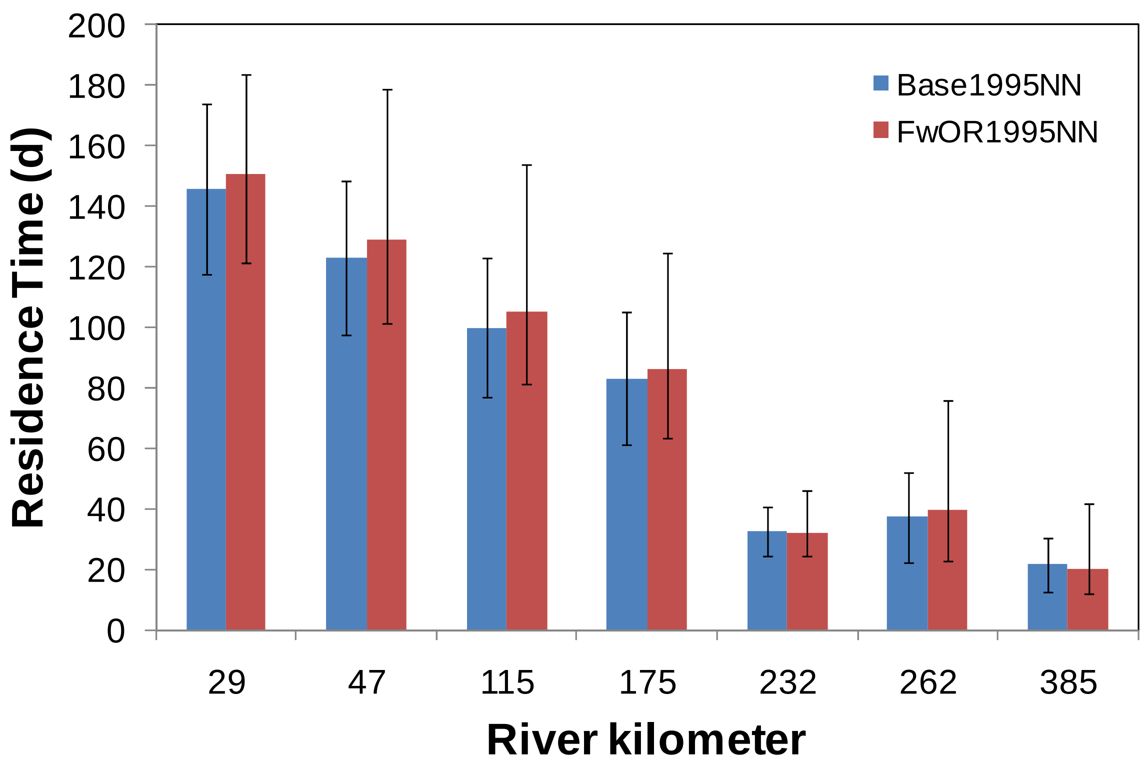

Figure 4-4. Longitudinal variation in modeled median 7-day residence times and deviations from the baseline condition for the most extreme test scenario (FwOR1995NN). Error bars indicate the 25th and 75th percentiles.

4.3 Relative Sensitivities of Ecological Attributes

There was considerable variation in the sensitivities of the ecological attributes to the modeled deviations from the baseline condition. Owing to the refractory nature of organic soils on the floodplain of segments potentially affected by deviations in water levels, we found no appreciable potential for effects associated with increased soil oxidation. Similarly, the potential distribution of submersed aquatic vegetation appeared to be relatively insensitive to modeled deviations in both water levels and salinities. Our analysis of the potential for changes in the distributions of wetland plant communities indicates that they would responds to deviations as large as those potentially associated with the most severe test scenarios. The response would be a change in elevational distributions of the communities and an associated change in the proportional representation of the communities within the floodplain wetlands. We found no potential for reductions in the aggregated total area of wetland plant communities.

The sensitivity to deviations in drivers seemed to be somewhat greater in ecological attributes for organisms occupying higher trophic levels. Animal communities (fish and benthic macroinvertebrates) appear to be more responsive to potential deviations in water levels and salinity than the lower level attributes (biogeochemistry, plankton, SAV). This is suggestive of augmentation of small, lower-level effects as they cascade upwards through food webs. Such augmentation of effects may be reflected in the relatively higher sensitivities of fish within the lower reach. The modeled deviations in population densities (as indicated by catch-per-unit-effort) for these species were often much greater than the modeled deviations in the primary driver (discharge) or in potential secondary drivers (salinity, changes in SAV, changes in macroinvertebrate communities). For floodplain wildlife, we lack quantitative HE models, so we cannot assess the potential for further augmentation of effects at trophic levels above the fish communities. However, at this juncture, it seems prudent to conclude that floodplain wildlife at higher trophic levels will be at least as sensitive to deviations in drivers as the fish communities.

This reasoning indicates that, for the eight segments of the river examined, plankton communities, biogeochemical cycles, SAV distribution and abundance, and wetland vegetation will be relatively insensitive to hydrologic changes of the scale considered in this analysis. Higher level ecological attributes, however, appear responsive to H&H changes of the magnitude of the modeled deviations. If deviations in these attributes are to be avoided, a precautionary approach to water allocations seems warranted.

4.4 Potential for Avoidance or Reduction of Withdrawal Effects

We considered whether there were apparent options for avoiding or reducing deleterious ecological effects that could be associated with additional surface water withdrawals. Only a few of the many ecological attributes considered have an appreciable modeled potential for deleterious effects under the hindcast and forecast scenarios.

An important potential deleterious effect is entrainment and impingement (E&I) of ichthyoplankton. These E&I effects could be avoided by appropriate design, orientation, and location of intake structures (Gowan et al. 1999) and by timing of water withdrawals so that they do not coincide with peak ichthyoplankton density. This study (details in Chapter 12. Fish) indicates where (segments 5-7; see Figure 3-5 and Figure 3-7) and when (February – May; Figure 3-14) to expect high abundance of ichthyoplankton and, thereby, can provide guidance locating and operating intake structures. According to Gowan et al. (1999), maximum intake velocity should not exceed 0.25 ft s^{-1} and sweeping velocity (the velocity parallel to the intake screen) should exceed the intake velocity.

The potential for deleterious water level effects can be substantially reduced by using variable withdrawal rates and by locating water withdrawals in sections of the river where water level and discharge are poorly correlated, such as in the middle and lower reaches.

The potential for both water level and salinity effects could be reduced by additional inflows of freshwater to the river. Our modeling indicates that this will occur due to intensification of land uses and completion of projects in the Upper St. Johns River Basin. In addition, rising sea level will increase mean water levels in the lower and middle reaches.

4.5 Impairment in the Baseline Condition

In considering whether deviations from the baseline condition would be acceptable, it can be important to consider whether there is pre-existing impairment in the baseline condition for the potentially affected ecological attributes.

4.5.1 Nutrient Loads and Plankton

Most reaches of the St. Johns River do not meet state water quality standards and have nutrient TMDLs or are listed as impaired for nutrients by the Florida Department of Environmental Protection. Furthermore, we found pre-existing major impairment with respect to algal blooms in all segments of the river that we assessed except segment 8 (Lake Poinsett), where the pre-existing impairment was minor (Chapter 8. Plankton).

4.5.2 Submersed Aquatic Vegetation

An impaired condition is generally established by comparing a metric of the current condition either to an established standard or to some antecedent baseline condition. Standards for SAV could include average spatial extent, depth distribution, species composition, or some other metric of community condition. No standards have been proposed or adopted for SAV in the St. Johns River. Therefore, a “standards” approach cannot be applied. Likewise, baseline SAV conditions have not been adequately established. Historical and anecdotal accounts provide some information but it is not quantitative and many physiographic features have changed. Reliable, quantitative SAV data were not collected until 1998 in the lower St. Johns River where salinity effects are expected. Data show that SAV is sensitive to drought patterns and consequent salinity levels. Variability in the current data set makes it difficult to even identify an average or typical condition. Furthermore, sensitivity of SAV to river flow and drought requires some knowledge of how the SAV responds to multi-decadal weather patterns and where the ecosystem sits within the pertinent weather cycle. Thus, existing SAV impairment in the baseline condition was not assessed.

4.5.3 Wetlands Vegetation

In the baseline condition, substantial areas of historical floodplain wetland of the upper reaches of the river were no longer wetland, particularly in the upper segments of the river (segments 8 and 9). This loss of wetland acreage can be considered an existing impairment. Further, baseline plant communities had been altered by changes in hydrologic and fire regimes, invasive species, logging, and increased nutrient loadings. In the lower reaches of the river, plant communities had responded to channel deepening and straightening, sea level rise, and other alterations that have affected discharge, salinity, and water levels. Plant community boundaries and species composition were altered by these factors.

4.5.4 Benthic Macroinvertebrates

Benthic macroinvertebrate communities have been used for decades as biological indicators of water quality (Gaufin 1973; Rosenberg and Resh 1993). Reduced dissolved oxygen (DO) due to organic or nutrient loading, toxicity due to heavy metals or other chemicals, and sediment loading from poor land use practices have all been shown to adversely affect various attributes of benthic communities, including taxonomic richness and diversity, biomass, functional characteristics, and physiology (Rosenberg and Resh 1993). Studies relating benthic macroinvertebrate communities to water quality in the St. Johns River indicate that water quality degradation in the river has affected the structure of the benthic community. In a synoptic survey of the river conducted in 1999, 72 of 148 lake and stream sites were classified as “severely impaired” based on characteristics of the benthic communities, which included density, taxa richness, and Shannon-Wiener diversity (Water and Air Research 2000). An additional 58 sites were classified as “moderately impaired” based on the benthic community characteristics. A study in the lower St. Johns River and estuary (Evans et al. 2011) indicated that at 40 benthic sampling sites, the proportion of pollution tolerant taxa in the benthic community ranged from 17-100%, with most sites exhibiting 80-100% tolerant taxa, indicating a considerable degree of impairment. Similarly, a sizeable fraction of the chironomids at these sites had deformities indicative of toxicity effects (Evans et al. 2011). Thus, the benthic macroinvertebrate communities of much of the St. Johns River basin reflect the documented water quality impairment in the river system.

4.6 Other Significant Drivers

It is important to appreciate that our analysis considered only the potential effects attributable to an increase in water withdrawal. Many other important ecological drivers can affect the ecological attributes we considered. Variation in the status of these other drivers can markedly alter the effects of a water withdrawal. In most cases, our ecohydrologic models used empirical data derived from the baseline period or from a more extended past period. If the status of other significant drivers should change from the status of the period used for development of a model, then the response of the attribute to water withdrawals could materially deviate from the prediction of the model. Other SJRWMD programs and projects will address the potential effects of these other drivers. Ongoing water quality monitoring will assess status and trends in pollutant concentrations and loadings. Regional water projects have been constructed and managed to improve water quality and reduce pollutant loadings. SJRWMD scientists and engineers have used, and will continue to use, modeling, monitoring, and data analysis to determine pollutant loading limits in order to protect water quality, designated uses of water bodies, and aquatic life.

4.6.1 Pollutant Loadings

The character of drainage basins influences many facets of aquatic ecosystems. The effects of projected changes in land use on hydrology were assessed by the WSIS. However, one of the most important non-hydrologic drivers affected by land use, pollutant loading rates, was not assessed.

Nutrient Loading

Nutrient pollution is an important consequence of urbanization because nutrient concentrations affect all trophic levels within an ecosystem. For example, phytoplankton density, an important attribute evaluated in this study, is strongly affected by the availability of nitrogen and phosphorus. If the levels of these nutrients increase, then deleterious phytoplankton blooms would likely become more frequent and more intense than in the baseline condition, even if residence times were not extended by water withdrawals. Conversely, the influence of elemental nutrients on phytoplankton is strong enough to prevent attainment of bloom densities if nutrient concentrations were reduced to sufficiently low levels.

It is difficult to predict the changes in pollutant loadings that could occur contemporaneously with land use changes and changes in climatic factors, such as rainfall and sea level. However, we expect that nutrient loads will decline below baseline condition levels, given that Pollutant Load Reduction Goals (PLRGs) and Total Maximum Daily Loads (TMDLs) for nutrients are either adopted or in development throughout the St. Johns River Basin (Table 4-1). The setting of nutrient reduction targets through the PLRG and TMDL programs indicates that progressive reduction of nutrient loads is likely.

Table 4-1. Pollutant Load Reduction Goals and Total Maximum Daily Loads for the St. Johns River and major tributaries.

BASIN	PLRGs/TMDLs	Constituents	Range of Reductions
Upper St. Johns River	20	TN, TP, BOD, DO	27% - 73%
Middle St. Johns River	20	TN, TP, BOD, DO, NOx, Fecal Coliform	10% - 85%
Lower St. Johns River	4	TN, TP	34% - 46%
Ocklawaha River	24	TN, TP, BOD, Total Coliform, Fecal Coliforms, Selenium, Iron, Silver	0% - 85%
Orange Creek Basin	5	TN, TP, Total Coliforms, Iron	31% - 74%

Metals Loadings

Although nutrient loadings may fall in response to the requirements of the Total Maximum Daily Loads (TMDLs), there are loadings of many other pollutants that could increase due to intensification of land uses. In many cases, loadings of metals have not been addressed by TMDLs. Typically, these metals are released into the atmosphere by coal-fired power plants and are deposited on the landscape through wet and dry deposition processes. Intensification of land use results in increased impervious surface and greater run-off of metals into receiving waters. Although the water management districts and FDEP have regulations in place to prevent or minimize water quality degradation in surface and ground water caused by new development, the present regulatory best management practices are insufficient to prevent some increase in toxic metal loads.

Water column concentrations of toxic metals are already high enough to impair parts of the main stem of the St. Johns River and many of its tributaries (FDEP, 2010). Mercury impairment occurs in most of the river main stem, in some tributaries (e.g., Sisters, Broward, Moncrief), and in many lakes including Crescent, George, Woodruff, Monroe, Poinsett, Washington and Blue Cypress. Thallium contamination has been found in the main stem of the river upstream of Doctors Lake. Part of the main stem in the Jacksonville area and the tributaries, Peters Branch and Kendall Creek, are impaired due to iron. Another tributary, McCoys Creek is impaired due to copper and zinc contamination. Other tributaries including Black, Simms, Peters and Deer creeks are impaired due to lead contamination (FDEP 2010).

As a case study, the H&H Workgroup modeled changes in watershed runoff and pollutant loadings from the Little Econlockhatchee River, and included zinc as a surrogate for metal contaminants (Chapter 3. Watershed Hydrology). They predicted zinc loading would increase by 5.6% due to changes in watershed conditions from the 1995 to 2030. They concluded that the implementation of Best Management Practices (BMPs) would fail to effectively control the increase of heavy metal loads associated with future land use changes.

The removal process for many toxic metals is similar to that for zinc because (1) they have a similar affinity for sorption to suspended particles and (2) removal is primarily by settling of suspended solids. A comparable increased loading of 5 to 6% can be generalized, then, for other metals. There are a few caveats for this conclusion. First, zinc may not be a good proxy for metal salts that are more soluble than zinc (e.g., silver nitrate, chromium (VI) oxide). Second, atmospheric loading of metals (e.g. mercury) could increase contemporaneously with intensification of land uses. Finally, the simulated removal efficiency rate of 80% for zinc by dry and wet detention ponds may overestimate zinc and other metal removal rates for some environmental conditions (Michigan Dept. Transportation 1998). Nevertheless, the case study demonstrates the potential for increases in metal loadings due to intensification of land uses.

Organic Pollutants Loadings

Organic pollutants are present in the sediments of the lower St. Johns River and its tributaries. These pollutants constitute an existing stressor on the biota. Water draining from urban areas contains many organic pollutants stemming from pharmaceuticals, household products and synthetic compounds produced by industry (Brezonik and Arnold 2011). As such, the loadings of organic contaminants can also increase with intensification of land uses.

The highest concentrations of sediment pollutants have been found in areas near Jacksonville, especially PAHs, PCBs and some pesticides, such as DDT, chlordane, lindane and endosulfan (Durell et al. 2004a; Sonnenberg et al. 2011). Several tributaries to the lower St. Johns River have shown severe contamination from PAHs, PCBs, metals and pesticides over the years (Durell et al. 2004a). Of particular concern is the large Cedar-Ortega basin, which has exhibited the most diverse assemblage of contaminants as well as the highest concentrations.

Rice Creek is another tributary to the St. Johns River that has had some of the highest concentrations of organic pollutants and elevated concentrations of PAHs, PCBs, chlordane, endosulfan and other chlorinated industrial compounds have been found in this area (Durell et al. 2004a). Farther upstream, slightly elevated concentrations of several chlorinated pesticides including chlordane, BHCs, and DDT have been found in a few areas of Lake George and in Lake Monroe (Durell et al. 2004b).

Loadings for some of these constituents have been reduced and, in some areas, sediment contaminant remediation projects are proposed. Dredging and sediment removal have been proposed for the river channel and Fishweir Creek, a tributary to the lower St. Johns River, which will also remove contaminants from these areas. Plans for remediation of Rice Creek include transfer of the contaminant discharge from the Georgia-Pacific Paper Mill to the St. Johns River. Natural attenuation of pollutant levels in sediments in Rice Creek is expected to follow. However, remaining pollutant loads will flow directly into the St. Johns River.

The case study of the Little Econlockhatchee did not consider the potential for increased loadings of organic pollutants. However, this potential can be coarsely estimated using data from the literature and the hydrologic modeling runoff results. Many dissolved organic contaminants, like metals, tend to sorb to suspended particles, and could be expected to settle with the suspended solids. This removal process parallels the metal removal, so an increase in loading similar to that modeled for metals, 5 to 6%, can be inferred for organic pollutants. As with metals and nutrients, it seems reasonable to conclude that implementation of BMPs would not prevent an increase in loadings of some organic pollutants.

4.6.2 CLIMATE CHANGE

Global climate change can result in changes to temperature and rainfall patterns, two key hydrologic parameters that affect the amount and intensity of runoff from watersheds. Global climate change can thus alter river discharge, a key hydrologic driver for the WSIS. Early in our study, we determined that there was no reasonable expectation that climate change would modify local climatology by the year 2030. The temporal scale of climate change was considered too long to appreciably affect hydrologic predictions over the 1995 to 2030 period used for the study. The NRC requested that potential hydrologic alterations to climate change be evaluated as part of the WSIS, however, and this evaluation was made to the year 2100.

The National Center for Atmospheric Research generated daily temperature and rainfall scenarios for the period 2020 to 2100 that are statistically consistent with the analyzed results of 21 Global Climate Models (GCMs). The results of the 21 GCMs first were used to develop a probability distribution of predicted outcomes for the IPCC A1B population and energy use scenario. The generated temperature and rainfall time-series next were supplied to the HSPF hydrologic model of the St. Johns River for prediction of river discharge.

The central tendency of the GCMs showed little or no change to annual precipitation or drought frequency by the year 2100, but did show an increase in rainfall intensity. There was an increase in winter precipitation, which reduced drought intensity. Mean temperature increased, primarily because of higher winter temperatures, particularly at night.

The increase in temperature caused greater evaporation in the HSPF hydrologic model simulations. Higher evaporation with little change in annual precipitation resulted in slightly lower river discharge in the 2100 scenario. The HSPF hydrologic model simulation does not consider the possible reduction in plant transpiration by increased atmospheric CO_2 levels that could offset the effect of temperature on evaporation. The simulated discharge reduction was small, however, and considered *de minimus* relative to the proposed water withdrawal.

The results of the global climate change analysis indicate rainfall patterns for northeast Florida are unlikely to change dramatically due to global climate change. This conclusion is subject to considerable uncertainty resulting from the global gas emission scenario and variability of the GCM output. Evaporation is expected to increase due to increased temperatures, particularly in winter. SJRWMD's response to hydrologic alteration by global climate change should be met by the same iterative process for addressing water quality issues using existing programs of long-term observation, data and trend analysis, continued updating of models, and development of management policies consistent with our understanding of the system.

4.6.3 Sea Level Rise

Sea level rise (SLR) has been occurring at the river mouth since at least the beginning of observations at Mayport in the 1920s. The average rate of SLR over the observed record at Mayport (1928–2010) is 2.4 mm yr^{-1}. This observed rate represents relative SLR and includes both global SLR, the rise of the world oceans, as well as local subsidence of land. Global SLR over the previous century is estimated at 1.7 mm yr^{-1}(CCSP 2009), indicating that land subsidence of the northeast Florida coast has been equivalent to global SLR over that period. The annual rate of relative SLR at Mayport has varied over the previous century and is presently estimated as 4 mm yr^{-1} (Rahmstorf 2007). It is highly likely that the rate of relative SLR will be equal to or greater than the present rate over the next century.

Changes in sea level have occurred throughout geological history and were central to shaping the hydrogeologic features of the present day St. Johns River. Following the last glaciation, sea level rose 100-150 m between 15,000 and 6,000 years ago. Sea level has remained relatively stable over the last 6,000 years with fluctuations of 1-2 meters occurring over several hundreds of years (National Research Council 1987). Global warming, primarily caused by an increase in atmospheric carbon dioxide, can increase the rate of global SLR by thermal expansion of ocean water and melting of glaciers and ice sheets.

SLR can affect both natural systems and human infrastructure. Effects on human infrastructure generally focus on coastal areas where seawalls and structures are exposed to waves and shoreline erosion and where storm surge can cause inundation of residential and urban areas (National Research Council 1987). Effects on natural physical features focus on beach and shoreline erosion, salinity increase up estuaries, and salinity intrusion into coastal aquifers. Effects on biological systems focus primarily on loss of wetland habitats, but also on shallow water habitats such as SAV (CCSP 2009). Loss of wetlands and shallow water habitats are dependent on how the rate of SLR compares with the rate at which natural systems can adjust. At moderate rates of SLR, for example, tidal marshes can maintain themselves by vertical accretion and horizontal migration, but if SLR exceeds the rate at which the marsh can adjust, then the marsh converts to open water.

For the WSIS, we examined how SLR alters the hydrodynamic drivers of water level, salinity, and water age for the period 1995 to 2030. We examined two rates of relative SLR: the present rate (4 mm yr^{-1}) and a moderate global warming scenario (8 mm yr^{-1}) (USACE 2009). These rates result in sea level increases of 14 and 28 cm, respectively, over the 35 yr period considered.

Because of the extreme low slope of the lower and middle St. Johns River (0.27 cm km^{-1}), SLR effects extend throughout the lower 300 km of the river to Lake Harney. The difference in effects between the two SLR scenarios on hydrodynamic variables is essentially linear, that is, the effect of a 28 cm increase in sea level is double the effect of a 14 cm increase. For this reason, we present only the effect of a 28 cm increase here.

SLR will continue to increase water level throughout the lower and middle St. Johns River and offsets reduction of water level caused by water withdrawal. The greatest reduction of water level due to a water withdrawal in the lower and middle St. Johns River occurs in Lake Harney where maximum withdrawal resulted in a mean water level reduction of 4 cm. In comparison, the moderate global warming scenario increases mean water level in Lake Harney 18 cm, a net increase in water level of 14 cm. In downstream areas, the effect of water withdrawal on water levels is less and the effect of SLR is greater, so that mean water levels in the lower river increase nearly 28 cm.

SLR increases salinity in the lower St. Johns River, with the greatest increase occurring in the channelized portion of the river below river km 20. SLR, then, enhances the effect on salinity of water withdrawal. The moderate global warming scenario increased mean salinity 0.8 psu at RK 20. For comparison, a 6.79 m s^{-1} (155 mgd) water withdrawal increased mean salinity 0.3 psu at this location. This portion of the river is predominately polyhaline with a mean salinity of 22.8 psu and a range of 10 to 32 psu. SLR had a negligible effect on salinity in the oligohaline reach of the river where the effects of increased salinity on SAV are a concern.

SLR increases water age throughout the river, likely because of increased volume. The greatest effect of SLR on water age occurs in Lake George (RK 190) where mean water age increased 8 days. For comparison, a 6.79 m s^{-1} (155 mgd) water withdrawal increased mean water age in Lake George by 4 days. These increases are modest relative to the baseline water age, which has a mean of 56 days and a range of 21 to 117 days.

The greatest effect of SLR relative to the WSIS is its effect on water level in the lower and middle St. Johns River. SLR is the dominant factor affecting water levels over the lower 300 km of the river and completely offsets the lowering of water levels by water withdrawal. Over the next century, water levels will continue to rise in this reach and rates of SLR could continue to increase. The rate at which adjacent salt marshes and tidal freshwater marshes can adjust to increased rates of SLR will determine the fate of these communities. The St. Johns River may be unique in having large areas of freshwater marshes and forested wetlands affected by SLR. The mechanisms controlling adjustment of wetland communities adjacent to the river should be evaluated for long-term management of the system.

The effect of SLR on salinity and water age in the St. Johns River appears relatively modest. Salinity will continue to increase slowly in the future, but the alteration of salinity regimes over even the next century is unlikely. The WSIS did not consider the effects of increasing depth on either SAV or phytoplankton production. Similar to marshes, SAV communities will need to adjust at a rate commensurate with the rate of future SLR by either vertical accretion or horizontal migration. For phytoplankton, the increased depths will cause a net reduction of light, since many areas of the river (e.g. Lake George) now have depths greater than the photic zone depth. Consideration of SLR and global warming scenarios remains a valid concern for management of the river, but are not equivalent in priority to immediate concerns of water quality related to intensification of land use.

4.6.4 Channel Dredging

The lower St. Johns River has a long history of channel improvement beginning in 1880 with construction of two jetties at the river mouth. Jacksonville was already a thriving port at this time, but ship passage through the mouth was particularly treacherous because of shifting sands that rapidly altered the narrow and shallow natural channel. The completion of the jetties in 1895 stabilized the channel at the mouth and scoured the shallowest portion of the channel from 6 to 15 feet. Depths upstream of the mouth were 18 feet to Jacksonville (river km 40), aided by limited dredging between 1892 and 1895. Following completion of the jetties, the navigational channel to Jacksonville was deepened five times: to 24 ft in 1902, to 30 ft in 1916, to 34 ft in 1951, to 38 ft in 1978, and finally to 40 ft in 2003. The 1951 dredging was accompanied by a major alteration of the river channel with the creation of the Dames Point Cut between Fulton Point and Dames Point. Spoil from the creation of the Dames Point Cut was used to create Blount Island, now site of a major marine terminal.

At present, the U.S. Army Corps of Engineers Jacksonville is examining channel deepening scenarios that would allow post-Panamax ships to reach cargo terminals at Blount Island and perhaps Jacksonville. Post-Panamax ships require a 50 ft channel depth. The size, scope, and economic viability of a channel deepening project are under study and the scope of the final project is uncertain. The channel deepening scenario used for this study is conservatively large and assumes creation of a 50-ft (NGVD29) navigational channel from the jetties at the mouth of the St. Johns River to Jacksonville (Talleyrand Terminal) and including the north Blount Island channel.

The channel deepening scenario (CHND2030PS, see Chapter 6. River Hydrodynamics Results for details), had no effect on water level or water age. Salinity downstream of Jacksonville increased for the channel deepening scenario, while the effect on salinity upstream of Jacksonville was negligible. The greatest increase in salinity occurred near Dames Point (RK 27) where mean salinity increased 2 psu. Although this increase in salinity is large relative to the mean increase caused by water withdrawal (0.3 psu), it occurs in a polyhaline segment of the river where mean salinity is 22.8 and salinity ranges from 10 to 32 psu. The increase in salinity caused by the channel dredging did not shift the general salinity regime at any location in the river.

The environmental effects of channel dredging were not examined for the WSIS. The US Army Corps of Engineers Jacksonville is presently undertaking a thorough study of the physical, ecological, and economic costs and benefits associated with a range of possible channel deepening options for Jacksonville Harbor. SJRWMD and FDEP are reviewers of that work and will continue oversight of future dredging projects.

4.7 Adaptive Management and the Precautionary Principle

Adaptive management and the precautionary principle are concepts designed to cope with the need to make management decisions for natural systems under conditions of scientific uncertainty (Shipworth and Kenley, 1999; Prato, 2003). The uncertainty associated with scientific predictions for complex ecosystems, such as the St. Johns River, especially when compounded by the potential for changes in other important drivers, indicates a need for adaptive management. A critical step in adaptive management is monitoring to detect trends in ecological drivers and ecological attributes (Moir and Block 2001). Monitoring, accompanied by thorough analysis of the monitoring data to assess status and trends, is the means by which the predicted potentials for change can be verified or refuted (Figure 4-5). If model predictions are found to be inaccurate, then models will need to be re-examined, adjusted, and used to make new predictions that more closely approximate ecosystem responses. As the cycle of adaptive management continues, our understanding of the ecosystem will improve and our predictive models will improve. These improvements will enhance our ability to utilize water resources without eliciting unacceptable levels of ecological harm.

The precautionary principle recognizes that some decisions that carry risk (defined as the product of the probability and the seriousness of an event) must be made in the face of uncertainty. It argues that "… if the probability and the magnitude [of an event] are relatively unknown, because, for instance, it is not known what cause and effect relationships are involved, or exactly what the nature of the involved causal relationships is, then the standards would be precautionary because of the relative uncertainties involved" (Shipworth and Kenley 1999). In its essence, the principle is that when a proposed action has a predicted effect that is significant and uncertain, then its proponents must demonstrate its safety; otherwise, decisions should favor the environment (Shipworth and Kenley 1999). The intent of the principle is to reduce the potential for environmental harm before there is strong proof for harm (Harremoes et al. 2001).

Adaptive management fits well with the precautionary principle. It offers a means for discerning whether there are early indications of harm; and, if there are none, it strengthens the case for the safety of the action. Applying both principles argues for a stepped approach to use of surface water. First, the effects of withdrawal levels that had no more than minor modeled effects would be monitored to reduce uncertainty and to ensure the safety of withdrawals at that level. Then, if the observed effects of the initial withdrawal levels were, indeed, found to be no more than minor, higher levels of withdrawals would be considered, if they are needed.

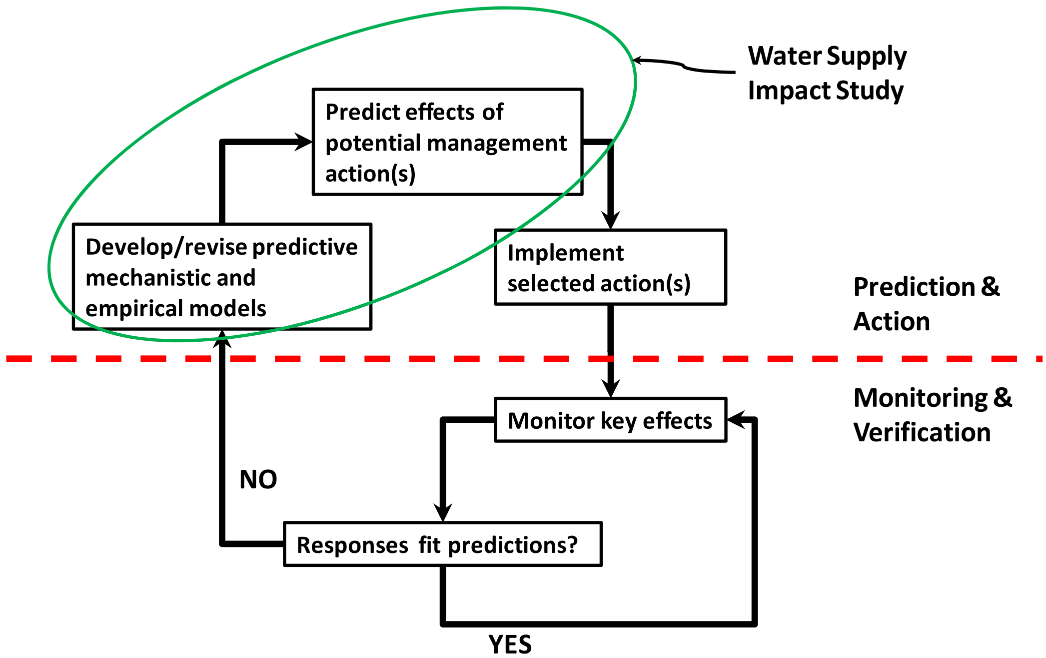

Figure 4-5. The cycle of adaptive management and the Water Supply Impact Study.

5 GENERAL CONCLUSIONS

Our analysis indicates that many ecological attributes could be affected by additional surface water withdrawals. In most cases, however, the effects would be sufficiently small to be considered negligible or minor. We found that there are hindcast and forecast scenarios under which the full withdrawal from the St. Johns River allowed by established MFLs (6.8 m^3 s^{-1}; 155 mgd), would have no more than minor effects on ecological attributes. We also found, however, that forecast scenarios that include water withdrawals from the Ocklawaha River (11.5 m^3 s-1; 107 mgd) would have moderate effects on fish populations and floodplain wildlife in the lower reach of the river. Further, we found that completion of the USJRB restoration projects, additional runoff expected from intensification of land use, and sea level rise significantly reduced the potential for effects on ecological attributes. Lacking these conditions, full withdrawal test scenarios had more than minor modeled effects for several attributes. The only scenario that had no appreciable reduction effects was Half2030PS: [3.4 m3 s^{-1} (77.5 mgd); completed USJRB restoration projects, 2030 land use, higher sea level].

With the increase in discharge predicted to be associated with intensification of land uses, a phenomenon that is expected to accompany an increase in need for public water supplies, additional surface water withdrawals will cause few reduction effects with respect to the baseline condition. Consequently, the environmental effects of the modeled water withdrawals for hindcast and forecast scenarios were in most cases negligible to minor.

There is scientific uncertainty associated with our findings that argues for a precautionary approach utilizing adaptive management. Consequently, it seems prudent to use a stepped approach wherein initial increases in surface water withdrawals entail a margin of safety. If monitoring confirms the predictions upon which initial allocations were based, then additional allocation could be considered. Using this stepped approach with adaptive management will ensure that environmental harm does not become an unintended consequence of increased surface water withdrawals.

It is important to recognize that the sensitivity of the river to water withdrawals varies significantly along its length. Monitoring within the context of adaptive management of water withdrawals should consider this variability. This study provides good guidance with respect to the locations where effects from various H&H drivers would be strongest.

Because changes in other important drivers of the ecological attributes considered could affect their responses to water withdrawals, and because there is scientific uncertainty associated with predictions, adaptive management is the prudent approach to development of additional surface water sources. Moreover, it would be prudent to work towards incorporating the effects of other drivers into the modeling sequence.

In addition, we note that, in most cases, empirical models were used to model the potential ecological effects. These empirical models do not reveal the underlying mechanisms of cause and effect. To bolster our ability to adaptively manage the river system and to reduce the potential for ecological harm, the causal mechanisms underlying the empirical relationships should be further investigated.

The findings of the WSIS are specific to the modeled scenarios but proposed water withdrawals can use the WSIS as guidance for reducing the potential for adverse environmental effects. The models and scientific understanding developed in the WSIS provide a sound basis for conceptualizing and evaluating further development of the St. Johns River as an alternative water supply.

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7 APPENDICES **See separate document**