CHAPTER 11. BENTHIC MACROINVERTEBRATES

by

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Acronyms, Abbreviations, and Conversion Factors

μmhos	micromhos—a unit of measurement of conductivity
μm	micron
AW	Alligator weed
BCMCA	Blue Cypress Marsh Conservation Area
BCPOM	Benthic course particulate organic matter
BFPOM	Benthic fine particulate organic matter
BIO-ENV	Trade name of software
CPOM	Coarse particulate organic matter
DO	Dissolved oxygen
EFDC	Environmental Fluid Dynamics Code
EMAP-E	Environmental Monitoring and Assessment Program-
F.A.C.	Florida Administrative Code
FDEP	Florida Department of Environmental Protection
FFG	Functional feeding group
FPOM	Fine particulate organic matter
FWC	Florida Fish and Wildlife Conservation Commission
GIS	Geographic Information System
HSPF	Hydrologic Simulation Program-FORTRAN
IFAS	Institute of Food and Agricultural Sciences
INVERTCAL	Trade name of software
LCI	Lake Condition Index
LED	Limiting environmental difference
LGB	Lake George basin
LSJRB	Lower St. Johns River basin
MDS	Nonmetric multidimensional scaling
MFLs	Minimum flows and levels
MSJRB	Middle St. Johns River basin
NA	Not applicable
NGVD29	National Geodetic Vertical Datum, 1929
NR	Not ranked
NTU	Nephelometric Turbidity Units
OR	Ocklawaha River
PAH	Polycyclic Aromatic Hydrocarbons
PCB	Poly-chorinated Biphenyls
PCU	Platinum Cobalt Units
RELATE	Trade name of software
SAV	Submersed aquatic vegetation
SCI	Stream Condition Index
SFPOM	Suspended fine particulate organic matter
SIMPROF	Trade name of software
SJR	St. Johns River
SJRWMD; District	St. Johns River Water Management District

SLR	Sea level rise
SM	Shallow marsh
SR	State Road
TDS	Total dissolved solids
USEPA	United States Environmental Protection Agency
USGS	United States Geological Survey
USJRB	Upper St. Johns River basin
WP	Wet prairie

1 ABSTRACT

Benthic macroinvertebrate communities in aquatic ecosystems are significant both ecologically and economically. They transform primary production by plants into animal biomass for use by higher trophic levels and are also a key food base for fish and wildlife of recreational and commercial significance. Benthic macroinvertebrate communities in freshwater portions of the St. Johns River system are expected to respond to hydrologic changes, based on studies in other freshwater ecosystems throughout the world, but there have been no studies of the effects of hydrology on benthic communities in the St. Johns River main stem (there have been a limited number of studies in tributaries of the river, but these cannot be used to assess mainstem effects) because of differences between characteristics of the tributaries and the river main stem. Because of the lentic nature of the St. Johns River main stem, changes in water level are likely to be more significant to benthic macroinvertebrate communities, ecologically, than changes in flow. In the upper St. Johns River basin, water level reductions of 7 to 10 cm at certain flows may have negative effects on benthic macroinvertebrate communities due to loss of habitat, but hydrologic changes due to construction of the upper basin projects and future land use will mitigate the effects of water level reductions. Downstream, water level reductions in response to withdrawal are substantially dampened, with fewer effects on benthic macroinvertebrate communities. In the estuary, changes in freshwater inflow will primarily affect benthic macroinvertebrate communities by way of changes in salinity regimes. Salinity was the main driver influencing benthic infaunal community structure, and highest total abundance was seen at lowest salinities (< 5 ppt). Salinity changes due to upstream withdrawals projected by the Environmental Fluid Dynamics Code (EFDC) hydrodynamic modeling were generally small (see Chapter 6. River Hydrodynamics Results) and had minimal impact on benthic macroinvertebrate communities and the commercially and recreationally important species blue crab (*Callinectes sapidus*) and penaeid shrimp, principally the white shrimp (*Litopenaeus setiferus*).

2 INTRODUCTION

Benthic macroinvertebrates are an important group of animals associated with bottom habitats in freshwater, estuarine, and marine aquatic ecosystems. They are generally defined as those invertebrate organisms retained by a mesh size of 200 to 500 µm (Stickney 1984; Rosenberg and Resh 1993). From a management perspective, benthic macroinvertebrates have been used for decades as key biological indicators of water quality and indicators of biological integrity (Gaufin 1973; Rosenberg and Resh 1993; Davis and Simon 1995). More recent efforts have employed benthic macroinvertebrates to assess habitat conditions, effects of hydrologic alteration, and water quality (Boon et al. 1992; Gore et al. 2001), and also to serve as surrogates for ecosystem characteristics (Merritt et al. 1996; 2002; 2008).

Ecologically, benthic macroinvertebrates are primary consumers of plant material (live and detrital) and predators in aquatic food webs. They transform primary production into animal biomass for use by higher trophic levels in aquatic food webs (Cummins et al. 2008). Across a variety of aquatic ecosystems, from freshwater to marine, they are principal consumers of plant production—filtering phytoplankton and detrital plant particles from the water column, grazing on periphytic microalgae, consuming macroalgae or vascular macrophytes in bulk, or digesting organic material accumulated in or on bottom sediments. Macroinvertebrates are key components of the diets of a number of higher animal taxa of interest to people, including recreational and commercial fishery species and wildlife species of conservation concern. Benthic macroinvertebrates are also a major component of the overall faunal biodiversity in aquatic ecosystems; total taxa richness of invertebrates may exceed that of all vertebrate groups combined by two to three times. In consideration of these features, benthic macroinvertebrates were one of the principal elements of focus for the St. Johns River Water Management District (SJRWMD; District) assessment of potential ecological effects of surface water withdrawals on the St. Johns River and estuary ecosystem.

2.1 Benthic Macroinvertebrate Communities of the St. Johns River and Estuary

Historically, benthic macroinvertebrate monitoring and research in the St. Johns River drainage have been limited to short-term, focused efforts in selected portions of the basin. There has been no routine, long-term macroinvertebrate monitoring ($\geq$ 10 years of sampling) anywhere in the basin. Based on the studies conducted, 1,063 species of freshwater, estuarine, and marine invertebrates have been collected in the St. Johns River and estuary ecosystem (Appendix 11.A). Dominant taxa (based on numbers of species) in freshwater reaches include aquatic insects, mollusks, and oligochaete worms. In estuarine reaches, dominant taxa measured by species richness include mollusks, crustaceans, and polychaete worms (Appendix 11.A).

Table 2–1 summarizes past studies of benthic macroinvertebrate communities (benthic community or benthic communities) conducted in the St. Johns River basin. There are a few studies we likely did not find, conducted by private consulting firms for various industries and utilities in the basin (D. Blancher, Sustainable Ecosystem Restoration, LLC, pers. comm. 2011). These are difficult to access, and the data were not available to us. The overall amount of benthic community sampling in the middle and upper St. Johns River has been very limited, and the data that do exist are not suitable for analysis of the effects of hydrology on freshwater benthic communities, or they were collected in tributaries and cannot be used to assess mainstem effects. Somewhat more benthic sampling has been conducted in the lower St. Johns River and in the estuary (Table 2–1).

Table 2–1. Summary table showing previous benthic community sampling projects in the St. Johns River.

Description and Sampling Agency	Basin	Period of Record	Reference
Freshwater			
One time sampling in main river channel (petite ponar grab) and tributaries; SJRWMD staff and contractor	Throughout St. Johns River basin in freshwater reaches	January to June 1999	Water and Air Research, Inc. 2000
SCI and BioRecon sampling (dip net) in tributary streams; FDEP staff	Basinwide	1992 to current	FDEP 2004 (Bioassessment Ecosummaries website: http://tlhdwf2.dep.state.fl.us/eswizard/eco_query.asp)
Intermittent sampling in river channel (petite ponar grab and Hester-Dendy samplers); FDEP staff	USJRB	1974 to 1992	D. Denson, unpubl. FDEP data
One time sampling in selected lakes and wetland habitats; Stetson University	USJRB	Dec 2001	Work and Gibbs, n.d.
Bimonthly sampling in Blue Cypress Lake – nine sites; FWC staff	USJRB	May 1998 to April 1999	Warren and Hohlt, n.d.
Qualitative sampling in marsh habitats in Blue Cypress Marsh Conservation Area (BCMCA); SJRWMD staff	USJRB	Unknown	M. Minno, unpubl. SJRWMD data
One time sampling in marsh habitats in BCMCA with throw trap; contractor for SJRWMD	USJRB	August 2002	Evans et al. 2004a
Irregular interval sampling in marsh habitats in the upper basin; contractor for SJRWMD	USJRB	August 1992 to November 1995	F. Jordan, unpubl. data
Sampling of chironomid midge populations (larvae and adults) in Lake Monroe at 16 sites; UF-IFAS	MSJRB	January 1979 to December 1987	Ali 1989
Monthly sampling of soft-sediment benthic communities in Lake Jesup at 15 sites; UF-IFAS	MSJRB	December 1996 to December 1997	Ali et al. 2003
Sampling of benthic communities in various aquatic habitats (multiple gear types) in the Wekiva River; FWC staff	MSJRB	May to June 1997 and Oct 1997	Warren et al. 2000
Sampling of benthic communities in various aquatic habitats (BioRecon) in the Little Wekiva River; contractors for SJRWMD	MSJRB	1999 to current	Water and Air Research Inc. 2008; Entrix 2010
Annual and biennial sampling of benthic communities with SCI, BioRecon, and LCI in small streams and lakes of Seminole County	MSJRB	2003 to current	M. Pluchino, Seminole County pers. comm. 2011

Description and Sampling Agency	Basin	Period of Record	Reference
Monthly sampling of sediment benthic communities in Blackwater Creek and Rock Springs Run (Eckman grab); UCF	MSJRB	February 1993 to January 1995	Lobinske et al. 1997
Intermittent sampling in river channel (petite ponar grab and Hester-Dendy samplers); FDEP staff	MSJRB	1975 to 1992	D. Denson, unpubl. FDEP data
Monthly sampling of *Hexagenia limbata* populations in Blackwater Creek and Rock Springs Run (Eckman grab); UCF	MSJRB	February 1993 to January 1995	Lobinske et al. 1996
Monthly sampling of chironomid midge populations (larvae and adults) in Blackwater Creek and Rock Springs Run; Univ. of Central Florida	MSJRB	February 1993 to January 1995	Lobinske 1995
Quarterly sampling of Asian clam (*Corbicula*) populations in the Wekiva River; UCF	MSJRB	August 1976 to June 1977	Gottfried and Osborne 1982
Short-term study of benthic communities in Econlockhatchee River using BioRecon and SCI; FDEP	MSJRB	January and July 1999	FDEP 2000
One time sampling of crayfish and other crustaceans at selected springs; contractor for FDEP	MSJRB/LGB	January to June 2002	Franz 2002
Intermittent qualitative sampling of unionid mussel populations in selected springs; USGS	MSJRB/LGB	2000 to 2002	Walsh and Williams 2003
Qualitative sampling (SCI) of benthic communities in selected springs; USGS	MSJRB/LGB	Varied by spring; overall January 1994 to October 2007	Walsh et al. 2009
One-time sampling of benthic communities (litter bag samples) in 18 small tributaries of the lower river; UA (contractor to SJRWMD)	LSJRB	2003 to 2004	Chadwick et al. 2006
Estuarine			
Quarterly sampling of benthic communities in various habitats (multiple gear types) in the lower river and estuary; contractors for SJRWMD	LSJRB	October 1993 to June 1995	Cichra and Adicks 1998; Mason 1998
Quarterly sampling of benthic communities associated with SAV habitat; SJRWMD staff	LSJRB	April 2002 to current	SJRWMD unpubl. data; Montagna et al. 2008a
Intermittent sampling in river channel (petite ponar grab and Hester-Dendy samplers); FDEP staff	LSJRB	1973 to 1996	L. Banks, unpubl. FDEP data

Description and Sampling Agency	Basin	Period of Record	Reference
One time sampling of benthic communities (petite ponar grab) at 20 sites in the lower river main stem and selected tributary creeks; SJRWMD staff and contractor	LSJRB	March to August 2000	Evans and Higman 2001
One time sampling of benthic communities (petite ponar grab) at 20 sites in the lower river main stem and selected tributary creeks; SJRWMD staff and contractor	LSJRB	October 2002 to August 2003	Evans et al. 2004b
One time sampling of benthic communities (modified Van Veen grab) at seven sites in the estuary; USEPA EMAP Program	LSJRB	2000	V. Engle unpubl. EMAP data
Short-term study of damselfly populations in SAV habitat in the lower river and estuary	LSJRB	May 1996 to August 1997	DeSalvo 2000
Short-term study of shrimp populations in the genus *Palaemonetes* in SAV habitat in the lower river and estuary	LSJRB	August to November 1996 and February to May 1997	Soulen 1998

Note:
SCI = Stream Condition Index
USJRB = Upper St. Johns River Basin
BCMCA = Blue Cypress Marsh Conservation Area
UF-IFAS = University of Florida–Institute of Food and Agricultural Science
MSJRB = Middle St. Johns River Basin
LCI = Lake Condition Index
UCF = University of Central Florida
FDEP = Florida Department of Environmental Protection
USGS = United States Geological Survey
UA = University of Alabama
LGB = Lake George Basin
LSJRB = Lower St. Johns River Basin
SAV = Submersed aquatic vegetation
USEPA = United States Environmental Protection Agency
EMAP = Environmental Monitoring and Assessment Program

2.2 Hydrologic Drivers of Benthic Communities and Populations

2.2.1 Water Level and Flow

Dewson et al. (2007) provided a detailed review of the effects of river hydrology on benthic communities in lotic ecosystems; and Merritt and Cummins (2008) provided an annotated list of literature that could be used to assess effects of hydrologic alteration in the St. Johns River. Various components of the hydrologic regime in rivers (e.g., water velocity, water level and depth, wetted perimeter) influence benthic communities, affecting abundance, taxa richness and diversity, species composition, behavior (e.g., drift), and species interactions. These changes may ultimately affect ecosystem function (productivity, etc.). Changes in flow may also influence water quality (temperature, dissolved oxygen, sedimentation) and habitat characteristics (amount of inundated habitat), which will be reviewed in more detail in relevant sections below. Benthic community attributes, such as diversity, abundance, etc., exhibited mixed responses to reductions in flow, in some studies increasing and in others decreasing (Dewson et al. 2007; Poff and Zimmerman 2010).

In lakes and wetlands, water level and its associated characteristics (duration, timing, predictability, and permanence) are the major drivers influencing benthic community characteristics (Ward 1992; Wissinger 1999), particularly in shallow littoral marsh areas in lakes, river channels, and floodplain wetlands. River floodplains and their associated wetland communities constitute a distinct type of temporary aquatic habitat (Williams 1996), where water level and its characteristics (magnitude, frequency, and duration of various levels) are major influences on benthic communities (Williams 1996). Lillie (2003) studied the effects of duration of inundation on benthic communities in isolated wetlands in Wisconsin. A duration of inundation of $\geq$ 8 months appeared to separate wetlands with a benthic fauna of permanent aquatic species (e.g., amphipods, mayfly nymphs) from those with a fauna composed of species adapted to temporary aquatic habitat conditions (e.g., fairy shrimp, mosquito larvae). Williams (1996) found similar results in three water-filled ditch systems in Canada, with a fauna composed of more aquatic insect taxa in ditches inundated 8 to 11 months versus 6 months each year.

2.2.2 Salinity

In estuarine ecosystems, freshwater inflow (hydrology) affects benthic communities primarily because of salinity regime alterations (Day et al. 1989; Montagna et al. 2002a); therefore, preservation of a protective salinity regime is a function of managing inflows to estuaries (Alber 2002). Montagna et al. (2008a) provided a review of the general effects of freshwater inflow and salinity on estuarine benthic communities and of specific inflow effects in estuaries of Texas and Florida. In Texas estuaries, freshwater inflow affected both benthic community productivity and community structure. Responses to salinity were nonlinear in many cases, and various components of the benthic community (i.e., abundance, biomass, and diversity) exhibited different salinity optima (Montagna et al. 2008a). In Florida estuaries, estuarine salinity structure (in terms of isohaline position) responded to changes in freshwater inflow in a curvilinear fashion. Salinity was the variable that explained most of the variation in mollusk communities in several southwest Florida estuaries (Montagna et al. 2008b).

Changes in estuarine salinity can also affect critical macroinvertebrate habitats. In the lower St. Johns River and estuary, beds of submersed aquatic vegetation (SAV) dominated by wild celery (*Vallisneria americana*) are an extensive and important habitat for benthic macroinvertebrates. *V. americana* is tolerant of moderate levels of salinity (Sagan 2009; Chapter 9. Submersed Aquatic Vegetation), but can be adversely impacted if salinities get too high (Chapter 9. Submersed Aquatic Vegetation).

2.2.3 Effects of Flow and Level on Water Quality Characteristics

Changes in water flows and levels may influence benthic communities via alterations in water quality (Williams 1996; Dewson et al. 2007). The variable most often considered is dissolved oxygen concentration (DO). It might be predicted that lower DO is associated with lower flows and water velocity, due to reduced mixing. However, Dewson et al. (2007) found no evidence in the literature that reduced or low flows affect DO levels, although there were limitations on the data used for documentation. DO levels may be affected more by increases in water temperature associated with lower flows (Allan 1995). Many invertebrate taxa appear to tolerate some degree of reduced DO (Merritt and Cummins 2010), with some taxa (e.g., stoneflies, mayflies, and caddisflies) more sensitive than others (e.g., chironomids, oligochaetes). Up to 30-day durations of reduced DO were sometimes required to induce lethal responses; and current was shown to mitigate reduced DO due to provision of a constant, reliable supply of DO (Merritt and Cummins 2010). In estuaries, low DO (hypoxia) affects benthic community structure and function, with more severe effects generally seen at DO levels below 2 to 3 mg L^{-1} (Diaz and Rosenberg 1995; Ritter and Montagna 1999).

The other water quality variable related to flow often considered when assessing effects of hydrology is sedimentation. Dewson et al. (2007) indicated that increased sedimentation has been associated with lower flows, due to lower water velocities reducing the capacity to carry suspended sediment in the water column. Fine sediment deposition affects benthic communities via effects on primary producers, direct effects on benthic invertebrates (i.e., physiology, behavior, or habitat), or effects on fish predators (Wood and Armitage 1997).

2.2.4 Effects of Flow and Level on Habitat

Changes in river flow or water level affect macroinvertebrate habitat in various ways. Changes in flow can influence water and sediment quality as described in Section 2.2.3. In the estuary, changes in salinity can affect important benthic habitats, such as submersed aquatic vegetation or oyster reef, which will influence benthic community composition and abundance. On the floodplain, various wetland plant community types exist based on inundation characteristics, primarily water depth and duration of inundation and soil saturation (Chapter 10. Wetland Vegetation; Lowe 1986; Light et al. 2002). Changes in water level may affect habitat by

- Reducing areal coverage of key habitats for various invertebrate taxa
- Stranding of pupal stages of aquatic insect taxa
- Creation of zero-flow, anoxic backwaters

Figure 2–1 presents the conceptual linkages among water levels, habitat, benthic communities, and higher trophic levels.

Changes in benthic communities can be predicted as a response to changes in river level because of different vulnerabilities of various taxa. Semivoltine taxa, which have life cycles longer than one year, and those with poor migratory capabilities would be most affected by reduced water levels. That is, they would not readily move with the water as levels recede or would not be able to rapidly recolonize dewatered areas once they were wetted again. Examples would be found in the Odonata, Megaloptera, Lepidoptera, and Coleoptera. By contrast, uni- and polyvoltine taxa, which have annual or shorter life cycles, would be expected to adjust to changing water levels more readily because of their rapid reproduction. Any taxa with nonmotile pupal stages and/or very slow movements potentially would be seasonally vulnerable to stranding when water levels decline, especially if the water withdrawal is rapid and of large volume. Because spatterdock *(Nuphar advena)* and eelgrass *(Vallisneria americana)* plant beds occur in areas along the river channel from shallow to deep water, the relative effect of stranding in these habitats would be far less than on floodplain marsh communities at higher elevations. Macroinvertebrates that have obligate associations with those higher elevation habitats would be more vulnerable to water level decline.

2.2.5 Linkages to Other Ecosystem Components

As noted in the Section 2, benthic macroinvertebrates are key components in the diet of many fish species, many of which are important as recreational or commercial fishery species (Seaman 1985). Older studies of feeding habits of young-of-the-year largemouth bass (*Micropterus salmoides*) by the Florida Fish and Wildlife Conservation Commission (FWC) (Figure 2–2) indicate that decapod crustaceans (i.e., crayfish and grass shrimp, *Palaemonetes paludosus*) and aquatic insects are important elements of the diet of these sport fish. In the lower river, young-of-the-year bass appear to take advantage of the estuarine taxa that occur in the benthic community (Figure 2–2), feeding on mysids along with insects and decapods.

Benthic macroinvertebrates are also an important food base for many wildlife species, some of which have special conservation status (e.g., threatened, endangered, etc.). In the upper St. Johns River basin (USJRB), water conservation areas are used by the endangered Everglades snail kite (*Rostrhamus sociabilis plumbeus*; Miller et al. 1996). These raptors feed almost exclusively on apple snails (*Pomacea paludosa*). Crayfish, grass shrimp, odonate larvae, amphipods, oligochaetes, and chironomids are consumed by a variety of wading birds, shore birds, and waterfowl throughout the St. Johns River basin (Chapter 13. Floodplain Wildlife). Invertebrates, such as crayfish and mussels, are also important food items for mammals, such as river otter (*Lutra canadensis*) and raccoon (*Procyon lotor*). Changes in the invertebrate food base would have ramifications for these higher trophic levels.

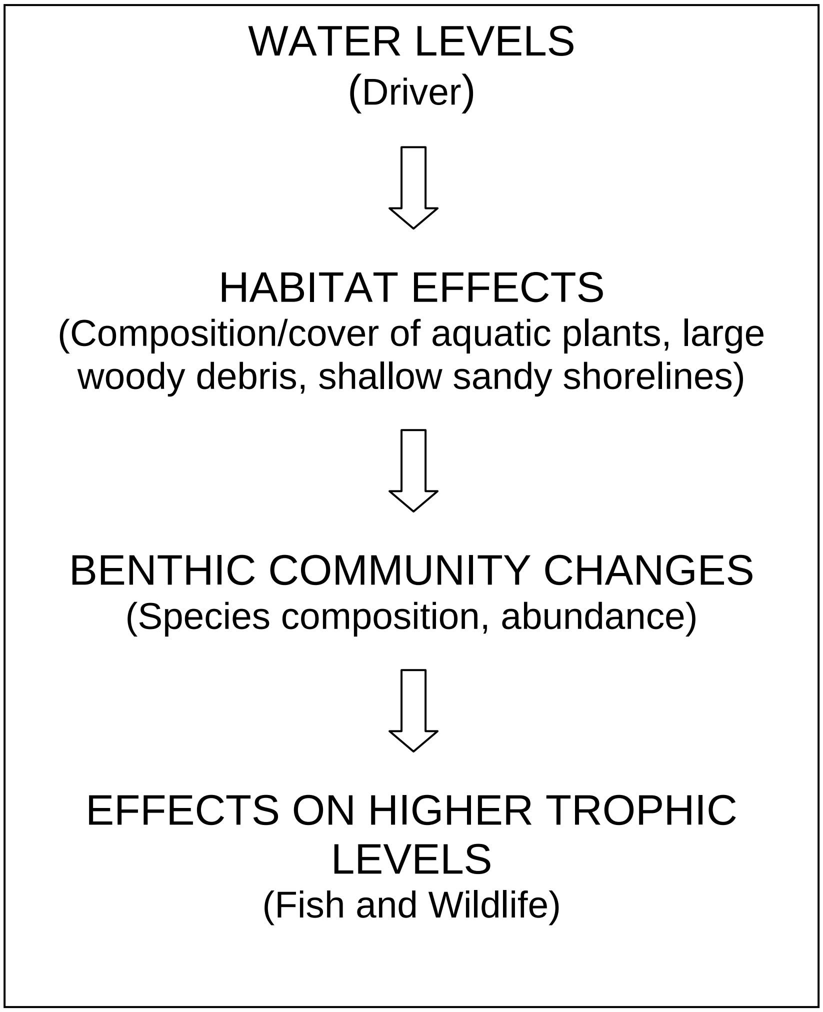


Figure 2–1. Flow chart showing linkages between changes in water level, habitat, effects on benthic communities and higher trophic levels.

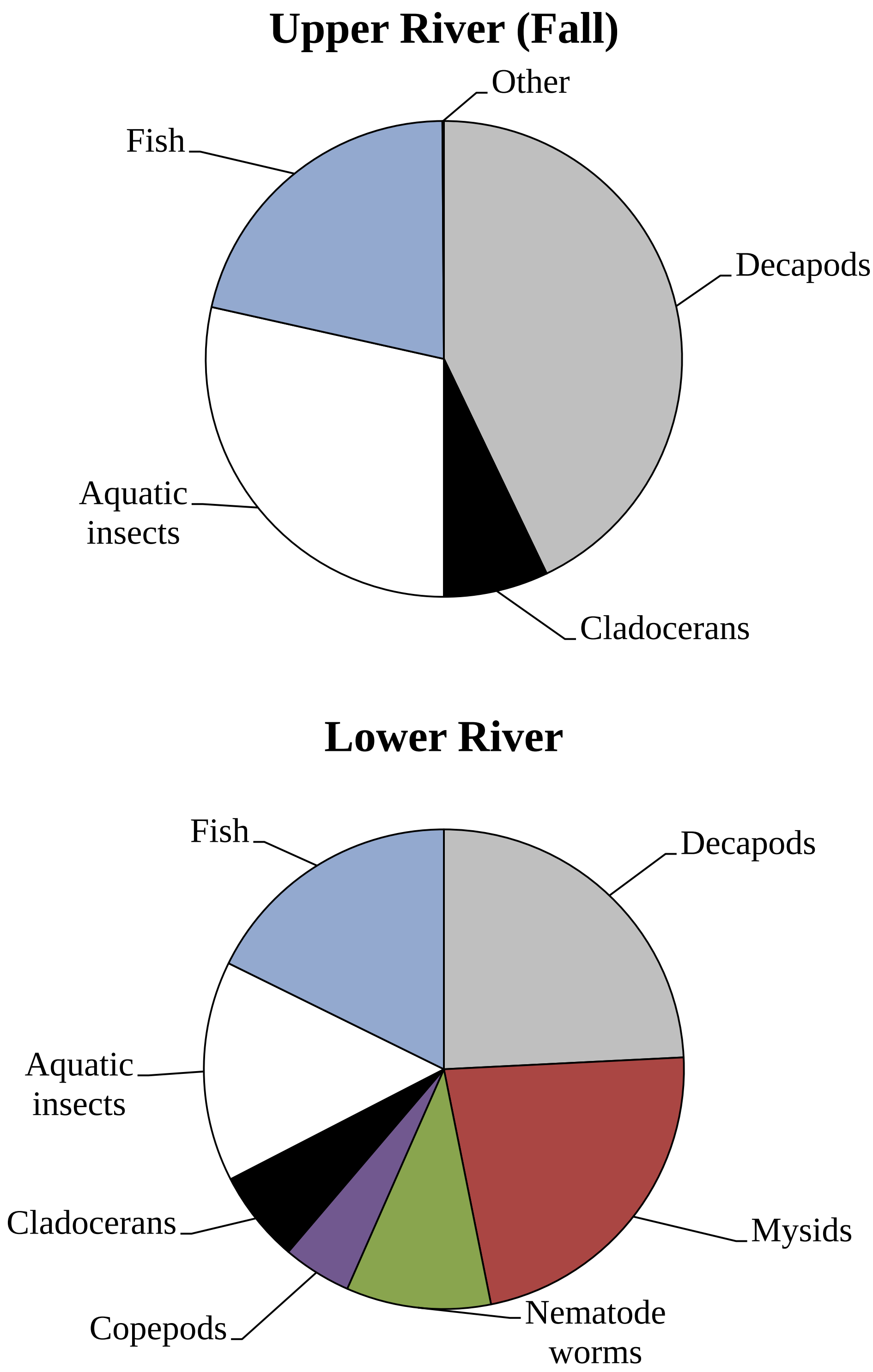


Figure 2–2. Feeding habits of young-of-the-year (juvenile) largemouth bass in the St. Johns River. Data from studies conducted from 1976 to 1981 by the Florida Game and Freshwater Fish Commission (Cox et al. 1982).

2.3 Potential Effects of Water Withdrawals on Macroinvertebrates

In freshwater reaches of the river, the two main physical effects of water withdrawals would be reduced water flows and water levels (Dewson et al. 2007). In the estuary, hydrologic change (i.e., changes in freshwater inflow) will primarily affect benthic communities by changing salinity patterns. This was discussed in general terms in Section 2.2.2.

Reduced flow would affect benthic communities because of reduced water velocity. Although there have been numerous studies quantitatively linking benthic community and population characteristics to water velocity (Allan 1995; Dewson et al. 2007), the complex nature of variation in water velocity across the river channel cross section makes it very difficult to model and predict what changes in flow and velocity would occur and how these might affect benthic communities. Lacking a quantitative predictive capability, we did not quantitatively assess the effects of changes in water velocity. This effect was considered in a general sense, using the existing scientific literature.

Changes in water level will primarily affect benthic communities in the shallow channel margins and in the floodplains. Water level can be accurately modeled and measured and is an important driver of benthic community characteristics. It will be the primary hydrologic driver examined for effects on benthic communities. Also, benthic communities in the deeper areas of the river channel are mainly dominated by pollution-tolerant taxa, resistant to many types of environmental stress (e.g., chironomid midges in the genus *Chironomus* and the oligochaete *Limnodrilus hoffmeisteri*). Benthic communities in the shallow channel margins are subjected to fewer water quality stresses (primarily hypoxic and anoxic conditions) due to better mixing and there is better quality invertebrate habitat in these areas (submersed and emergent aquatic vegetation and wood debris). Changes in water level will have the greatest effect in these shallow littoral areas and in the adjacent floodplain wetlands.

Macroinvertebrates could be selectively impacted by water level reduction via loss of specific habitats, for example, reduction of areal coverage by particular plant communities, or increases in anaerobic habitats with little or no flow. As was seen in the Kissimmee and Caloosahatchee rivers (Cummins and Merritt, unpubl. data), reduced flows (and water levels), especially during periods of warmer temperatures, can produce backwaters with no flow that become anaerobic. This condition results in very high mortality of macroinvertebrates, especially those taxa that are not very mobile. Such anaerobic or low oxygen environments are dominated by macroinvertebrates with adaptations to breathe air, for example, all aquatic hemipterans, many adult and pupal Coleoptera, and some larval dipterans. Some midges in the chironomid tribe Chironomini are known for their ability to tolerate low dissolved oxygen concentrations because they have respiratory pigments.

2.4 Conceptual Models of Chain of Causation and Discussion of Scientific Support

Based on the discussion above, Figure 2–3 and Figure 2–4 present conceptual models of the linkages between hydrology and benthic communities in freshwater (Figure 2–3) and estuarine (Figure 2–4) reaches of the river. Hydrology can affect benthic communities directly or indirectly. Direct effects include effects on physiology and behavior of individuals, changes in population characteristics (e.g., reproductive rate, growth rate), and effects on community attributes (i.e., species composition, changes in productivity). Indirect effects of hydrology act through changes in habitat or water quality characteristics (due to hydrologic change), which in turn affect benthic communities. The scientific basis for the linkages shown in these conceptual models was discussed in Section 2.2.

2.5 Key Effects, Indicators, and Potential Predictive Tools Linking Cause and Effect

Five components of hydrology (water flow and/or level) are ecologically significant: magnitude, frequency, duration, timing, and rate of change (Richter et al. 1996; Poff et al. 1997). Collectively these constitute the hydrologic regime (Poff et al. 1997). Timing will probably not be a major issue of concern on the St. Johns River because it is an unimpounded river system; periods of high and low flow will occur seasonally, as they do now, and would not be affected by water withdrawal. Rate of change could be an issue on the river floodplain, where more rapid decrease of water levels could affect benthic communities of floodplain wetlands. The three major characteristics to evaluate in terms of withdrawal effects are magnitude, frequency, and duration.

The hydrologic regime, as described above, influences all attributes of aquatic ecosystems (Poff et al. 1997) including biotic composition, community structure, and ecosystem function. In this analysis, we focused on structural measures of the benthic community and how these relate to hydrology (e.g., taxa richness versus water level or duration of inundation). Functional characteristics were not considered directly in this study (in terms of actual measurements of productivity, growth rates, etc., which were not made in this analysis). However, Merritt et al. (1996; 1999) and Cummins et al. (2008) have shown that certain ratios of structural benthic community measures can serve as surrogates of ecosystem function, allowing functional ecosystem attributes to be assessed indirectly.

We relied primarily on associative statistical models (e.g., correlation and regression) to link cause and effect, supplemented by multivariate analyses. While it is axiomatic that correlation does not prove causation, the mechanistic linkages between hydrology and various biotic attributes have been well established in numerous experimental and field studies in the scientific literature (Poff and Zimmerman 2010). These were used to support the analysis as needed. Correlation, coupled with a mechanistic basis for cause and effect, can provide strong evidence for causation and, where both lines of evidence exist, we used statistical correlation as a measure of the strength of the hydrologic effect. Thus, relationships between hydrology and ecology and predictions of the potential effects of hydrologic alteration on ecology can be made using statistical models. A similar logic applies to our analysis of salinity and ecological measures in the estuary of the river, as the association between salinity and ecological structure and function has been established mechanistically in the estuarine scientific literature. The putative causative links that support our use of correlative models are indicated in the conceptual models shown in Figure 2–3 and Figure 2–4.

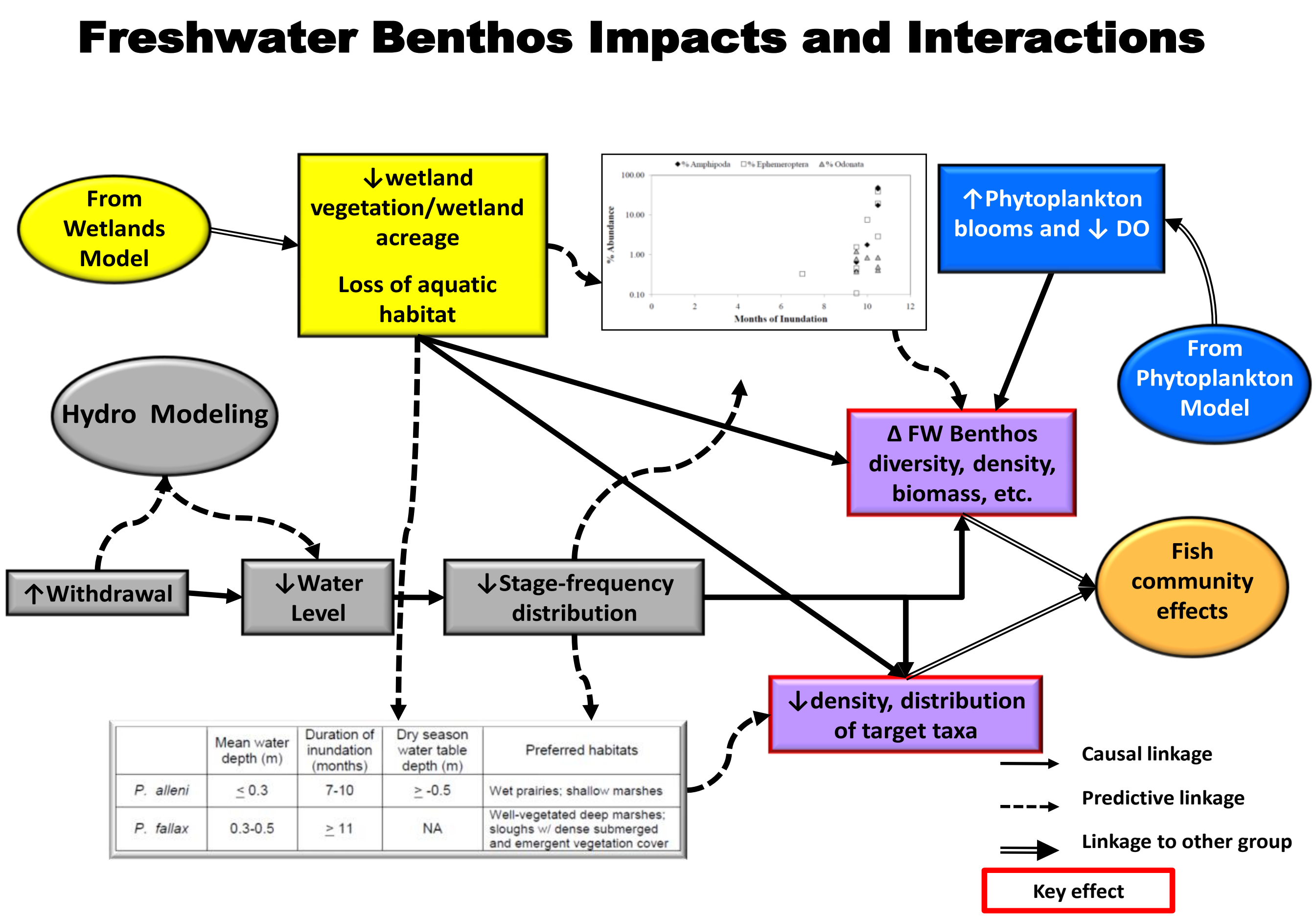


Figure 2–3. Conceptual model showing linkages among hydrology, freshwater benthic communities, and other ecological components.

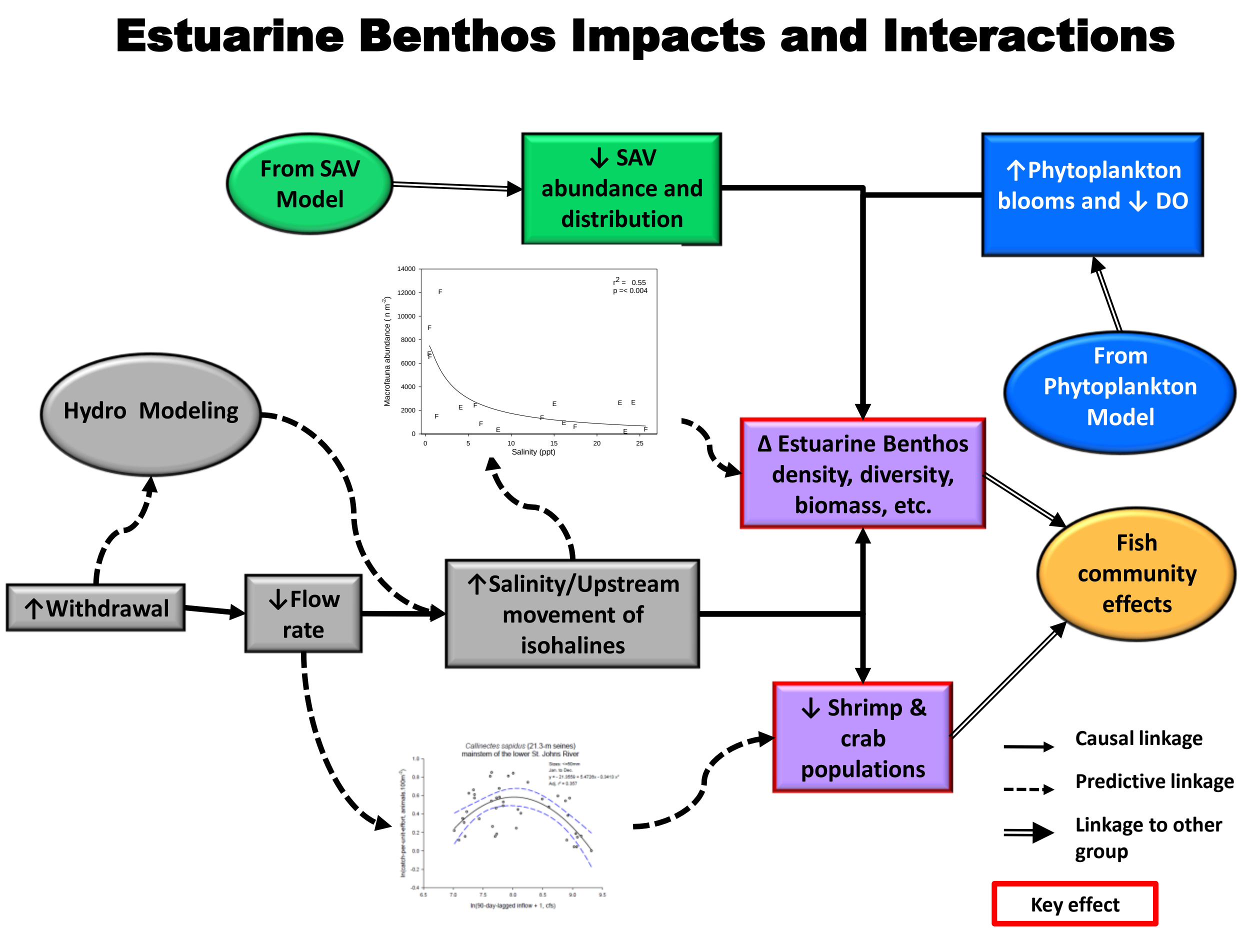


Figure 2–4. Conceptual model showing linkages among hydrology, estuarine benthic communities, and other ecological components.

3 METHODS

3.1 STUDY AREA DESCRIPTION

The area of study for this project is the entire main stem of the St. Johns River. No tributary streams were included in this study. Chapter 10, Wetlands Vegetation, provides a description of division of the St. Johns River main stem into segments, based on hydrology, water quality, channel geomorphology, and floodplain vegetation and morphology (Figure 3–1). From the river mouth, these segments are:

Segment 1, Mill Cove—The stretch of river adjacent to the river mouth; includes the commercial navigation channel, the Port of Jacksonville, Mayport Navy Base, and much of downtown Jacksonville. Salinity mesohaline to polyhaline.

Segment 2, Doctor's Lake—segment includes an embayment known as Doctor's Lake, Cedar/Ortega River is a major tributary; includes some of Jacksonville and most of Orange Park. Salinity oligohaline to mesohaline.

Segment 3, Deep Creek—segment includes inflow of Crescent Lake via Dunn's Creek and the inflow of the Ocklawaha River at the upper end. Major tributaries also include Black, Rice, and Deep creeks.Salinity is oligohaline to freshwater.

Segment 4, Lake George—segment includes Lake George; several springs contribute to this segment, including Croaker Hole Spring, Salt Spring, Silver Glen Spring, and Juniper Creek (fed by Juniper, Fern Hammock, Sweetwater, and Little Sweetwater springs).

Segment 5, Lake Woodruff—segment includes Lakes Dexter and Woodruff. Springs contributing to this reach include DeLeon Spring (via Spring Garden Lake), Alexander Spring (via Alexander Springs Creek), and Volusia Blue Spring.

Segment 6, Central Lakes—segment includes Lakes Monroe, Jesup, and Harney. The Wekiva River is a major tributary in the lower end of this segment, and the Econlockhatchee River enters at the upper end. A number of springs contribute to the Wekiva River, and Gemini and Green Springs flow into Lake Monroe. Includes highly urbanized areas of Sanford, Altamonte Springs, and some of Orlando.

Segment 7, SR 50—segment largely consists of the Puzzle Lake reach, where the river channel braids extensively (numerous side branches). Includes urbanized areas of Orlando.

Segment 8, Chain of Lakes—segment includes Lakes Poinsett, Winder, and Washington.

Segment 9, Blue Cypress Lake—This segment was not included in the study area because it is upstream of a weir located at the downstream end of Lake Washington, and therefore will not be affected by surface water withdrawals downstream.

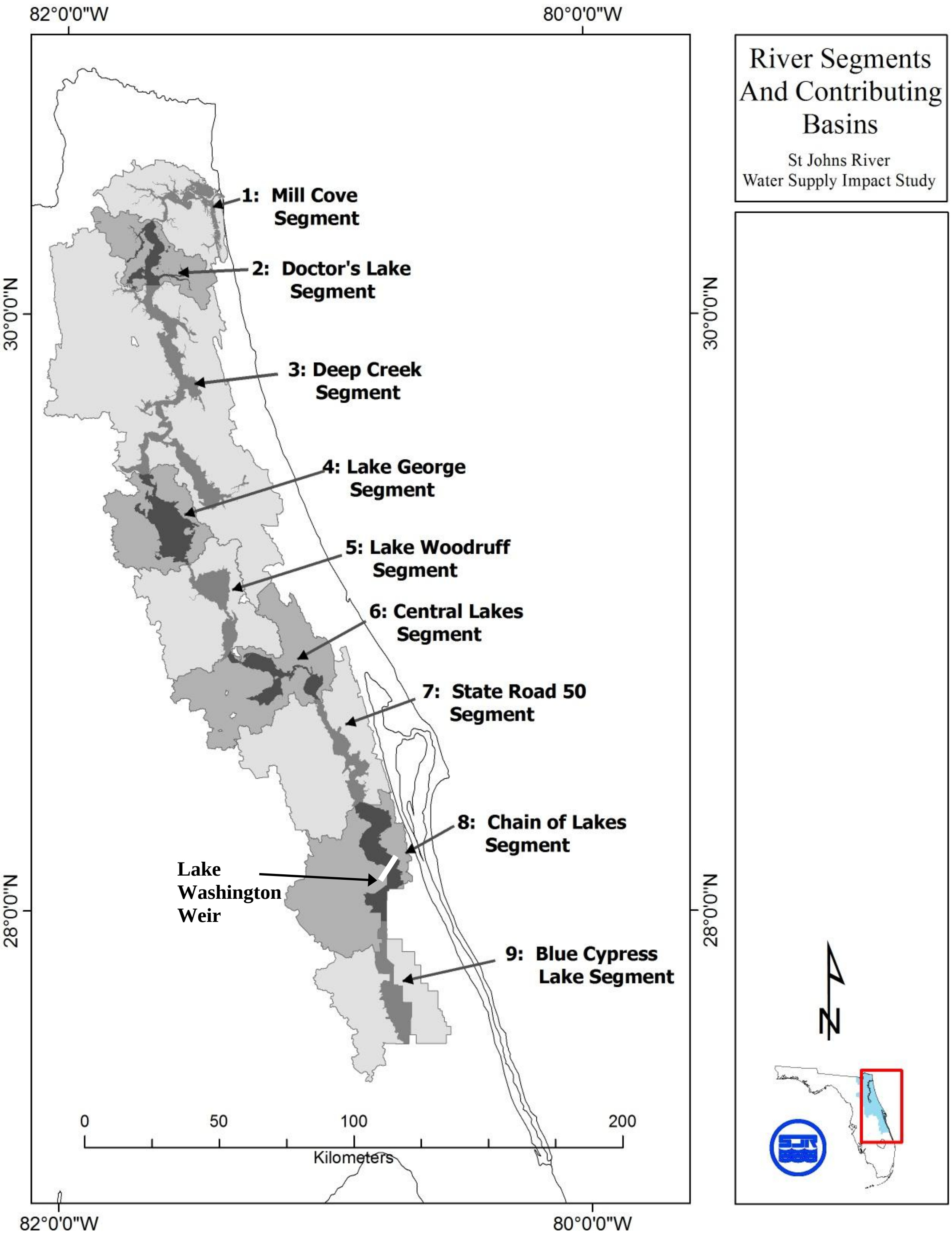


Figure 3–1. River segments in the St. Johns River with their associated wetland areas (shaded in dark grays) and drainage basins (shaded in light grays). The white line in segment 8 shows the approximate location of the Lake Washington weir, which is the southern terminus of the analyses of possible effects from water withdrawals.

3.2 Freshwater Benthic Communities—Approach and Methods

3.2.1 Approach to Assessment of Effects

To examine the effects of hydrology on freshwater benthic communities, we employed a multilevel, weight of evidence approach by looking at benthic communities at multiple levels of organization: at the community level (overall taxa richness, diversity, abundance and biomass), the population level (richness and relative abundance of various taxonomic levels, such as order, family, genus, and species), and species level by examining hydrology and habitat requirements of particular target taxa. We defined the target taxa as those taxa that, by their abundance, ecological roles, importance to other taxa, and conservation and/or economic value, warrant special focus for protection. A corollary to the selection of target taxa was that there needed to be enough existing data and knowledge about their habitat and life history requirements to enable us to evaluate the effects of hydrology and hydrologic alteration on their populations. No separate studies or data collection efforts were made on the target taxa in this analysis.

The target taxa chosen for freshwater areas of the river were crayfish (as a group) and the Florida apple snail, *Pomacea paludosa*. Crayfish were chosen due to their importance in the diets of many aquatic vertebrates; the two main taxa of crayfish in the upper and middle river are *Procambarus alleni* and *P. fallax* (Hobbs 1942; Franz and Franz 1990). The apple snail was chosen due to its importance in the diet of the Everglades snail kite and other vertebrate taxa. A number of studies have been conducted on the ecology and habitat requirements of crayfish and apple snails in the USJRB and in the Everglades (Section 4.1.4; Appendix II.E). We used the results of these to derive hydrologic criteria protective of these target taxa.

3.2.2 Short-Term Field Study of Effects of Hydrology on Macroinvertebrate Community Metrics and Linkages to Ecosystem Function

Because of limited data on freshwater benthic communities in the upper and middle reaches of the river (and the unsuitability of the existing data for use in this analysis), it was determined that a short-term field study of benthic communities at locations in the upper and middle St. Johns River would be needed to assess relationships between hydrology and benthic communities ,and to establish some type of baseline against which to compare future conditions. The Florida Department of Environmental Protection currently has in place two types of methodologies using benthic macroinvertebrates as tools for bioassessment in stream ecosystems: the Stream Condition Index (SCI) and the BioRecon procedure. The latter is meant to be used as a quick-and-dirty screening tool and was judged not rigorous enough for use in this study. The SCI is more rigorous and consists of sampling in multiple habitats and determination of an array of metrics indicating various biological community attributes (richness, composition, pollution tolerance, trophic measures, etc.). The SCI was specifically developed and calibrated in smaller, wadeable Florida streams (generally third order or less in size) and, therefore, was judged not suitable for use in a large river system such as the St. Johns River. We also evaluated suites of metrics that were shown to be indicative of environmental disturbance and stress in lotic systems throughout the world (Karr and Chu 1997, Table 7). Some of these could not be used because the taxa used to calculate them do not occur in the St. Johns River. The river also has a much more pronounced lentic character than many large river ecosystems in the continental United States. For these reasons, we employed a methodology used in the Kissimmee River drainage (Merritt et al. 1996; 1999), a river in south Florida having lentic qualities somewhat similar to those of the

St. Johns River, albeit due in part to impoundment of the river system. In designing the field study, we also followed the EPA's "Concepts and Approaches for the Bioassessment of Non-wadeable Streams and Rivers" (Flotemersch et al. 2006) and methodologies developed in non-wadeable rivers (Wessell et al. 2006).

Study Areas

The basic intent of the field study was to compare a more hydrologically dynamic ecosystem (Lake Poinsett) with a less dynamic one (Lake Monroe and the St. Johns River near Yankee Lake) under various water level conditions to determine if consistent, hydrology-related patterns could be seen in the benthic. This study was conducted in July (average flow and stage) and November (lower flow and stage) 2009 in three locations: Lake Poinsett, Lake Monroe, and the St. Johns River near Yankee Lake (site of a recently permitted new surface water withdrawal). Lake Poinsett is located in the USJRB (segment 8) and Lake Monroe and St. Johns River near Yankee Lake are in the middle St. Johns River basin (MSJRB) (segment 6). Figure 3–2 illustrates the general locations of the sampling areas at these three locations. Geographic coordinates of sampling sites at each reach location are listed in Appendix 11.B.

Macroinvertebrate Sampling and Processing

Macroinvertebrates were collected from four different vegetation types: emergent marsh vegetation (mixed taxa), bulrush (*Scirpus* sp.), spatterdock (*Nuphar advena*) and hydrilla (*Hydrilla verticillata*). The main habitats sampled were bulrush and spatterdock, and these were present and sampled at both lake locations. Sampling stations were established in each of these habitats at each reach location; and at each sampling effort (July and November), three replicate samples were collected from each station as indicated:

- Lake Monroe—three bulrush stations (three replicates at each in July and November); four spatterdock stations (three replicates at each in July and November); three hydrilla stations (three replicates at each in July and November); one emergent marsh vegetation station (three replicates in July and November)
- Lake Poinsett—three bulrush stations (three replicates at each in July and November); three spatterdock stations (three replicates at each in July and November); nine emergent marsh vegetation stations (one replicate at each in July); six emergent marsh vegetation stations (one replicate at each in November); one river channel benthic ponar station (three replicates in November)
- St. Johns River near Yankee Lake—one spatterdock station (three replicates in July and November); one emergent marsh vegetation station (three replicates in July and November)

At Lake Poinsett, we were able to sample a variety of emergent marsh habitats in the floodplain in July, and collected one sample at nine different emergent marsh vegetation sampling stations. In November, lower water levels restricted access to the floodplain in Lake Poinsett, and we sampled six emergent marsh vegetation stations (one sample at each) closer to the lake shoreline. We also collected three petite ponar grab samples from the river channel downstream of Lake Poinsett. We collected six samples in St. Johns River near Yankee Lake in both July and

November: 27 samples in July and November in Lake Poinsett, and 33 samples in July and November in Lake Monroe. More samples were collected in Lake Monroe because this was the only location that also had a submersed vegetation zone of hydrilla. In total, we collected 66 samples in July and November 2009 from the three study locations.

We collected macroinvertebrates with a D-frame dip net (500 µm mesh) within a 1 m^2 PVC frame (randomly located) and 30 sec of sampling effort with the net. A similar technique was used in the Kissimmee and Caloosahatchee rivers (Merritt et al. 1996, 2002). Macroinvertebrate specimens in each sample were washed through a 500 µm mesh sieve, labeled, and preserved in 1-quart jars in 90% to 100% isopropanol. Samples were processed and identified at Michigan State University's aquatic entomology lab. In the lab, larger benthic samples were split into two equal subsamples with an Aquatic Research Instruments Folsom Plankton Splitter™. Only one subsample was processed to reduce sorting time. Macroinvertebrates were then picked out of detritus in the samples using forceps under a dissecting microscope. Invertebrates were identified to the lowest practicable taxon using Pennak (1989), Thorp and Covich (2001) and Merritt et al. (2008) and enumerated by taxon. Noninsect invertebrates were typically identified to family and genus, and insects were identified to genus and species (except the Chironomidae, which were left at family). Invertebrates also were assigned to a functional feeding group (FFG) as described by Merritt et al. (2008). All specimens were measured to the nearest mm to allow for biomass estimates using published length-dry mass regression data from Benke et al. (1999) and a computer software program, INVERTCAL, previously developed and used by Merritt et al. (2002).

Data Analysis

Data collected were summarized as various metrics (Merritt et al. 1996, 1999, 2002) indicating taxonomic composition and abundance, trophic measures (i.e., FFG), life span, and pollution sensitivity and tolerance (Table 3–1). Ratios of macroinvertebrate FFGs were used as surrogates for selected functional attributes of the aquatic ecosystem. Plots were constructed of mean values ($n = 3$) with standard errors to compare differences in these metrics between sampling dates, between sampling locations, and between habitats within locations. Analyses were conducted using Microsoft™ EXCEL, MiniTab™, and Sigma Plot™.

For assessment of hydrologic and water quality effects on benthic communities, stage, and water quality from existing monitoring stations were used (USGS for stage; SJRWMD stations for water quality). Stage at St. Johns River near Yankee Lake was assumed very similar to that at Lake Monroe; the gauge used to measure stage at Lake Monroe is at the lake outlet and 4.2 km upstream of St. Johns River near Yankee Lake. The river gradient here is very flat, so this seems a reasonable assumption. Changes in stage in Lake Monroe due to water withdrawal are likewise assumed the same as at St. Johns River near Yankee Lake due to their proximity.

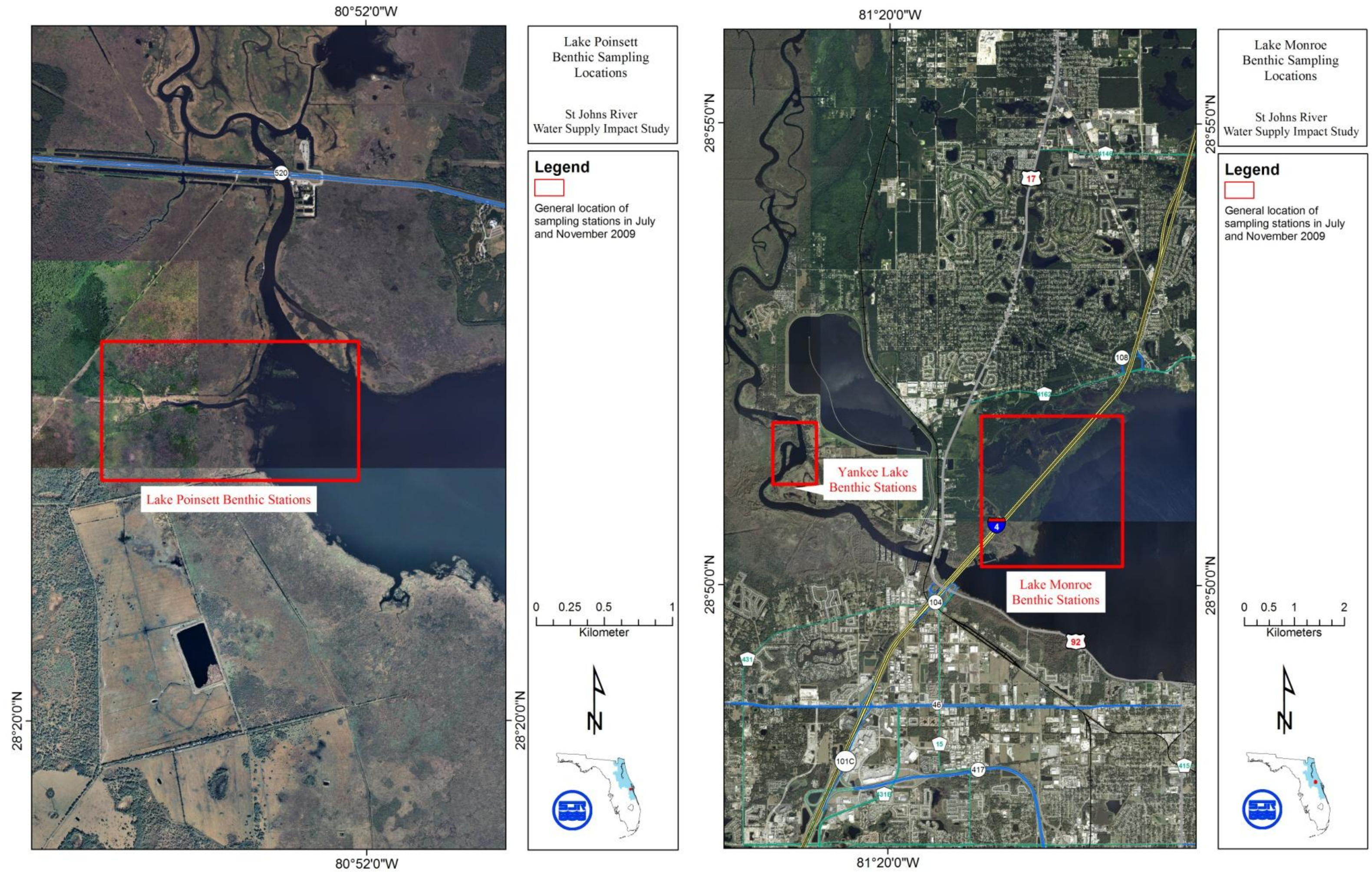


Figure 3–2. Maps of Lake Poinsett and Lake Monroe regions, showing general areas where sampling stations were located for the short-term field study of freshwater benthic communities in 2009.

Table 3–1. Summary of benthic macroinvertebrate metrics calculated from the raw data in the 2009 field study.

Metric	What It Describes	Source
Total number taxa	Taxonomic richness and composition	Kashian and Burton 2000; Apfelbeck 2001; Helgen and Gernes 2001
Number of individuals	Relative abundance	Kashian and Burton 2000; Apfelbeck 2001
Odonata (%)	Taxonomic composition; relative abundance of long-lived taxon; pollution sensitive taxon	Burton et al. 1999; Kashian and Burton 2000
Trichoptera (%)	Taxonomic composition; pollution sensitive taxon	Kashian and Burton 2000; Helgen and Gernes 2001
Chironomidae (%)	Taxonomic composition; relative abundance of short life cycle taxon; pollution tolerant taxon	Kashian and Burton 2000; Apfelbeck 2001; Helgen and Gernes 2001
Diptera (%)	Taxonomic composition; relative abundance of short life cycle taxon; pollution tolerant taxon	Kashian and Burton 2000; Helgen and Gernes 2001
Gastropoda (%)	Taxonomic composition; relative abundance of long-lived taxon	Burton et al. 1999; Kashian and Burton 2000
Amphipoda (%)	Taxonomic composition; relative abundance of true aquatic taxon	Burton et al. 1999; Kashian and Burton 2000
Ephemeroptera (%)	Taxonomic composition; relative abundance of true aquatic taxon; pollution sensitive taxon	Helgen and Gernes 2001
Evenness	Taxonomic composition; relative abundance	Burton et al. 1999; Kashian and Burton 2000
Shannon diversity	Taxonomic composition; relative abundance	Burton et al. 1999; Kashian and Burton 2000; Helgen and Gernes 2001
Simpson’s diversity	Taxonomic composition	Burton et al. 1999; Kashian and Burton 2000

3.3 Estuarine Benthic Communities—Approach and Methods

3.3.1 Approach to Assessment of Effects

We used existing data to assess the potential effects of salinity changes on estuarine infaunal benthic communities. As in the freshwater areas, we examined the relationship of salinity and estuarine benthic macroinvertebrates at multiple levels: community, population, and individual species. More conventional uni- and multivariate statistics were used for the analysis of basic community characteristics (taxa richness, abundance), because a specialized estuarine bioassessment sampling protocol with individualized metrics has not been developed in Florida.

Two target epifaunal taxa were also analyzed, using data and analyses supplied by the Fisheries Independent Monitoring program, a monitoring program conducted by the FWC Fish and Wildlife Research Institute. This program collects juvenile finfish and shellfish species from the mainstem and tributary creek habitats of the lower river and estuary with multiple gear types. The target taxa were blue crab (*Callinectes sapidus*), and penaeid shrimp, principally the white shrimp (*Litopenaeus setiferus*). Both of these decapod crustaceans support important commercial and recreational fisheries in the St. Johns River; and both are important ecologically as food for higher trophic levels and as key predators in estuarine benthic communities. The existing literature on life history and environmental requirements of these two target taxa was also reviewed and included in this analysis.

3.3.2 Existing Data Sets Used for Analysis

Two data sets were used for the statistical analysis of infaunal benthic communities and populations; long-term monitoring data collected 1973 to 1996 in the estuary by the Northeast District of the Florida Department of Environmental Protection (FDEP) and a benthic dataset collected by the Environmental Monitoring and Assessment Program–Estuaries (EMAP-E) in 2000. Figure 3–3 illustrates the locations of the FDEP and EMAP study sites in the estuary used for this analysis.

Ross (1990) summarized the standard procedures for FDEP benthic sampling. In brief, the FDEP samples were collected with a petite ponar grab (sampling area of 15.24 by 15.24 cm) and sieved with a U.S. Standard No. 30 sieve (595 µm). In some cases, samples were composites of several grabs whereas other samples were the contents of a single grab. EMAP-E samples were collected with a Young-modified Van Veen grab (440 cm^2 sampling area) and processed with a 500- µm sieve (Strobel and Heitmuller 2001). At least three replicate grab samples were collected at each EMAP location.

For the target epifaunal taxa, the FWC analyzed data collected in the lower St. Johns River and estuary by the their Fisheries Independent Monitoring program. The program used a stratified random method for selecting sampling sites each year and employed three gear types (21.3 and 183-m seine and 6.1-m otter trawl) to target juvenile species of finfish and shellfish. Details of the sampling design and field methods are described (MacDonald et al. 2009) and are not repeated here.

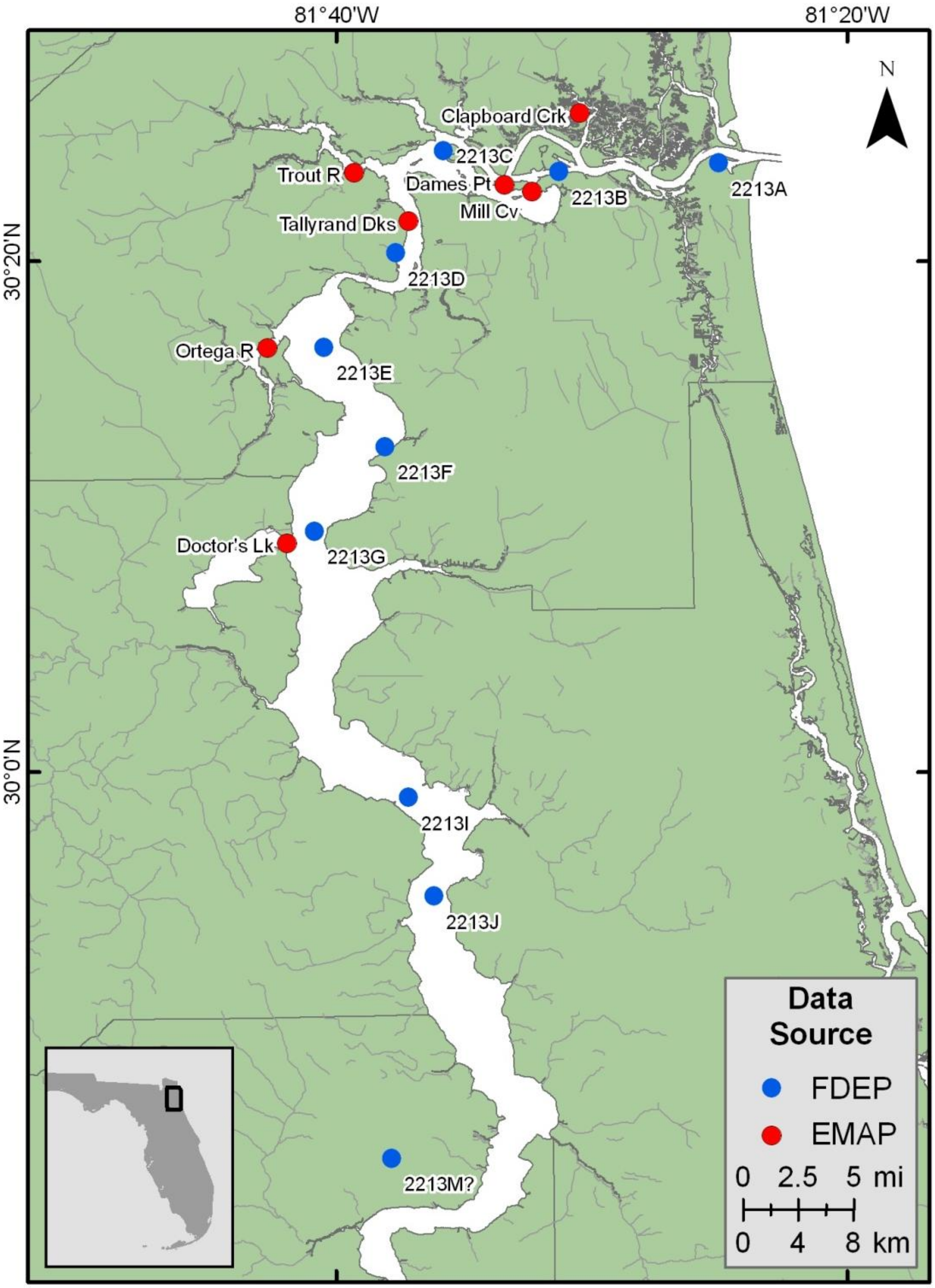


Figure 3–3. Map of the lower St. Johns River and estuary showing locations of sampling sites where data were collected. FDEP = Florida Department of Environmental Protection stations, EMAP = Environmental Monitoring and Assessment Program stations.

3.3.3 Description of Statistical Approach for Benthic Community Analysis

Macrofaunal Community Structure

Nonmetric multidimensional scaling (MDS) was used to compare macrofaunal community structure among station-substrate combinations. The spacing of stations in an MDS plot is related to the degree of community similarity among stations. Differences and similarities among communities were highlighted using cluster analysis. Significant clusters were determined using the SIMPROF permutation (Clarke 1993). Macrofaunal communities were analyzed using different taxonomic levels—species, genus, family and higher taxa (predominantly using classifications of phyla and class). MDS and cluster analysis were performed using a Bray-Curtis similarity matrix on $\log_n(x + 1)$ transformed data in Primer software (Clarke and Warwick, 2001), where x is a measure of taxon abundance.

Comparing Macrofauna and Salinity

We compared mean abundance of all macrofaunal taxa and the most abundant taxa with mean salinity values for each station. We used a nonlinear model to examine the relationships between mean salinity and macrofaunal abundance. The model was previously used in Texas and Florida estuaries in the Gulf of Mexico (Montagna et al. 2002b; Montagna et al. 2008b). The assumption behind the model is that there is an optimal range for salinity, and abundance declines prior to and after meeting this maximum value. The shape of this curve can be predicted with a three-parameter, log-normal model:

$$Y = ae^{\left[-0.5\left(\frac{\ln\left(\frac{X}{X_c}\right)}{b}\right)^2\right]}$$

The model was used to characterize the nonlinear relationship between a biological characteristic (*Y*) and salinity (*X*). The three parameters characterize different attributes of the curve, where *a* is the maximum value, *b* is the skewness or rate of change of the response as a function of salinity, and *Xc* the location of the peak response value on the salinity axis. The model was fit to data using the Regression Wizard in SigmaPlot™, which uses the Marquardt-Levenberg algorithm to find coefficients (parameters) of the independent variables that give the best fit between the equation and the data (Systat 2006). We then constructed a spreadsheet version of this nonlinear model to evaluate effects of changes in salinity on total macroinvertebrate abundance in the estuary.

We used the BIO-ENV procedure to correlate between multivariate macrofauna community structure and physical water quality variables. The BIO-ENV procedure is a multivariate method that matches biotic (i.e., community structure) with environmental variables (Clarke and Warwick 2001) by calculating weighted Spearman rank correlations (ρ_w) between sample ordinations from all of the environmental variables and an ordination of biotic variables (Clarke and Ainsworth 1993). Correlations are then compared to determine the best match. The BIO-ENV procedure uses different numbers of abiotic sample variables in calculating correlations to investigate the different levels of environmental complexity. For this study, we compared the species abundance MDS ordination with all hydrographic variables (temperature, salinity, dissolved oxygen, and pH). The significance of relationships were tested using RELATE, a nonparametric form of the Mantel test. The BIO-ENV and RELATE procedures were calculated with Primer software (Clarke and Warwick 2001).

Comparing Macrofauna and Sediment Grain Size

Sediment grain size is often an important factor that can influence estuarine macrofaunal species distributions and abundances (Rhoads 1974; Mannino and Montagna 1997; Kennish et al. 2004). Sediment grain size data collected by SJRWMD from more than 300 sampling stations within the estuary were available for analysis. However, few of the sediment samples were taken within 1 km of the macrofauna sampling stations and very few samples were taken within 40 km (25 mi) of the estuary mouth—the location of nine macrofauna stations. Grain size data included the proportion of clay, sand, and silt in sediments. We approximated spatial trends in grain size throughout the estuary by interpolating the grain size values by ordinary kriging using a spherical semivariogram model as the interpolation method calculated in ArcGIS 9.3.1.

3.4 Ranking and Selection of Withdrawal Scenarios

Table 3–2 presents the various scenarios modeled by the HSPF hydrologic and EFDC hydrodynamic models (see Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results) and their ranks based on predicted hydrologic effects in freshwater reaches of the upper and middle river. Five withdrawal scenarios were selected for evaluation of effects on freshwater benthic communities, based on the ranking and rationale in Table 3–2. These included three test scenarios that model specific conditions, but which are realistically unlikely, and two potential realistic future scenarios:

- Full1995NN—this test scenario was evaluated in segments 4 through 8.
- Half1995NN—this test scenario was evaluated in segments 4 through 8.
- Full1995PN—this test scenario was evaluated in segments 4 through 8.
- Full2030PN—this potential future scenario was evaluated only in the upper basin (segments 7 and 8), where sea level rise has no appreciable effect but the USJRB projects will.
- Full2030PS—this scenario represents a potential future condition and was evaluated only in segments 4 through 6, where sea level rise may have material effects.
- Base1995NN—this was the base scenario to which each of the scenarios above were compared to evaluate withdrawal effects.

Table 3–2. Ranking of modeled withdrawal scenarios for evaluation of effects of water level changes on freshwater benthic communities in the upper and middle St. Johns River main stem (Segments 4 through 8).

Scenario	Land Use	USJRB Project	Water Withdrawal (mgd)	Sea Level	Predicted Δ mean stage (m) Poinsett	Predicted Δ mean stage (m) Monroe	Rank	Rationale
Base1995NN	1995	No	0	1995	0	0	Baseline	Base condition
Full1995NN	1995	No	155	1995	-0.05	-0.03	1	Worst case withdrawal
Half1995NN	1995	No	77.5	1995	-0.03	-0.02	2	Lesser withdrawal
Full1995PN	1995	Yes	155	1995	+0.02	-0.03	3	Addition of USJRB projects
Half1995PN	1995	Yes	77.5	1995	+0.04	Not calculated	NR	Effect less than Full1995PN
Full2030PN	2030	Yes	155	1995	+0.06	Not calculated	4	Projected 2030 land use plus projects
Half2030PN	2030	Yes	77.5	1995	+0.08	Not calculated	NR	Effect less than Full2030PN
Full1995PS	1995	Yes	155	2030	Not calculated	Not calculated	NR	Doesn't add any new results
Full2030PS	2030	Yes	155	2030	Not calculated	+0.12	5	2030 land use, projects and SLR
Half2030PS	2030	Yes	77.5	2030	Not calculated	Not calculated	NR	Effect less than Full2030PS
FwOR1995NN	1995	No	262	1995	Not calculated	Not calculated	NR	OR withdrawal not applicable
FwOR1995PN	1995	Yes	262	1995	Not calculated	Not calculated	NR	OR withdrawal not applicable
FwOR1995PS	1995	Yes	262	2030	Not calculated	Not calculated	NR	OR withdrawal not applicable
FwOR2030PN	2030	Yes	262	1995	Not calculated	Not calculated	NR	OR withdrawal not applicable
FwOR2030PS	2030	Yes	262	2030	Not calculated	Not calculated	NR	OR withdrawal not applicable

* See Chapter 6, River Hydrodynamics Results, for discussion on scenarios.
Note:
USJRB = Upper St. Johns River basin
MSJRB = Middle St. Johns River basin
NR = Not ranked
SLR = Sea level rise
OR = Ocklawaha River

For purposes of assessing predicted changes in salinity, the Submersed Aquatic Vegetation Working Group evaluated and ranked the scenarios based on increase in the 7-day maximum salinity (see Table 4-3 in Chapter 9. Submersed Aquatic Vegetation). We used their ranking and evaluated two scenarios (FwOR1995NN ranked #2 and FwOR2030PS ranked #7), reflecting a worst case situation and a potential future condition:

- FwOR1995NN—this test scenario involved a withdrawal of 262 mgd from the St. Johns River and Ocklawaha River, 1995 land use, without completion of the upper basin projects and with no sea level rise.
- FwOR2030PS—this potential future scenario involved a withdrawal of 262 mgd from the St. Johns River and Ocklawaha River), with 2030 land use, with the upper basin projects complete and functional and with the historic rate of sea level rise projected out to 2030.

Both of these scenarios were compared to the base scenario (Base1995NN; no new surface water withdrawals, 1995 land use, no upper basin projects, and no sea level rise).

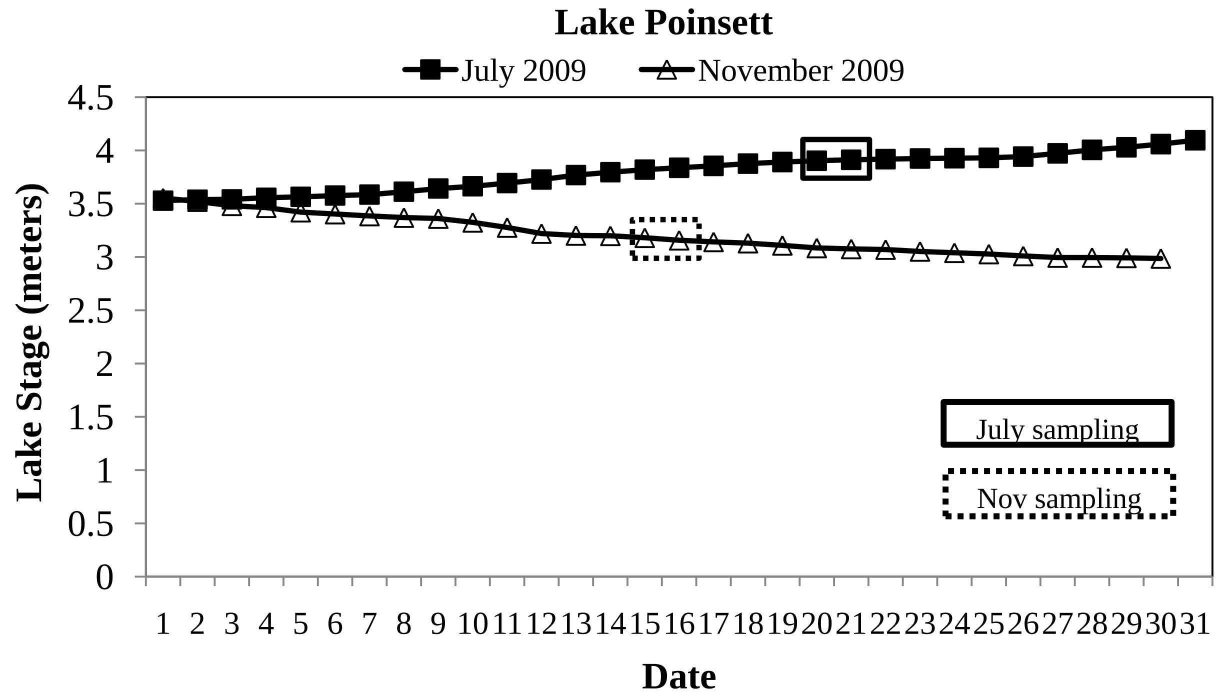
4 RESULTS AND DISCUSSION

4.1 FRESHWATER MACROINVERTEBRATE COMMUNITIES

4.1.1 PHYSICAL AND CHEMICAL CONDITIONS AT THE FIELD STUDY LOCATIONS

Stage in Lakes Poinsett and Monroe at the time benthic community sampling was conducted in July and November 2009 is shown in Figure 4–1. Water levels declined 0.75 m between July and November in Lake Poinsett, whereas they increased slightly in Lake Monroe. Stage at St. Johns River near Yankee Lake was assumed to be very similar to that at Lake Monroe; the gauge used to measure Monroe lake stage is at the lake's outlet and 4.2 km upstream of St. Johns River near Yankee Lake. The river gradient here is very flat so this seems a reasonable assumption. Changes in stage in Lake Monroe due to water withdrawal are likewise assumed to be nearly the same as at St. Johns River near Yankee Lake due to their proximity.

A summary of water quality in both lakes is shown in Table 4–1, as this could also affect the differences seen in benthic communities at the three sampling locations. Water quality at St. Johns River near Yankee Lake is assumed to be similar to that at Lake Monroe, because this site is just downstream of the lake outlet. Water quality in both lakes is quite similar (Table 4–1). Conductivities are somewhat high in both lakes due to inflow of saline groundwater, with higher conductivities in Lake Monroe. Mean dissolved oxygen concentrations are somewhat higher in Lake Monroe, but both lakes periodically exhibit very hypoxic conditions (Table 4–1). Alkalinity is slightly higher in Lake Poinsett. Secchi depth, turbidity, and total suspended solids are similar in both lakes. Color is slightly higher in Lake Poinsett. Nutrient (total N and P) and organic carbon levels are similar in both lakes. Chlorophyll-a is slightly higher in Lake Monroe, most likely due to higher color in Lake Poinsett depressing phytoplankton populations (see Chapter 8. Plankton).



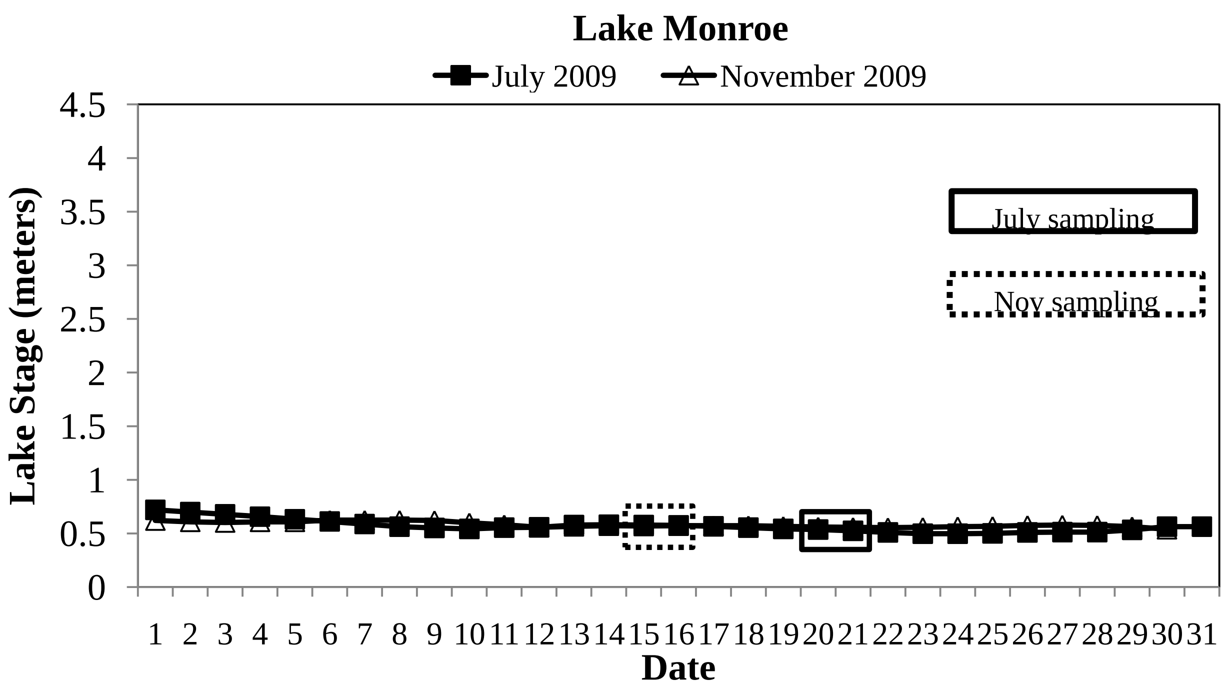


Figure 4–1. Stage in Lakes Poinsett and Monroe during the months benthic sampling was conducted. Boxes show when sampling was conducted each month.

Table 4–1. Summary of water quality in Lakes Poinsett and Monroe. Data from St. Johns River Water Management District monitoring, covering periods 1979 to 2010 (Lake Poinsett) and 1991 to 2010 (Lake Monroe).

Analyte	Lake	Mean	Standard Deviation	Minimum	Maximum
Conductivity (µmhos cm^{-1})	Poinsett	820	447	192	1888
	Monroe	1067.3	472.3	343.5	2230.0
Dissolved oxygen (mg L^{-1})	Poinsett	6.3	2.2	0.8	16.6
	Monroe	7.2	2.0	0.9	11.9
pH (units)	Poinsett	7.4	0.5	5.9	9.93
	Monroe	7.7	0.66	6.3	9.4
Alkalinity (mg L^{-1} as $CaCO_3$)	Poinsett	62	13	29	96
	Monroe	53.5	13.8	2.0	89.2
Secchi depth (m)	Poinsett	0.7	0.3	0.2	1.9
	Monroe	0.5	0.2	0.2	1.2
Turbidity (NTU)	Poinsett	5.8	7.6	0.3	64.3
	Monroe	5.6	4.1	0.85	21.6
Total suspended solids (mg L^{-1})	Poinsett	10	14	<0.1	117
	Monroe	11.0	9.6	0	68.5
Color (PCU)	Poinsett	192.5	114.1	37.5	700.0
	Monroe	172.1	115.5	30.0	500.0
Total N (mg L^{-1})	Poinsett	1.9	0.5	0.1	4.4
	Monroe	1.7	0.6	0.4	7.5
Total P (mg L^{-1})	Poinsett	0.1	0.1	<0.1	0.5
	Monroe	0.1	<0.1	<0.1	0.3
Total organic carbon (mg L^{-1})	Poinsett	26.8	4.7	17.8	44.0
	Monroe	23.5	4.8	13.4	43.2
Chlorophyll-a (µg L^{-1})	Poinsett	10.8	13.5	0	136.2
	Monroe	19.8	23.6	0	165.8

Note:

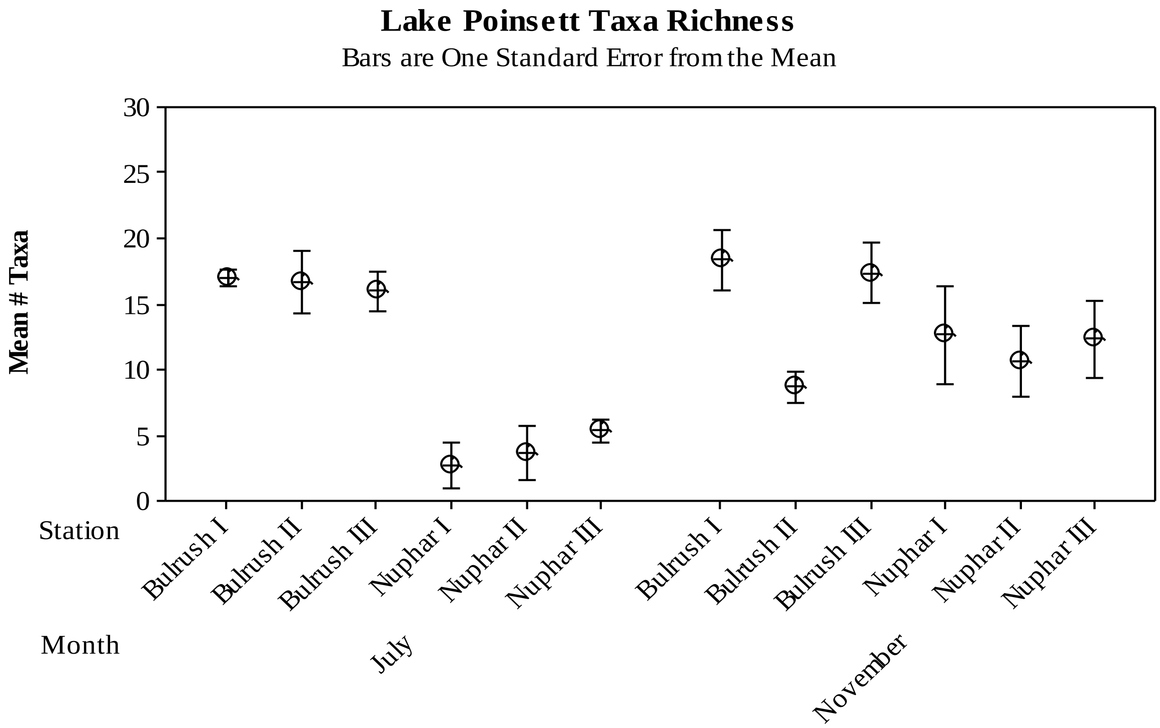
µmhos	=	micromhos
µg	=	micrograms
NTU	=	Nephelometric Turbidity Units
PCU	=	Platinum Cobalt Units
N	=	Nitrogen
P	=	Phosphorus

4.1.2 Description of Benthic Community Metrics and Ecosystem Functional Surrogate Measures

Mean taxa richness (Figure 4–2 and Figure 4–3) was generally highest in bulrush and spatterdock habitat in Lake Monroe. Taxa richness was lower overall in both habitats in Lake Poinsett and in spatterdock habitat at St. Johns River near Yankee Lake. The figure shows standard error about the mean, and nonoverlap of the error bars suggests a statistically significant difference. Overall mean taxa richness varied considerably among all habitats at the three locations (Figure 4–2 and Figure 4–3); the only clear significant difference appeared to be lower taxa richness in spatterdock versus bulrush habitat in Lake Poinsett in July. This is probably more likely due to some factor such as increased susceptibility to fish predation in spatterdock versus bulrush due to higher stem density in the latter habitat. July and November data were not combined due to the likelihood of seasonal differences between summer and fall samples (respectively). Higher taxa richness in spatterdock habitat in November versus July in Lake Poinsett and St. Johns River near Yankee Lake may reflect seasonal changes in the benthic community and could not be attributed to hydrology.

Graphical summaries of the other macroinvertebrate metrics are presented in Appendices 11.C and 11.D. Mean relative abundance of longer lived taxa (e.g., Odonata) and true aquatic taxa (e.g., Ephemeroptera) were low overall, but were both generally higher in Lake Monroe and St. Johns River near Yankee Lake. More sampling sites in Lake Monroe and St. Johns River near Yankee Lake exhibited a mean relative abundance of mayflies (Ephemeroptera) of > 5%, whereas most of the sampling sites at Lake Poinsett were < 5% Ephemeroptera. This difference could reflect the more variable hydrologic regime of Lake Poinsett, but could also be due to water quality effects such as periodic low DO events in Lake Poinsett (Chapter 7. Biogeochemistry). Between the two habitats within each lake, there were no clear differences in relative abundance of odonates or mayflies. Other taxa (Amphipoda, Mollusca, Diptera, Hemiptera, Oligochaeta, Other Crustacea, and Trichoptera) exhibited varying patterns of relative abundance among the sampling sites at the three locations, with no clear trends (Appendix 11.C).

The dominant FFGs expressed as biomass at all three locations (Lake Poinsett, Lake Monroe, and St. Johns River near Yankee Lake) were shredders, collectors, and scrapers. Shredder-herbivores accounted for 80% of the total shredder taxa biomass. Collectors and scrapers dominated the relative abundance (Appendix 11.D). By season and location, shredders, collectors, and scrapers dominated by both biomass and relative abundance (Appendix 11.D).



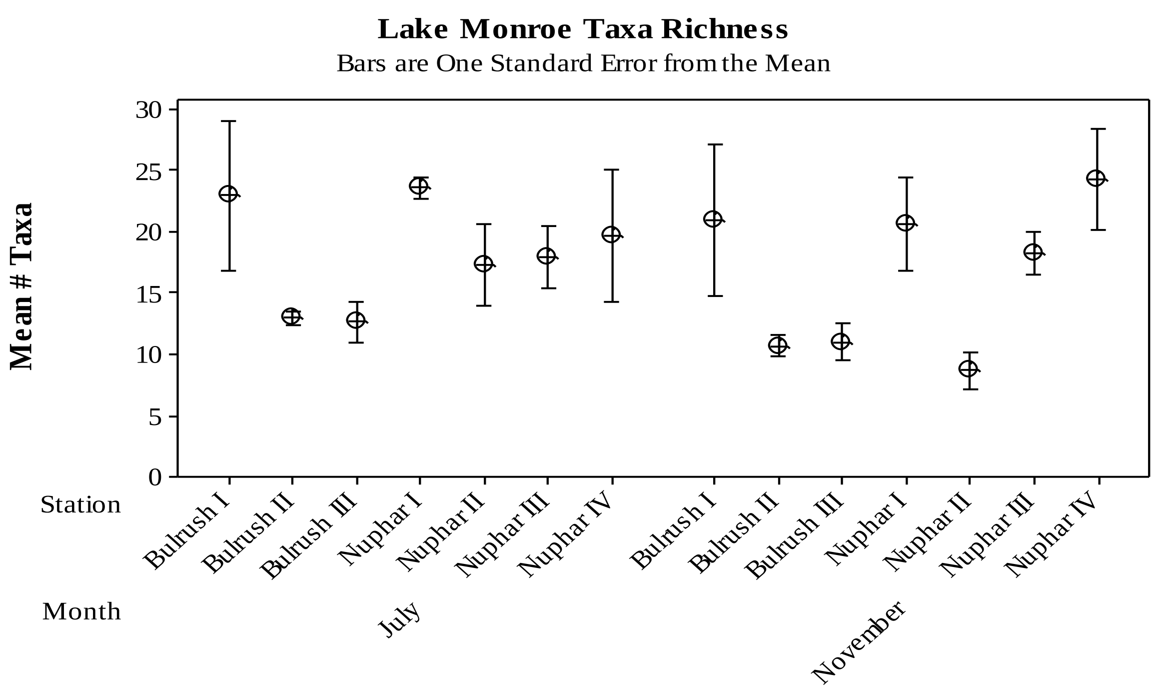


Figure 4–2. Mean taxa richness in the benthic communities in bulrush and spatterdock (*Nuphar*) habitat in Lakes Poinsett and Monroe. Bars are one standard error from the mean; n = 3.

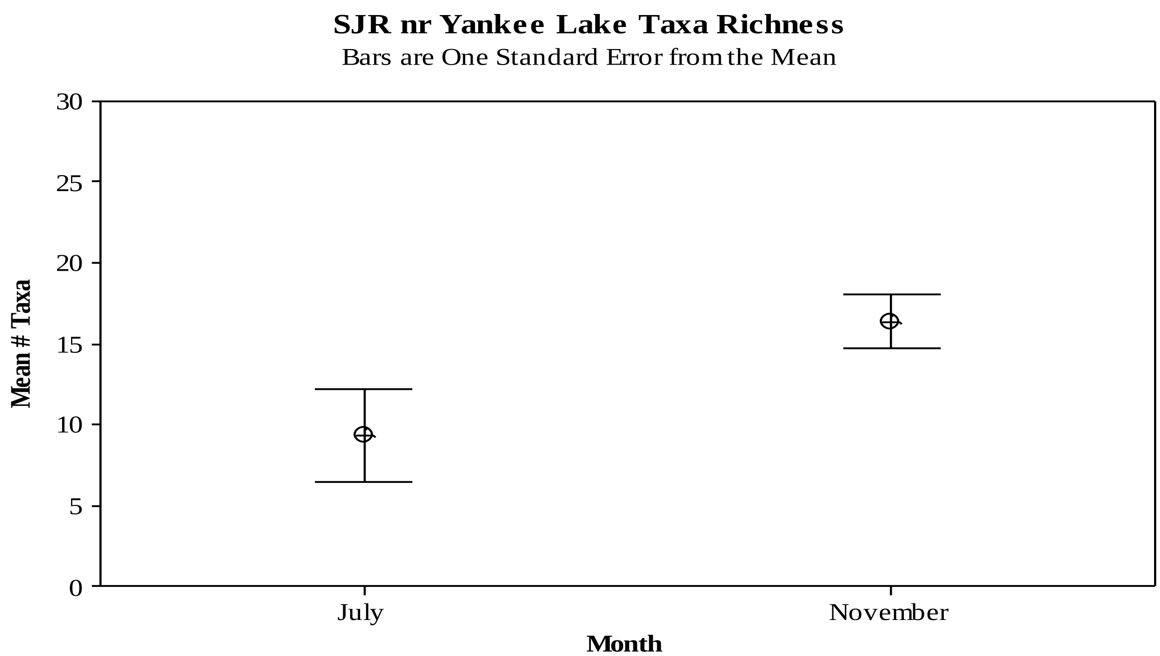


Figure 4–3. Mean taxa richness in the benthic communities in spatterdock at St. Johns River (SJR) near Yankee Lake. Bars are one standard error from the mean, n = 3.

Relative abundance of various FFGs in bulrush and spatterdock habitat exhibited considerable variability, making it difficult to discern trends and judge effects of hydrology. Appendix 11.D has figures summarizing all of the FFG comparisons. Relative abundance of filtering collectors appeared to be significantly lower in spatterdock versus bulrush habitat in Lake Poinsett in July, and significantly higher relative abundance in July versus November at St. Johns River near Yankee Lake. Scrapers appeared to be significantly higher in relative abundance in spatterdock habitat at St. Johns River near Yankee Lake, but exhibited no clear differences between bulrush and spatterdock habitat in the two lakes. Neither of these trends appeared to be driven specifically by hydrology. Higher scraper relative abundance in spatterdock is likely due to more abundant growth of periphyton on the stems of the plants, similar to what Merritt et al. (1999) saw in spatterdock habitat in the Kissimmee River. Shredders appeared to show significantly higher relative abundance in spatterdock habitat in Lake Poinsett in July. This difference is likely due to the abundance of the aquatic caterpillar *Bellura* sp. at that location. In November, relative abundance of shredders exhibited no clear trends in Lake Poinsett, Lake Monroe, or St. Johns River near Yankee Lake.

Sampling of benthic communities in emergent marsh habitat in the Lake Poinsett floodplain in July 2009 provides some evidence that hydrology exerts an influence on benthic community structure. The following habitats were sampled (ordered roughly from longer to shorter periods of inundation):

- AW—Alligator weed (*Alternanthera philoxeroides*); flooded 88% of the time (about 10 to 11 months each year)

- Sesban—*Sesbania* sp.; flooded 88% of the time (about 10 to 11 months each year)
- AW/grass—Mixed alligator weed and unidentified grass; flooded 88% of the time (about 10 to 11 months each year)
- SM—Shallow marsh (mixed *Spartina bakeri*, *Echinochloa* sp., other grasses, and sedges); flooded for 77% to 84% of the time (9 to 10 months each year)
- Para/cypress—Para grass (*Urochloa* sp.) and bald cypress (*Taxodium distichum*) knees; flooded 83% of the time (about 10 months each year)
- WP—Wet prairie (*Spartina bakeri*, other); flooded 50% of the time (roughly up to 6 months each year)
- Phragmites—Pure stand of *P. communis*; flooded for 55% of the time (about 7 months each year)

Total taxa richness was highest in shallow marsh areas flooded for 9 to 10 months in a typical year (Figure 4–4). Relative abundance of longer lived taxa, such as odonates, was higher in marsh habitats inundated for longer durations (Figure 4–5). Fauna indicative of more permanent aquatic habitat (e.g., Ephemeroptera, Amphipoda) also had higher relative abundance in habitats inundated for longer periods (Figure 4–5), while short generation taxa (Diptera) or those with good dispersal capability as adults (e.g., Coleoptera and Diptera) had higher relative abundance in habitats with shorter duration of inundation (Figure 4–5). These trends can be attributed in part to the duration of inundation in these habitats; the literature also supports this conclusion (Williams 1996; Wissinger 1999; Lillie 2003).

Merritt et al. (1996; 1999) and Cummins et al. (2008) showed that ratios of various macroinvertebrate life history attributes (FFGs and other life habits) can be used as surrogates to evaluate ecosystem functional characteristics (Table 4–2). For example, use of FFGs relates directly to food resources and ecosystem trophic characteristics (Merritt et al. 1999; Cummins et al. 2008). Comparisons of all the metrics listed in Table 4–2 in bulrush and spatterdock habitat at the three study locations are shown in Appendices 11.C and 11.D.

The macroinvertebrate surrogate ratio for the ecosystem production:respiration (P:R) ratio is shown in Figure 4–6, Figure 4–7, and Figure 4–8. For this ratio, a value >0.75 is a suggested threshold indicating an autotrophic ecosystem (sustained by autochthonous primary production); and < 0.75 indicates a heterotrophic ecosystem (dependent upon detrital organic matter input). Cummins et al. (2008) indicate that this threshold was determined based on comparisons with direct measures of production and respiration in streams (closed recirculating benthic chambers) and corresponds to the more typical P:R threshold = 1, which is usually used to indicate the break between autotrophy and heterotrophy based on actual measurement of production and respiration rates. The P:R ratio can be considered a fundamental ecosystem attribute indicating overall ecosystem function. Macroinvertebrates can provide a long-term assessment of this attribute by virtue of their linkages to the two foundations of river food webs: live plant production and plant detritus. FFG ratio data from the hydrilla and emergent marsh habitats are not shown in this report. The data from both of these habitats did not provide much insight as collected (a different sampling approach appeared to be needed in these habitats).

In sampling sites in bulrush and spatterdock habitat, macroinvertebrate relative abundance at most of the sample stations on Lake Monroe indicate a heterotrophic system (index < 0.75), while most of the stations in Lake Poinsett and St. Johns River near Yankee Lake indicate an autotrophic state (Figure 4–6 and Figure 4–7). In terms of biomass across all habitats, all locations appear to be autotrophic to strongly autotrophic (Figure 4–8), whereas count data indicate a slightly heterotrophic to moderately autotrophic state (Figure 4–8).

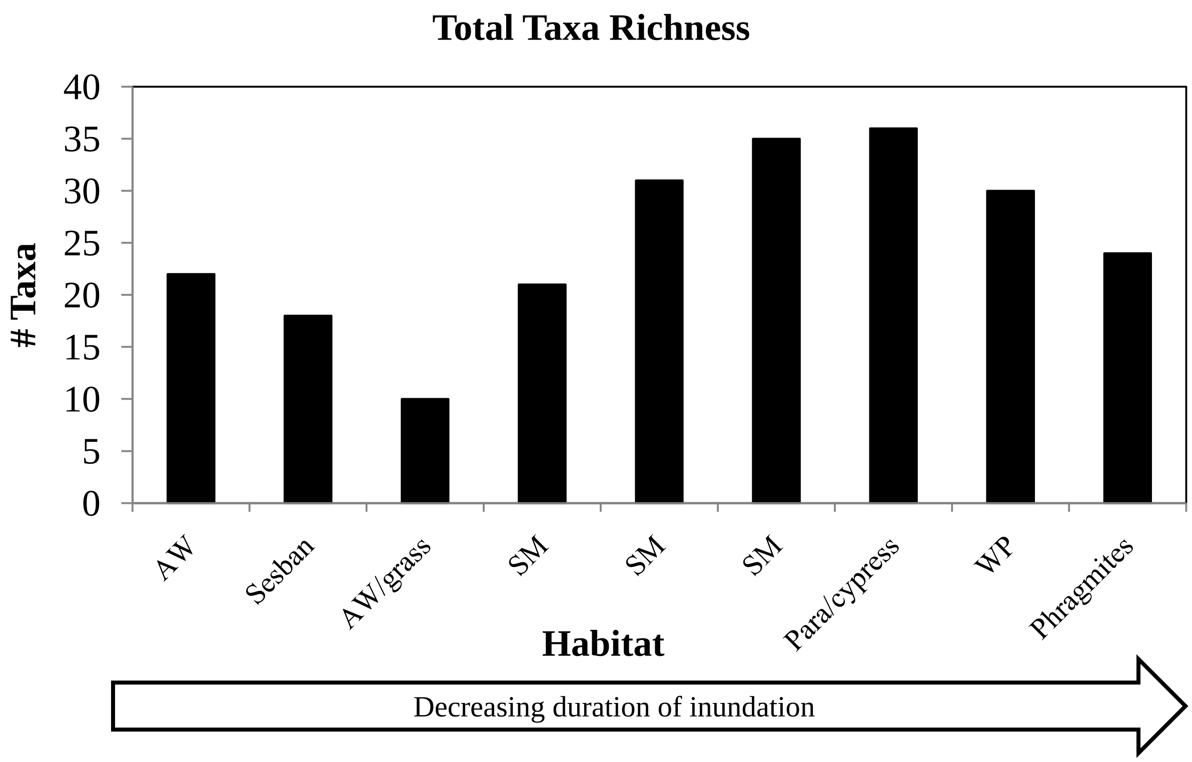


Figure 4–4. Total taxa richness of benthic macroinvertebrates in various emergent marsh habitats in Lake Poinsett in July 2009. AW = Alligator weed (*Alternanthera philoxeroides*); Sesban = *Sesbania* sp.; grass = unidentified grass; SM = shallow marsh; Para/cypress = mixed Para grass and bald cypress (*Taxodium distichum*) tree knees; WP = wet prairie; Phragmites = *P. communis*. In each habitat, n is a single dip net sample.

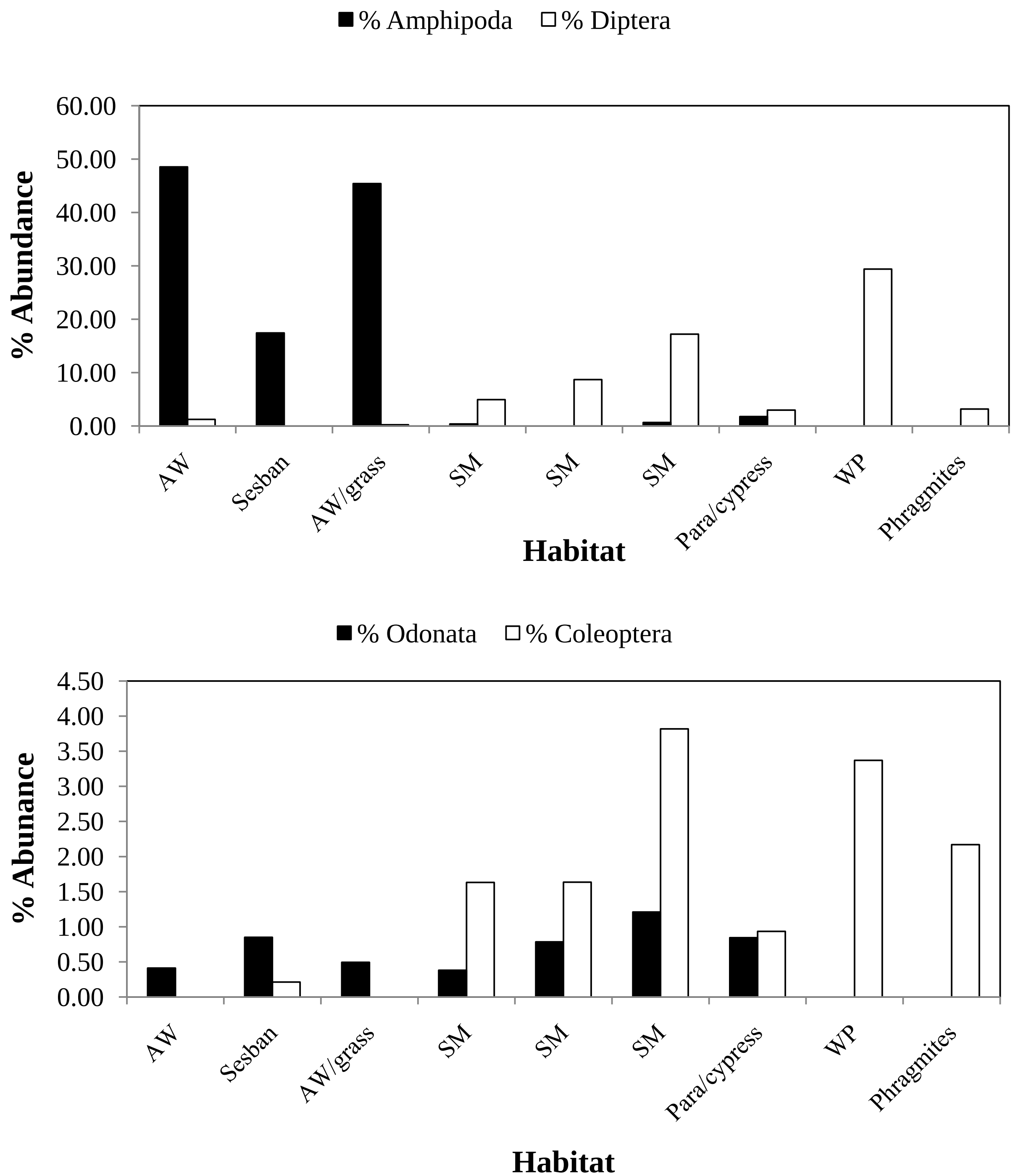


Figure 4–5. Relative abundance (%) of various taxa groups in the marsh habitats sampled in Lake Poinsett in July 2009. AW = Alligator weed (*Alternanthera philoxeroides*); Sesban = *Sesbania* sp.; grass = unidentified grass; SM = shallow marsh; Para/cypress = mixed Para grass and bald cypress (*Taxodium distichum*) tree knees; WP = wet prairie; Phragmites = *P. communis*. In each habitat, n is a single dip net sample.

Table 4–2. Benthic macroinvertebrate surrogate ratios used in this study to evaluate selected ecosystem functional attributes

Ecosystem Attribute	Description of Ecosystem Functional Measure	FFG or Other Life Habit Surrogate Ratio
Production:Respiration (P:R) Ratio	*Autotrophy/Heterotrophy Index*	Scrapers + Live Shredders + Piercers: Detrital Shredders + Total Collectors
	Ratio of gross primary production to total community respiration	Threshold = 0.75
BCPOM:BFPOM Ratio	*Shredder–Riparian Index*	Shredders: Total Collectors
	Ratio of benthic CPOM to benthic FPOM; indicates availability of coarse detritus for invertebrate food	Fall/Winter Threshold = 0.50 Spring/Summer Threshold = 0.25
SFPOM:BFPOM Ratio	*Suspended Load Index*	Filtering Collectors:Gathering Collectors
	Ratio of suspended FPOM (in transport in water column) to benthic FPOM; may indicate excessive load (enrichment) of organic matter to stream	Threshold = 0.50
Habitat Stability I	*Substrate Stability Index I*	Scrapers + Filtering Collectors:Shredders + Gathering Collectors
	Amount of hard habitat on the bottom (wood, vegetation, etc.); indicates availability of stable habitat for attachment, grazing, etc.	Threshold = 0.50
Top-Down Control	*Predator/Prey Index*	Predators:Total all other FFG
	ratio of predators to all other FFG; indicates whether predation may be a significant controlling factor	Threshold = 0.15
Habitat Stability II	*Substrate Stability Index II*	Clingers + Climbers:Burrowers +Sprawlers + Swimmers
	Ratio of organisms requiring stable substrate to live on versus those that burrow in sediments or are more motile; indicates availability of stable habitat	Threshold = 0.60
Benthic Fish Food Availability	*Food Availability Index*	Sprawlers: Clingers + Climbers + Burrowers + Swimmers
	Ratio of taxa vulnerable to predation by wading birds and benthic-feeding fish versus those less vulnerable; indicates food availability for wading birds and fish	Threshold = 0.60

Note:
FFG = Functional Feeding Group
BCPOM = Benthic coarse particulate organic matter
BFPOM = Benthic fine particulate organic matter
SFPOM = Suspended fine particulate organic matter

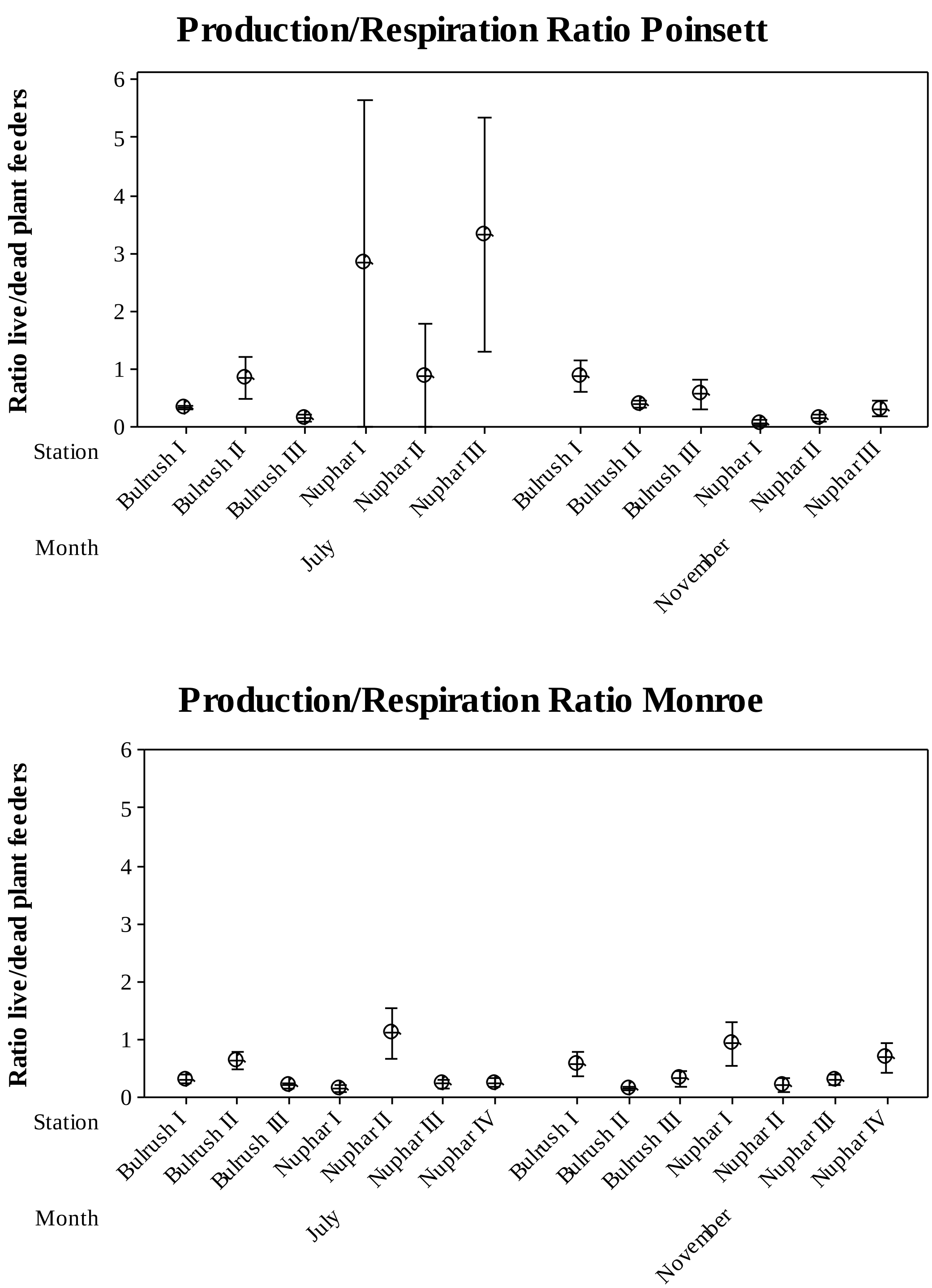


Figure 4–6. Production:respiration surrogate index calculated as ratio of live plant feeders (live plant shredders + scrapers) to detrital plant feeders (detrital shredders + total collectors) for Lakes Poinsett and Monroe. Bars are one standard error from the mean, n = 3 at each station. Poinsett = Lake Poinsett, Monroe = Lake Monroe.

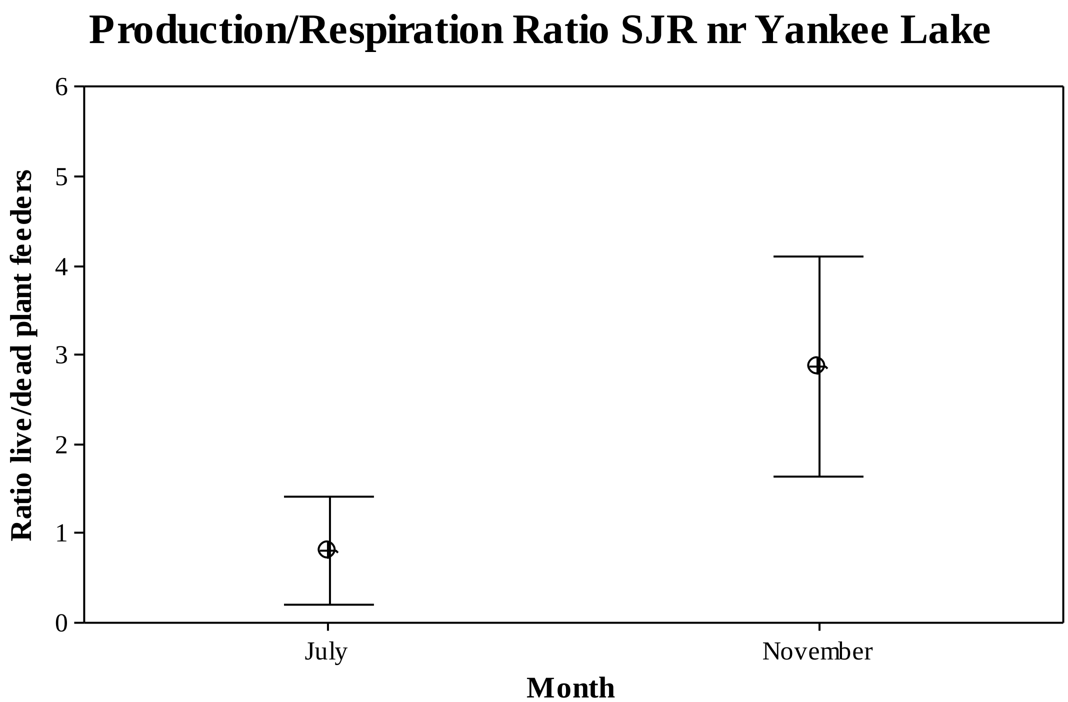


Figure 4–7. Production:respiration surrogate index calculated as ratio of live plant feeders (live plant shredders + scrapers) to detrital plant feeders (detrital shredders + total collectors) for the St. Johns River near Yankee Lake. Bars are one standard error from the mean, n = 3 at each station, SJR = St. Johns River.

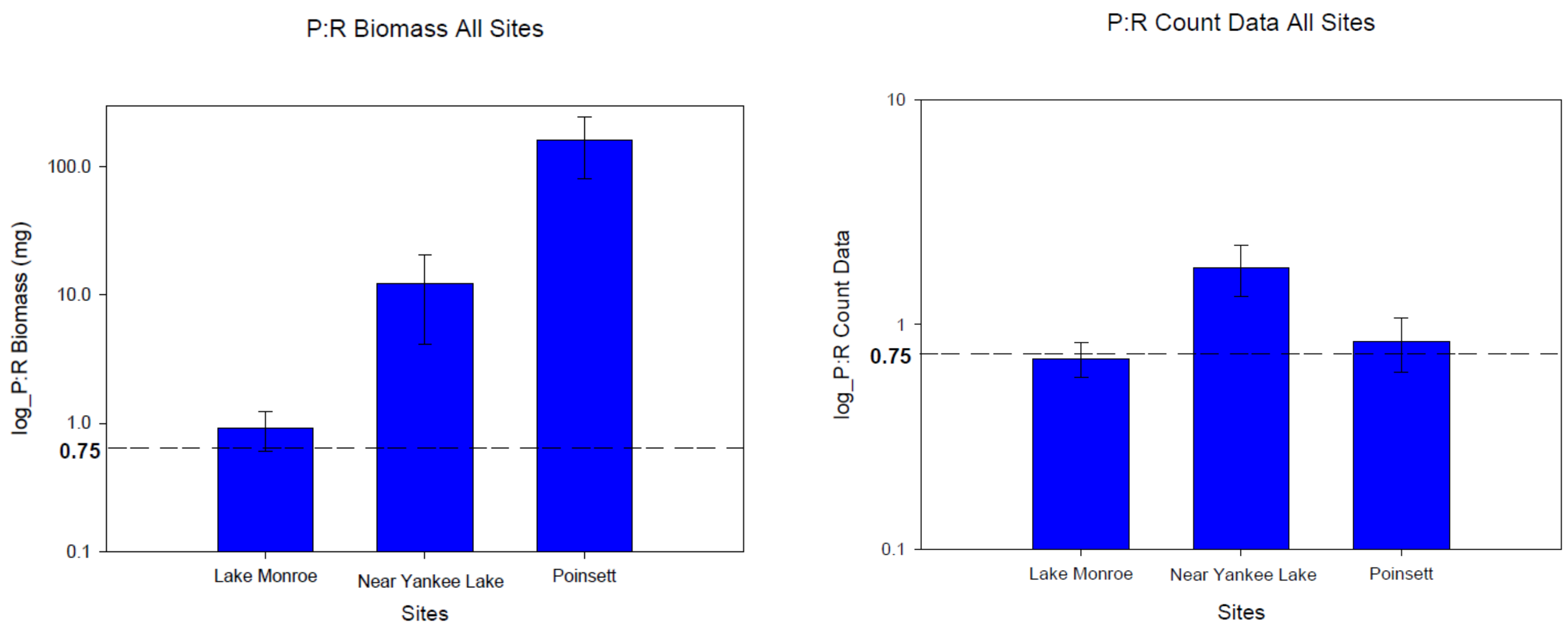


Figure 4–8. Production:respiration ratio (P:R) surrogate index calculated as ratio of live plant feeders (live plant shredders + scrapers) to detrital plant feeders (detrital shredders + total collectors) by biomass and relative abundance (count data) for all habitat types combined (bulrush, spatterdock, hydrilla, and emergent marsh vegetation) at the three sampling locations. Poinsett = Lake Poinsett.

4.1.3 Direct and Indirect Effects of Hydrology on Freshwater Benthic Communities

Table 4–3 presents summary statistics of modeled water level elevations for the base scenario (Base1995NN) versus other scenarios (Full1995NN, Half1995NN, Full1995PN, Full2030PN and Full2030PS) for Lakes Poinsett and Monroe. At Lake Poinsett, mean and median water levels decreased under both test scenarios (Full1995NN and Half1995NN), although declines were reduced in the "half" scenario. Water levels decreased across nearly the entire hydrologic regime under Full1995NN. With the 1995 land use scenario, addition of the upper basin projects (Full1995PN) resulted in an increase in mean water level, but median water level still declined, and water level decreases continue to be exhibited at full withdrawal under some portions of the hydrologic regime even with the upper basin projects. With the 2030 land use scenario, along with the completed upper basin projects (Full2030PN), water levels are predicted to increase across most of the hydrologic regime in Lake Poinsett, with a slight decrease occurring in the median. At Lake Monroe, mean and median water levels decreased by 2 to 3 cm in three of the scenarios (Full1995NN, Half1995NN and Full1995PN). Similar to Lake Poinsett, water levels were basically permanently reduced under the full withdrawal only scenario (Full1995NN), with lesser reductions seen under the Half1995NN case. Addition of the upper basin projects (Full1995PN) did not appreciably reduce predicted mean and median reductions in water levels in Lake Monroe, but increased the projected maximum water level (Table 4–3). The projects also do not appear to affect the reductions in lower water levels (the 25th percentile and minimum), which are similar in both the Full1995NN and Full1995PN scenarios. Predicted water levels are increased across the entire hydrologic regime with the 2030 land use and the addition of sea level rise in Lake Monroe (Full2030PS).

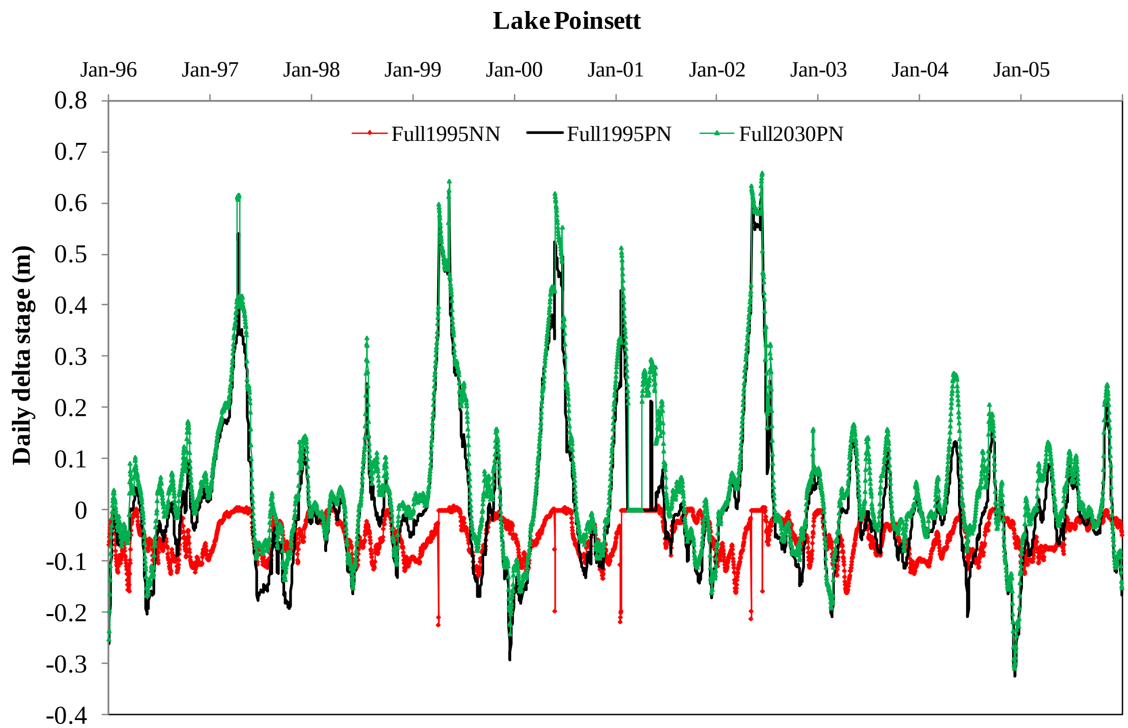
Figure 4–9 presents a time series of modeled delta stage in Lakes Poinsett and Monroe under three modeled scenarios (Full1995NN, Full1995PN, and Full2030PS).Variation in lake stage was higher in Lake Poinsett than in Lake Monroe. Water level decreases of 7 to 10 cm appear to occur regularly in Lake Poinsett under the Full1995NN and Full1995PN scenarios, and still occur under the Full2030PN scenario, although less frequently. Under some conditions, reductions of 10 cm or more last for periods up to several weeks. Addition of the upper basin projects results in increased magnitude and frequency of higher flows, but does not appear to affect low flows as much in Lake Poinsett. Water level reductions (25th percentile and minimum) were generally higher at Lake Poinsett, versus Lake Monroe, under both 1995 and 2030 withdrawal and land use scenarios (all). Water level reductions of up to 10 cm do occur in Lake Monroe, but with a lesser frequency than at Lake Poinsett.

On average, the predicted water level reductions in Lake Poinsett would appear to have minimal or no effect on benthic communities. Higher reductions (7 to 10 cm) that may occur at lower lake stages would have a more pronounced effect on benthic communities, particularly those in fringing littoral and floodplain wetland habitats, due to loss of aquatic habitat, but this would be mitigated by the increases in water levels due to the projects and changing land use. The benthic data available (from the 2009 field study and the historical data) do not enable quantitative assessment of how benthic communities would respond to frequent and sustained water level reductions of 7 to 10 cm either because we did not have a long enough record of data (the field study) or the data are not suitable for assessing changes in water level (the historical data, which were collected in the river channel bottom with some type of dredge sampler).

Reductions in model-predicted water levels downstream in Lake Monroe are smaller (see Table 4–3). Below Lake Harney, the bed of the river intercepts sea level, and at below-average flows, the flow-stage relationship in this reach and downstream becomes decoupled (Chapter 6. River Hydrodynamics Results). Modeled stage reductions in Lake Monroe of up to 10 cm are experienced less frequently (see Figure 4–9), but do still occur. Under the 2030 land use scenario, with full withdrawal and projected sea level rise (Full2030PS), water levels increase under all conditions (see Table 4–3 and Figure 4–9). It is expected that the effects of withdrawals in this area of the river on freshwater benthic communities would be minimal to none based on the generally smaller predicted water level reductions and the lack of a relationship between flow and stage at below-average flows.

Table 4–3. Summary statistics of modeled water surface elevations for the base scenario (Base1995NN) versus the Full1995NN, Half1995NN, Full1995PN, Full2030PN, and Full2030PS scenarios. All elevations are m NGVD29.

SR520 (Lake Poinsett)	**Base1995NN**	**Full1995NN**	**Half1995NN**	**Full1995PN**	**Full2030PN**
Mean (m, NGVD29)	3.71	3.66	3.68	3.72	3.76
Standard deviation	0.65	0.66	0.66	0.58	0.58
Maximum (m, NGVD29)	5.31	5.31	5.31	5.33	5.32
75th percentile (non-exceedence)	4.28	4.24	4.27	4.24	4.28
Median (m, NGVD29)	3.75	3.65	3.70	3.67	3.73
25th percentile (non-exceedence)	3.25	3.18	3.22	3.26	3.28
Minimum (m, NGVD29)	2.42	2.42	2.42	2.42	2.42
US17/92 (Lake Monroe)	**Base1995NN**	**Full1995NN**	**Half1995NN**	**Full1995PN**	**Full2030PS**
Mean (m, NGVD29)	0.66	0.63	0.64	0.63	0.78
Standard deviation	0.43	0.41	0.42	0.42	0.42
Maximum (m, NGVD29)	2.52	2.45	2.49	2.59	2.75
75th percentile (non-exceedence)	0.87	0.83	0.85	0.83	0.99
Median (m, NGVD29)	0.53	0.50	0.52	0.51	0.65
25th percentile (non-exceedence)	0.34	0.33	0.34	0.33	0.49
Minimum (m, NGVD29)	0.04	0.03	0.04	0.03	0.17



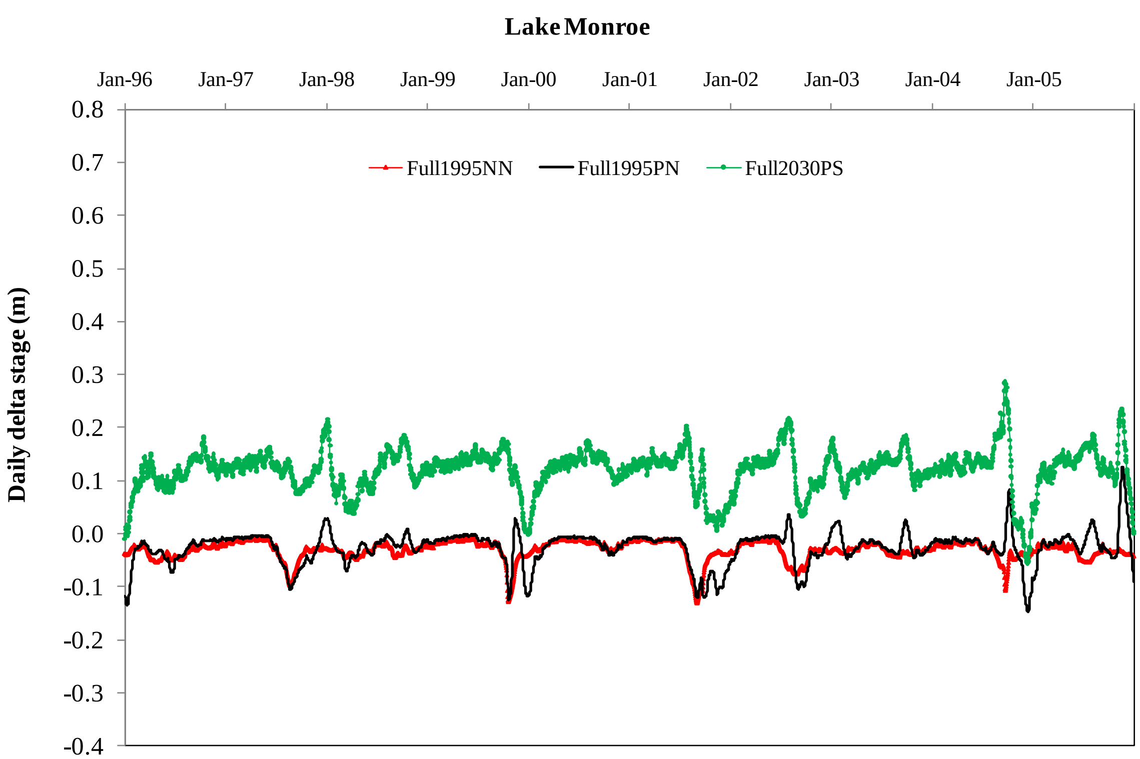


Figure 4–9. Time series of modeled delta stage at Lakes Poinsett and Monroe for the period 1996 to 2005 under three modeled withdrawal scenarios.

Direct Effects of Hydrology on Benthic Communities

The limited amount of data available for freshwater benthic communities limits the ability to make quantitative comparisons of benthic community characteristics and hydrologic measures. In the short-term field study conducted in 2009, high variability among the three sampling locations, and among sampling sites within each of those locations, did not permit any clear trends to be discerned that could be related to hydrology. The basic intent of the field study was to compare the benthic communities in similar habitats (bulrush and spatterdock) in a more hydrologically dynamic ecosystem (Lake Poinsett) with a less dynamic one (Lake Monroe and St. Johns River near Yankee Lake) to see if patterns in the benthic community could be discerned that could be related to hydrology. High variability in the data prevented this, along with inability to separate effects of hydrology from possible water quality effects. It is known that Lake Poinsett experiences periodic hypoxic events that occur throughout the upper basin under rising water level conditions (Chapter 7. Biogeochemistry). Permanent reductions in water level due to withdrawal, as seen in the test scenarios Full1995NN and Half1995NN, would be expected to have some adverse effects on benthic communities of the shallow fringing marshes and floodplains due to direct loss of aquatic habitat. The data from the 2009 field study represent an important baseline data set that can be used to compare to future conditions. Brooks et al. (2011) evaluated the effects of water withdrawals on benthic communities in unregulated rivers of New South Wales, Australia. They used limiting environmental difference (LED) modeling to compare invertebrate communities of reference streams, which were not supporting any withdrawals, to test streams, which were affected by withdrawals, and compared these results to field data collected from riffle and channel edge habitats. They assigned a level of adverse effect to invertebrate families based on the family being less or more frequently predicted to occur by the modeling than observed in the field studies (Table 4–4). They characterized withdrawal effects on edge habitat as being multifaceted, including water level reduction resulting in drying of habitat, degraded water quality, and reduction in area of aquatic habitat. Table 4–5 lists the families collected in our 2009 field study relative to the classification of Brooks et al. (2011) for invertebrate families collected from edge (channel margin) habitats similar to the habitats sampled in the 2009 field study. Seven families of invertebrates were listed as High likelihood of adverse effect from withdrawal (Table 4–5), eight were listed as Moderate likelihood of adverse effect and most families (22) were listed as Low likelihood of adverse effect.

Table 4–4. Levels of effect used by Brooks et al. (2011) in an evaluation of effects of water withdrawals on benthic communities in channel margin (edge) habitat in rivers of New South Wales, Australia.

Level	Description
High likelihood of adverse effect	Taxa consistently predicted to occur more frequently than observed; families within the upper 30% of statistically significant indicator values (IV) where the greater occurrence frequency was for the predicted assemblage.
Moderate likelihood of adverse effect	Taxa that were generally, but not consistently, predicted to occur more frequently than observed; families within the lower 70% of statistically significant IVs where the greater occurrence frequency was for the predicted assemblage.
Low likelihood of adverse effect	Taxa that occurred about as frequently as predicted; families with statistically non-significant IVs.
Moderate likelihood of beneficial effect	Taxa that were generally, but not consistently, observed more frequently than predicted to occur; families within the lower 70% of significant IVs where the greater occurrence frequency was for the observed assemblage.
High likelihood of beneficial effect	Taxa that were consistently observed more frequently than predicted; families within the higher 30% of significant IVs where the greater occurrence frequency was for the observed assemblage.

Source: Brooks et al. (2011)

Table 4–5. Classification of invertebrate families collected in the 2009 field study according to their potential sensitivity to water withdrawal, based on the assessment of Brooks et al. (2011) in channel margin (edge) habitat in southeastern Australian streams (see Table 4–4).

Group sensitivity to withdrawal	Families collected in 2009 field study
High likelihood of adverse effect	Dytiscidae, Hydraenidae, Hydrophilidae, Haliplidae, Ceratopogonidae, Baetidae, Corixidae
Moderate likelihood of adverse effect	Polycentropodidae, Planorbidae, Hydrometridae, Culicidae, Corduliidae, Aeshnidae, Psychodidae, Stratiomyidae
Low likelihood of adverse effect	Leptoceridae, Libellulidae, Coenagrionidae, Physidae, Hydroptilidae, Gomphidae*, Caenidae, Simuliidae, Palaemonidae, Ancylidae, Pyralidae, Hebridae, Lymnaeidae, Noteridae, Belostomatidae, Chironomidae, Hydrobiidae, Talitridae, Mesoveliidae, Nepidae, Pleidae, Naucoridae, Chrysomelidae
Moderate likelihood of beneficial effect	Thiaridae*
High likelihood of beneficial effect	None

* Known to occur in the study area but not collected in the 2009 field study (not included in counts in text).

Sampling of various marsh habitats in July 2009 in Lake Poinsett indicated some trends that appear to be related to hydrology. Highest taxa richness was seen in shallow marsh habitats with a duration of inundation of about 9 to 10 months (see Figure 4–4). This pattern could be due to a combination of protection from fish or invertebrate predation in combination with a hydroperiod suitable for macroinvertebrates that require longer duration aquatic habitat (Wellborn et al 1996; Wissinger et al. 2009). Lower taxa richness in the longer duration habitats could be due in part to better access for fish predators (or higher relative abundance of invertebrate predators such as odonates (see Figure 4–3 and Figure 4–4), which can also be tied back to hydrology. Marsh habitat subject to longer duration of inundation had higher relative abundance of aquatic and long-lived taxa. Using land surface elevation data collected by the SJRWMD Surveying Division at the July marsh stations at Lake Poinsett, in combination with the long-term stage-duration curve at the SR 520 gauge, estimates were made of the number of months of inundation at the nine marsh sites during a typical calendar year. Long-lived (e.g., Odonata) and aquatic taxa (e.g., Ephemeroptera, Amphipoda) appeared to reach peak abundance in areas subject to > 9 months of inundation (Figure 4–10). This is similar to the results obtained by Lillie (2003) in isolated wetlands in Wisconsin, who found highest relative abundance of some of the same taxa (Ephemeroptera and Amphipoda) in wetlands subject to 8 months or more of inundation. Motile taxa (Coleoptera and Diptera) and taxa with short life histories (Oligochaeta and Diptera) appeared to exhibit higher relative abundance in shorter duration wetland habitats. Taxa richness was lowest in the longest duration habitats (Figure 4–10), suggesting that reduced inundation should not affect overall benthic taxa richness, although species composition would be altered.

The GIS tools developed by the Wetland Vegetation Working Group were used to conduct a detailed analysis in Lake Poinsett wetlands in the area where benthic sampling was conducted in 2009 (Appendix 11.E). This analysis indicated minimal loss of acreage of wetlands inundated > 256 days yr^{-1}(about 8.5 months and greater) under the scenarios Full1995NN and Full2030PN. Wetlands inundated 45% to 55% of the time (164 to 201 days yr^{-1}) showed the greatest area experiencing reduced inundation (Appendix 11.E; Chapter 10. Wetland Vegetation). These wetland types appeared to have moderate to high macroinvertebrate taxa richness (see Figure 4–4), but many of these are short life cycle (e.g., Diptera) or motile taxa (e.g., Coleoptera, Diptera) that can move to new habitat as areas dry out (Figure 4–10). Thus the predicted changes in hydrology due to withdrawal likely will not affect species composition of the benthic community.

Seasonal changes in the P:R surrogate index (July to November) did not exhibit a clear pattern across all locations and sampling stations (Figure 4–11 and Figure 4–12). The expectation might be that all locations would shift more toward heterotrophy (i.e., a decrease in the index value) due to dieback of vegetation later in the year, in November. Because the entire study area is located in subtropical central Florida, it may be that the dieback is muted compared to northern locales; however, vegetation dieback may occur due to drying events (K. W. Cummins, Humboldt State University, pers. obs. 2008) and severe freezes (R. A. Mattson, SJRWMD, pers. obs. 2009). Merritt et al. (1999) likewise saw no seasonal changes in this index between spring and fall in the Kissimmee River system, to the south. Overall, it is difficult to draw inferences about seasonal changes in the P:R surrogate ratio with essentially two data points (July and November 2009).

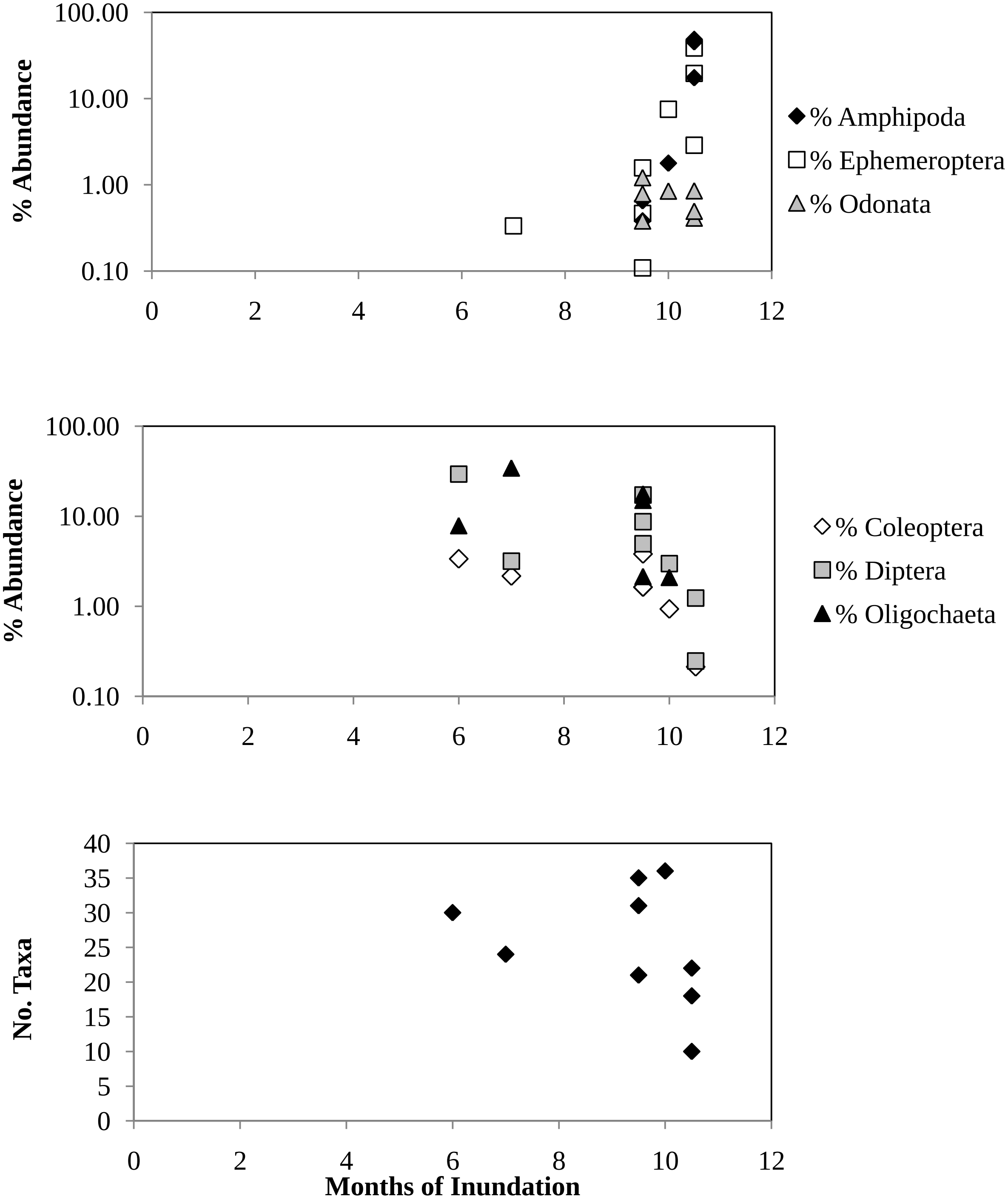


Figure 4–10. Relative abundance of long-lived benthic taxa (Odonata); taxa indicating more long-duration, permanently aquatic conditions (Amphipoda, Ephemeroptera); motile taxa (Coleoptera, Diptera); short life cycle/resistant taxa (Oligochaeta and Diptera); and taxa richness versus months of inundation in the nine marsh stations sampled in Lake Poinsett in July 2009. At each marsh station, n = 1.

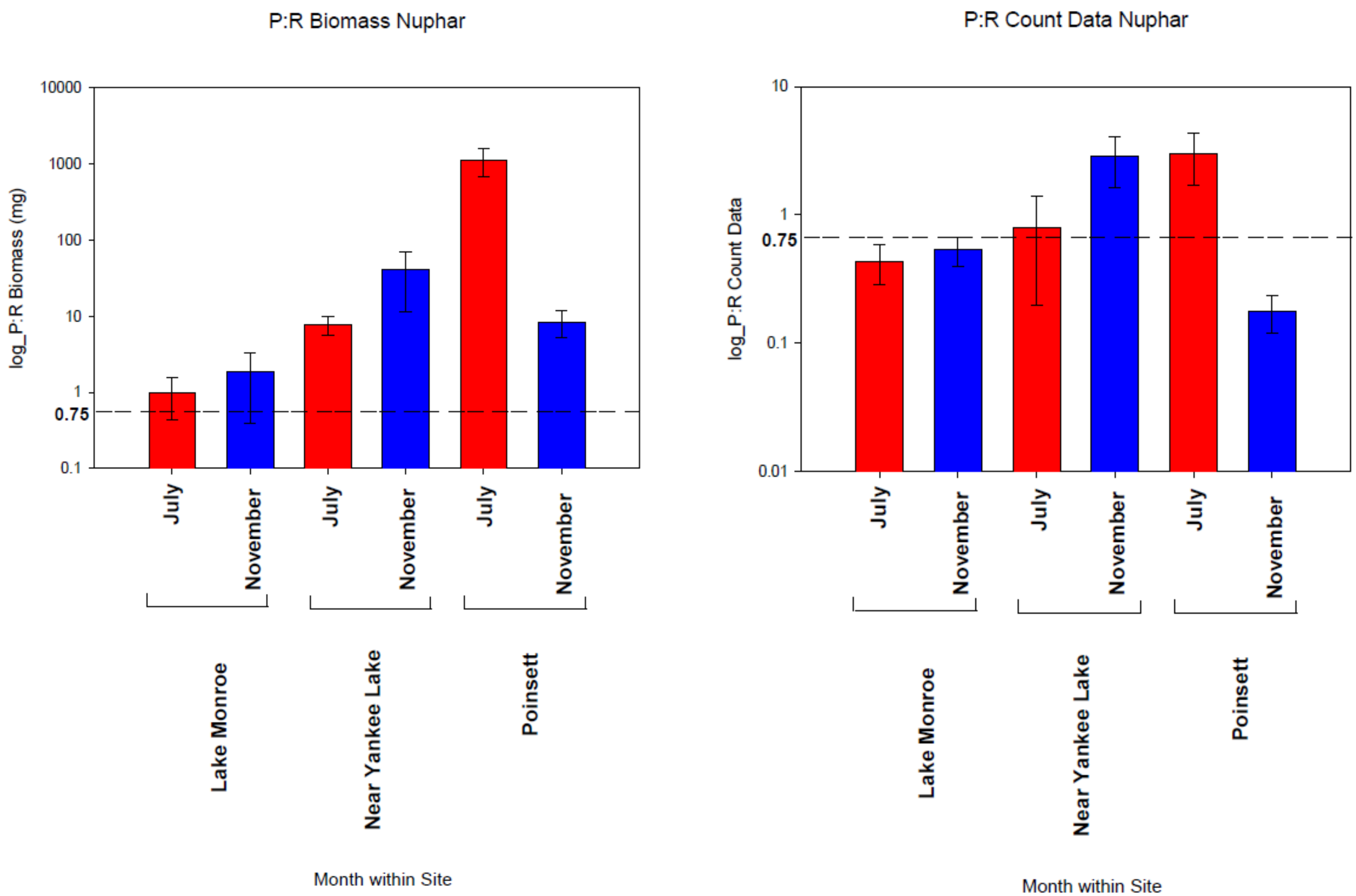


Figure 4–11. Production:respiration ratio surrogate index calculated as ratio of live plant feeders (live plant shredders + scrapers) to detrital plant feeders (detrital shredders + total collectors) for both biomass and relative abundance (count data) in spatterdock habitat (all sampling stations combined). Poinsett = Lake Poinsett.

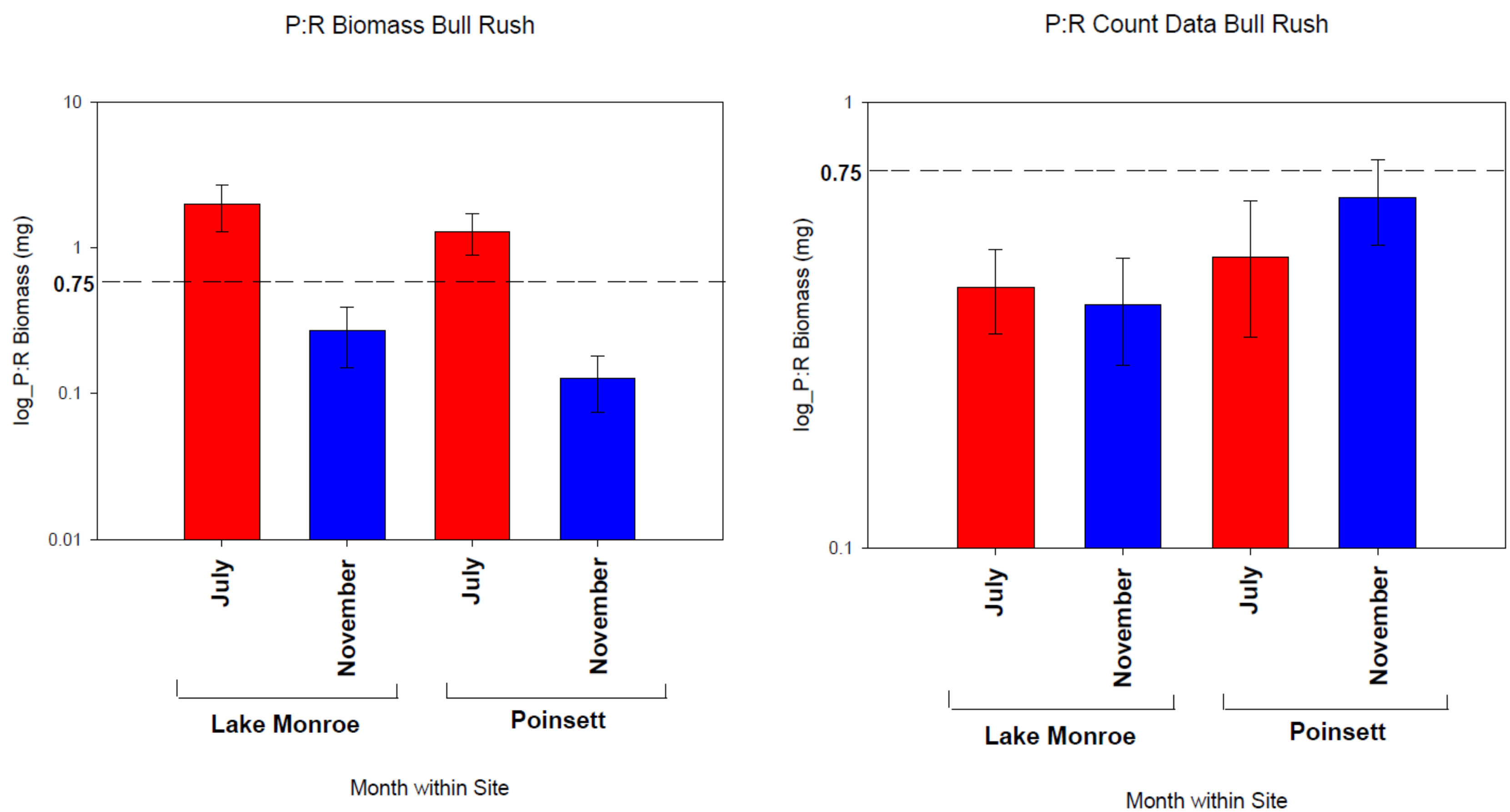


Figure 4–12. Production/respiration ratio surrogate index calculated as ratio of live plant feeders (live plant shredders + scrapers) to detrital plant feeders (detrital shredders + total collectors) for both biomass and relative abundance (count data) in bulrush habitat (sampling stations combined). Poinsett = Lake Poinsett.

We evaluated effects on benthic macroinvertebrate production a second way by obtaining annual production rates from several floodplain marsh habitats in the Kissimmee River (Koebel et al. 2005), which are also found in the USJRB:

- 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 of the above four values was 67.49 g m^{-2} yr^{-1}. We provided this average figure to the Biogeochemistry Working Group for use in their model that related area-weighted drying of floodplain habitat in Lake Poinsett (hectare-days) to constituent release rates (see Chapter 7. Biogeochemistry), substituting the average production rate for release rate. This estimated the potential loss of floodplain macroinvertebrate production under the test scenario Full1995NN. Scenarios that included the USJRB projects (Full1995PN and Full2030PN) predicted increased water levels, and hence no additional drying and related loss of invertebrate production. The worst case scenario resulted in a loss of 515.6 metric tons (MT) of floodplain invertebrate production annually for the period 1996 to 2005. Based on an estimated total annual production in the Lake Poinsett area of 3,500 MT, this represents a loss of about 14.7%.

<u>Indirect Effects of Hydrology on Benthic Communities</u>

The 2009 field study was designed to try to elucidate the effects of varying water levels on the benthic community and to provide baseline data for future comparison. This requires consideration of differences that may be attributed to site differences in other factors, such as vegetation structure and species composition and water quality. Sampling was focused mostly in similar vegetative habitats (bulrush and spatterdock) in both lakes to eliminate differences in vegetation structure. As seen in Table 4–1, water quality in both lakes is actually quite similar. Higher conductivity in Lake Monroe may affect invertebrate taxa sensitive to dissolved ionic content (Levings and Niu 2003). Periodic hypoxic conditions will affect benthic communities in both lakes due to loss of species requiring higher levels of DO (generally > 3.0 mg L^{-1}).

DO would be a water quality measure that could be affected by reduced flows due to less mixing and reduced re-aeration or higher temperatures. Much of the water quality monitoring data for the St. Johns River consists of synoptic sampling events, with single measurements of DO, usually conducted during daylight hours. These data are not particularly useful for evaluating existing effects of DO on benthic communities, or how changes in DO due to withdrawal may affect benthic communities. Continuous monitoring of DO at two locations on the river (Table 4–6) indicates that periods of hypoxia are common in Lake George and the LSJRB. The bottom environment in Lake George experienced DO < 2 mg L^{-1} about 41% of the time for the period June through September 2009 (J. Hendrickson, SJRWMD, pers. comm., 2010). The composition of the infaunal benthic community in Lake George indicates these conditions, with the more abundant taxa able to tolerate hypoxia.

Table 4–6. Continuous dissolved oxygen (DO) measurements in Lake George (segment 4; 11 June to 30 September 2009) and the St. Johns River at Federal Point (segment 3; 29 April to 8 September 2010).

Location	**Measurements, Excursions, and Durations**		
Lake George			
Continuous dissolved oxygen (DO)	< 1 mg L^{-1}	< 2 mg L^{-1}	< 5 mg L^{-1}
Number of Excursions below	126	184	127
Median	1 hour	1 hour	1.5 hours
Exceeded 25% of the time	3 hours	3 hours	6 hours
Exceeded 10% of the time	11 hours	9 hours	1.4 days
Maximum duration event	6.2 days	6.8 days	15.9 days
St. Johns River at Federal Point			
Continuous dissolved oxygen (DO)	< 1 mg L^{-1}	< 2 mg L^{-1}	< 5 mg L^{-1}
Number of Excursions below	28	40	76
Median	0.5 hour	1 hour	1.5 hours
Exceeded 25% of the time	3.5 hours	2 hours	5 hours
Exceeded 10% of the time	5 hours	7.5 hours	8.5 hours
Maximum duration event	7.3 days	7.9 days	8.3 days

The entire St. Johns River main stem is a designated impaired water under Florida's Impaired Waters Rule (Chapter 62-303 F.A.C.), primarily due to elevated nutrients and/or organic loading resulting in reduced DO in the river. The infaunal benthic communities of the unvegetated river bottom indicate these conditions (Water and Air Research, Inc. 2000), with a preponderance of individuals in these communities ($\geq$ 50%) consisting of tolerant taxa that are adapted to periodic low DO conditions (e.g., chironomid midges in the genus *Chironomus*, the oligochaete *Limnodrilus hoffmeisteri*). The overall condition of the benthic communities in the river main stem indicates a somewhat impaired system (Water and Air Research, Inc. 2000).

Small decreases in DO concentration are predicted by the Biogeochemistry Working Group that may be a result of reduced flows or increased organic loading (see Chapter 7. Biogeochemistry). These small changes will not significantly affect the in-channel benthic communities, because they are already adapted to periods of reduced DO. The benthic communities located in the littoral fringes of the river channel may be protected somewhat from reduced DO due to better mixing. The predicted increases in cyanobacterial bloom characteristics (see Chapter 8. Plankton) will also not have an adverse effect on benthic communities, but would be undesirable due to their continuing to promote proliferation of nuisance chironomid midge populations in portions of the river (Ali 1989, 1990). Alleviation of this issue will depend on reducing nutrient loading to the river and is not directly linked to withdrawal effects.

An additional invertebrate metric proposed for use in future monitoring is the Dissolved oxygen requirement index. This is calculated as the ratio of taxa that respire dissolved oxygen to those that are air breathers. Taxa that obtain oxygen from the water (via gills or cutaneous respiration) would be vulnerable to reduced DO levels resulting from decreased flow or increased decomposition of plant material from plant death related to falling water levels. Taxa that respire air or those with mechanisms to withstand periods of hypoxia (e.g., those with hemoglobin in their tissues) would be resistant to reduced DO levels. A threshold of 1.0 is used, with values greater than this indicating a benthic community potentially susceptible to withdrawal due to a majority of the taxa in the community requiring dissolved oxygen in the water to survive. This metric could be calculated using biomass or abundance data. It is evaluated here on a preliminary basis using taxa richness from the 2009 field study (Appendix 11.D)—52 of these taxa obtain their oxygen from the water column, and 26 are air breathers. This yields a ratio of 2.0 (52:26), which provisionally suggests that the benthic community could be vulnerable to water withdrawal effects if they promote lower oxygen conditions.

4.1.4 Effects of Hydrologic Change on Freshwater Target Taxa Populations

Table 4–7 presents the hydrologic criteria derived from the literature for the crayfish *Procambarus alleni* and *P. fallax*, and Table 4–8 presents the same for apple snail (discussion in Appendix 11.F). Various deep and shallow marsh habitats were shown to be preferred by both crayfish and apple snails (Table 4–7 and Table 4–8). At Lake Poinsett, marsh habitats extend from approximately 2.6 m NGVD29 to 3.5 m NGVD29 (Mace 2011). If the modeled change in elevation (referred to as the delta) between the base condition (Base199NN) and any scenario is subtracted from the measured stage (gauge data), one can judge the potential for effects on the target taxa. At Lake Poinsett, reductions predicted for two withdrawal scenarios appear to have minimal or no reduction effect on the hydrology of these habitats at median stage (Figure 4–13A). At lower lake levels (Figure 4–13B), shallow marsh habitat dries out naturally under existing conditions, thus inundation of these habitats would not be directly affected by stage reductions. With the predicted reductions, some loss of inundated deep marsh habitat would occur, but in general, there would still be available inundated deep marsh habitat that meets the criteria in Table 4–7 and Table 4–8. Even in shallow marsh habitat, if we assume that the shallow groundwater table near the lake tracks lake levels, the water table should be within 0.5 m of the land surface to meet the requirements of the burrowing crayfish *P. alleni* in at least some of the shallow marsh areas.

An alternative way to examine different hydrologic regimes and target taxa population effects is through frequency analysis (described in Appendix 11.G). Frequency analysis estimates how often, on average, a given event will occur. An event is typically a high or low water level and flow, which either floods or dewaters a target elevation for a specific duration significant to the ecological survival of the target taxon and/or its habitat at that elevation. For example, adult apple snail mortality may occur when dewatering happens for more than 12 weeks in their preferred shallow marsh habitat (Darby et al. 2008). Frequency analysis encompasses four types of events: (1) maximum average stages or flows; (2) minimum average stages or flows; (3) maximum stages or flows continuously exceeded; and (4) minimum stages or flows continuously not exceeded. Ultimately, the frequency analyses in Appendix 11.G provide a method for simple comparisons between the different hydrologic scenarios, estimating how often, on average, a given event will occur. An event as described for apple snails and crayfish is typically a low water level, which dewaters a target elevation for a specific duration significant to the ecological survival of target taxa and/or their habitats residing at that elevation.

Table 4–7. Hydrologic and habitat requirements for crayfish (*Procambarus alleni* and *Procambarus fallax*) in the St. Johns River basin wetlands.

Crayfish Species	Mean Water Depth (m)	Duration of Inundation (Months)	Dry Season Water Table Depth (m)	Preferred Habitats	Source
P. alleni	≤ 0.3	> 7 to 10	≤ 0.5	Shallow marshes; wet prairies	Dorn and Trexler 2007; Acosta and Perry 2001; Hendrix and Loftus 2000; Jordan et al. 1996; Evans et al. 2004a.
P. fallax	0.3 to 0.5	≥ 11	NA	Well-vegetated deep marshes; sloughs with dense submersed and emergent vegetation cover	Dorn and Trexler 2007; Hendrix and Loftus 2000; Jordan et al. 1996; Evans et al. 2004a.

Table 4–8. Apple snail (*Pomacea paludosa*) hydrologic and habitat requirements in the St. Johns River basin.

Hydrologic and Habitat Requirements	Adult	Juvenile	Source
Dry-down duration	< 12 weeks	4 to < 8 weeks	Darby et al. 2008
Dry-down timing	Minimize between April and June (during egg laying and recruitment)	Minimize between May and July (support juvenile recruitment)	Darby et al. 2008; Darby et al. 2003
Recurrence interval	2 to 3 years	Same as for adults	Darby et al. 2008; Darby et al. 2003
Habitat	Mixed shallow marsh	Same as for adults	Karunaratne et al. 2006; Turner et al. 2001

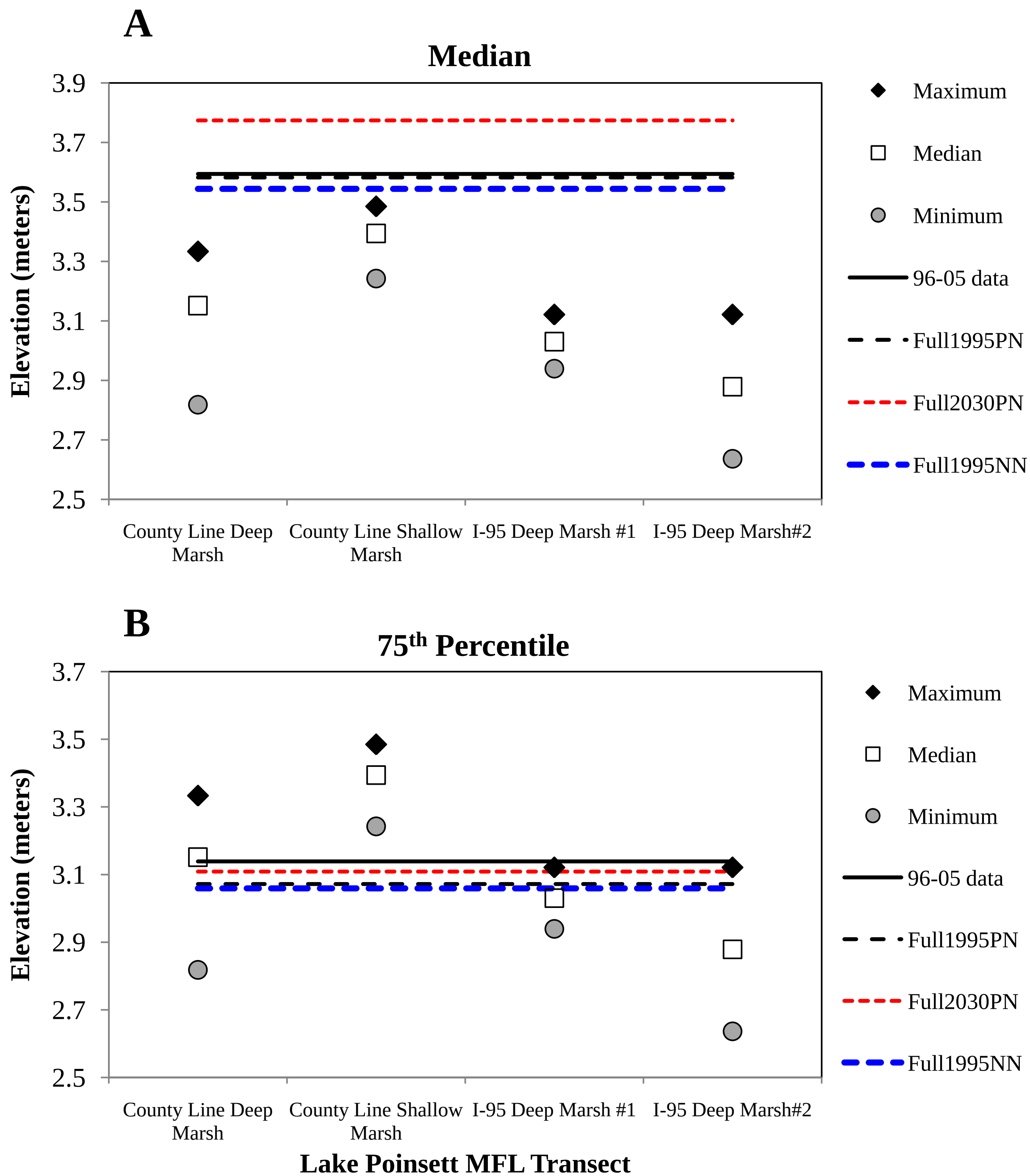


Figure 4–13. Effects of modeled water level reductions on target taxa habitats in Lake Poinsett at (A) median and (B) 75th percentile (exceedance) lake stages. Maximum, median, and minimum are surveyed land surface elevations (m NGVD29) for indicated marsh habitats at minimum flows and levels (MFLs) transects at the lake. Median and 75th percentile stage are actual measured gauge data; scenarios apply the modeled stage reduction or increase to the measured stage data.

Figure 4–14 illustrates the minimum continuously not exceeded frequency analysis for adult apple snails with three hydrologic scenarios. In this case, an event of importance for adult apple snail habitat occurs when the minimum shallow marsh ground elevation remains continuously dry for 90 days. The adult apple snail event criteria are based on pre-reproductive adult snail survival of up to 12 weeks (84 days) in dry (non-flooded) conditions in mesocosm experiments (Darby et al. 2008). Figure 4–14 illustrates the target shallow marsh elevation at Lake Poinsett with the modeled river stage frequencies (with the HSPF hydrologic model) for the St. Johns River at Cocoa (SR 520) base scenario (Base1995NN); modeled 1995 stage data with the full withdrawal scenario, the upper basin projects, and no sea level rise (Full1995PN); and modeled 2030 stage data with full withdrawal, the upper basin projects, and no sea level rise (Full2030PN). Comparing the frequencies for the 90-day continuously dry event for these three hydrologic scenarios at the average of the shallow marsh minimum elevations surveyed at Lake Poinsett provides a method to quantify the change in frequency between the different hydrologic regimes for this event. Figure 4–14 illustrates that this dry event is predicted to occur approximately 43 times in 100 years in the base scenario (Base1995NN), 25 times in 100 years under the Full1995PN scenario, and 20 times in 100 years with the Full2030PN scenario. Thus, HSPF hydrologic modeling predicts that this drying event would occur less frequently in the two future withdrawal scenarios than in the base scenario. In that there is no predicted increase in drying events, there appears to be no potential for adverse effects on apple snail. Table 4–9 and Table 4–10 summarize the above and other frequency analyses of the freshwater target taxa criteria.

4.1.5 Characterization of Key Effects for Freshwater Macroinvertebrate Communities

As reviewed in Dewson et al. (2007) and Poff and Zimmerman (2010), there is extensive literature relating benthic community structure (e.g., taxa richness, density, diversity, relative abundance, etc.) and function (e.g., productivity, species interactions, food webs, etc.) to hydrology and to alterations in hydrology due to human influence. Thus a material hydrologic change will alter benthic communities. The critical question is—will the hydrologic alterations predicted by the river models (both HSPF and EFDC) affect freshwater benthic macroinvertebrate communities in ways, and to a degree, that should cause management concern?

Five descriptors were developed to define and characterize levels of effect (discussed in more detail in Chapter 2. Comprehensive Integrated Assessment):

- Extreme—effect is persistent, strong and highly diverse; significant change in natural resource values
- Major— effect is persistent and 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 natural resource values
- Negligible— no appreciable change in any ecosystem attribute

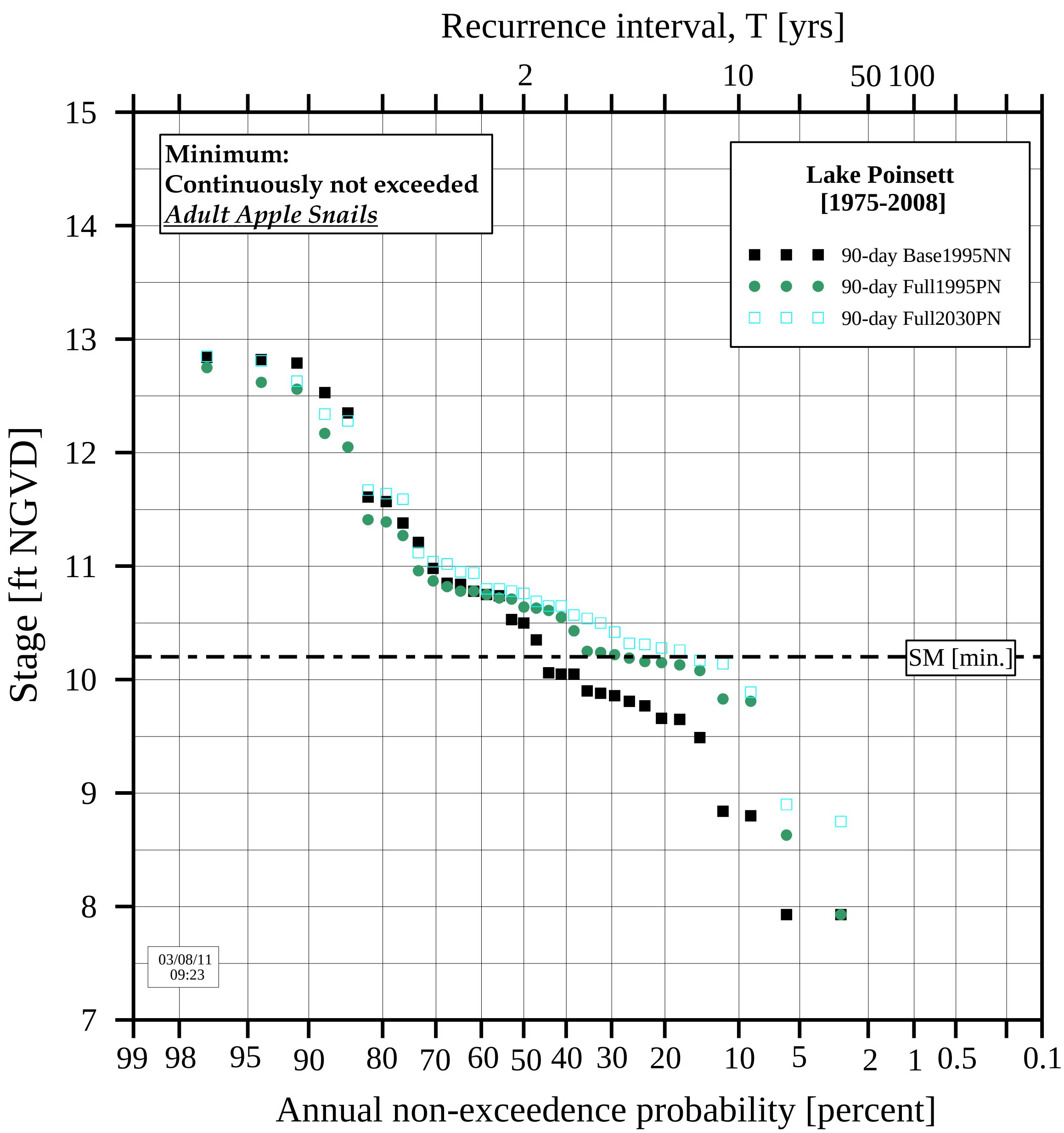


Figure 4–14. Frequency analysis of dewatering events of 90 days, corresponding to the 12-week threshold of drying for adult apple snail survival. Area below the shallow marsh (SM) minimum elevation represents critical dry down events for adult apple snails.

Table 4–9. Summary of results of frequency analyses in relation to proposed threshold hydrologic criteria for crayfish (*Procambarus alleni* and *Procambarus fallax)* and apple snail (*Pomacea paludosa)* species in Lake Poinsett (number of events in a 100-yr period).

Threshold		**Number of Events in a 100-Year Period**			
Crayfish	**Event**	**Base1995NN**	**Full1995NN**	**Full1995PN**	**Full2030PN**
P. alleni: 7 to 10-month inundation period	150-day duration dewatering event in shallow marsh (min. elevation)	64/100	64/100	62/100	60/100
P. fallax: > 11-month inundation period	365-day flooding event in deep marsh (max. elevation)	5/100	5/100	4/100	4/100
P. fallax: > 11-month inundation period	30-day dewatering event in deep marsh (max. elevation)	88/100	91/100	90/100	86/100
Apple snail	**Event**	**Base1995NN**	**Full1995NN**	**Full1995PN**	**Full2030PN**
Adults: ≤ 12 weeks (84 day) dry-down duration	90-day duration dewatering event in shallow marsh (min. elevation)	43/100	48/100	25/100	20/100
Juvenile: 4 to < 8 week (28 to 56 day) dry-down duration	60-day duration dewatering event in shallow marsh (min. elevation)	60/100	64/100	43/100	27/100

Table 4–10. Summary of results of frequency analyses in relation to proposed threshold hydrologic criteria for crayfish (*Procambarus alleni* and *Procambarus fallax)* and apple snail (*Pomacea paludosa)* species in Lake Monroe (number of events in a 100-yr period).

Threshold		**Number of Events in a 100-Year Period**			
Crayfish	**Event**	**Base1995NN**	**Full1995NN**	**Full1995PN**	**Full2030PS**
P. alleni: 7to 10- month inundation period	180-day duration dewatering event in shallow marsh (min. elevation)	2/100	2/100	2/100	< 1/100
P. fallax: >11-month inundation period	365-day flooding event in deep marsh (max. elevation)	18/100	18/100	18/100	35/100
P. fallax: >11-inundation period	30-day dewatering event in deep marsh (max. elevation)	40/100	40/100	40/100	10/100
Apple snail	**Event**	**Base1995NN**	**Full1995NN**	**Full1995PN**	**Full2030PS**
Adults: ≤ 12-week (84 day) dry-down duration	90-day duration dewatering event in shallow marsh (min. elevation)	< 1/100	< 1/100	< 1/100	< 1/100
Juvenile: 4 - < 8-week (28 to 56 days) dry-down duration	60-day duration dewatering event in shallow marsh (min. elevation)	< 1/100	< 1/100	< 1/100	< 1/100

Table 4–11 provides a summary of the predicted hydrologic changes under two scenarios with the least amount of water level reduction and associated biological responses of the benthic community. Based on the invertebrate data in hand, it appears that at higher flows (average and above), the effects of decreases in flow and water level would be relatively minor to negligible. Various marsh habitats critical to the support of freshwater target taxa in the upper basin would not be affected (Figure 4-14; Appendix 11.G). Predicted water level decreases in the middle and lower basins are generally similar to, or less than, those for the upper basin, and thus influences to the target taxa (and benthic communities in general) under average and above-average flow conditions would likewise be expected to be negligible. In the river channel bottom, the benthic communities are dominated by taxa adapted to sluggish flows and low DO conditions (Appendix 11.A); therefore, these communities should not be affected by the predicted reductions in flow throughout the middle and upper river. Under these conditions (average and greater flow), we would characterize effects of surface water withdrawal on benthic communities as negligible.

Impacts of flow reductions at lower flows would be expected to be greater, as a given unit of water withdrawn would have a proportionately larger effect on flow and/or water level (see Figure 4–13). In the middle and lower basins, it was shown that below average flow the flow-stage relationship is decoupled, because the river bottom is at or below sea level under these conditions, and the controlling influence on river and lake stage is the level of the Atlantic Ocean at the mouth of the river (Chapter 6. River Hydrodynamics Results). Because of this decoupling, water level reductions predicted by the EFDC hydrodynamic model in the middle and lower river (segments 1 through 6) are actually minimal at the lowest flow, and are highest at flood flows, which occur $\leq$ 10% of the time. This relationship suggests that the effects of withdrawal on benthic communities in the middle basin would be negligible. In the upper basin, water withdrawals appear to have more of an effect on flow and water levels at low flows, but the available macroinvertebrate data do not permit development of hydroecological models (e.g., regressions) that would enable quantitative assessment of the effects on benthic communities or detection of certain critical thresholds or break points.

As stated previously, the benthic communities in unvegetated, river channel bottom habitats of the upper river are generally dominated by taxa adapted to low DO and sluggish or no flows (Appendix 11.A), so they should not be affected by the projected flow decreases (either in terms of changes in water velocity or changes in DO). In the littoral marsh areas along the river channel and in Lakes Winder, Poinsett, and Puzzle reductions in water level at lower flows may affect the availability of habitat for macroinvertebrates, with higher water level reductions being of more concern due to greater loss of habitat volume or area. Because reductions of this magnitude are predicted on a regular basis (see Figure 4–9; Table 4–3) under the test scenario Full1995NN, we predict effects on benthic communities to be moderate, particularly in segment 8. The variability in the benthic data from the short-term field study conducted in 2009 did not permit detection of clear trends in community structure related to hydrology, thus we cannot quantitatively characterize effects on benthic communities in upper river littoral habitats due to lack of data, while in the river channel we would characterize effects as negligible. For the target taxa at lower water levels, effects of withdrawals are predicted to be minor due to the continuing availability of preferred marsh habitat. Under future scenarios, the target taxa would benefit due to reduced frequency of drying events (Appendix 11.G; Table 4–9 and Table 4–10).

Table 4–11. Summary of predicted effects on freshwater benthic communities in the upper and middle St. Johns River with hydrologic alterations predicted under modeling scenarios Full2030PN (segments 7 and 8) and Full2030PS (segments 4 through 6).

River Segment	Physical and Chemical Changes	Biological Changes
8	Increase in mean and median stage (Table 4–3; Figure 4–9	No adverse changes in benthic communities or populations (inferred based on projected increase in stage versus decline in stage)
	Larger Δ stage at lower river levels (Figure 4–13)	Minor effects on crayfish and apple snail habitat at lower stages (Figure 4–13)
	Periodic stage reductions of 7to 10 cm (Figure 4–9)	Potential effects on benthic communities due to loss of aquatic habitat (inferred); existing data do not permit quantitative assessment because of inadequacy
	Minimal reductions in dissolved oxygen concentrations (Chapter 7. Biogeochemistry)	Minimal effects on benthic communities due to existing periods of hypoxia (Chapter 7. Biogeochemistry) and dominance of taxa adapted to low dissolved oxygen conditions (Appendix 11.A)
7	Increase in mean stage (Chapter 3. Watershed Hydrology)	No adverse changes in benthic communities or populations (inferred based on projected increase in stage versus decline in stage)
	Larger Δ stage at lower river levels (Figure 4–13)	Minor effects on crayfish and apple snail habitat at lower stages (Figure 4–13)
	Minimal reductions in dissolved oxygen concentration (Chapter 7. Biogeochemistry)	Minimal effects on benthic communities due to existing periods of hypoxia (Chapter 7. Biogeochemistry) and dominance of taxa adapted to low dissolved oxygen conditions (Appendix 11.A)
6	Increase in mean and median stage (Table 4–3; Figure 4–9)	No adverse changes in benthic communities or populations (inferred based on projected increase in stage versus decline in stage)
	Reduced frequency of critical drying events (Appendix 11.G Table 4–9 and Table 4–10)	No adverse effects on crayfish and apple snail habitat (inferred based on projected increases in stage or minimal Δ stage)
	Minimal reductions in dissolved oxygen concentration (Chapter 7. Biogeochemistry)	Minimal effects on benthic communities due to existing periods of hypoxia (inferred) and dominance of taxa adapted to low dissolved oxygen conditions (Appendix 11.A)
5	Increase in mean stage (Chapter 6. River Hydrodynamics Results)	No adverse changes in benthic communities or populations (inferred based on projected increase in stage versus decline in stage). Effects on crayfish and apple snail habitat not assessed
	Minimal reductions in dissolved oxygen concentration (Chapter 7. Biogeochemistry)	Minimal effects on benthic communities due to existing periods of hypoxia (inferred) and dominance of taxa adapted to low dissolved oxygen conditions (Appendix 11.A)
4	Increase in mean stage (Chapter 6. River Hydrodynamics Results)	No adverse changes in benthic communities or populations (inferred based on projected increase in stage versus decline in stage). Effects on crayfish and apple snail habitat not assessed
	Minimal reductions in dissolved oxygen concentration (Chapter 7. Biogeochemistry)	Minimal effects on benthic communities due to existing periods of hypoxia (Table 3–2) and dominance of taxa adapted to low dissolved oxygen conditions (Appendix 11.A)

Water level reductions might have a greater effect on benthic communities in the floodplain, due to drying of wetlands as a result of decreases in water level. Because of the flat topography of the floodplain, apparently small decreases in water level could affect substantial areas of wetlands. The benthic invertebrates of the floodplain are adapted to its predictable seasonal drying (Williams 1996), and so they will likely adapt to small changes in wetland inundation characteristics (e.g., fewer days of inundation, etc.). Slight to moderate decreases in depth and duration of inundation in floodplain wetland habitats could actually result in higher macroinvertebrate taxa richness (Figure 4–4) due to protection from fish predation, although this might reduce the availability of invertebrate food to fishes due to restricting their access to the floodplain. Invertebrate productivity is higher in floodplain wetlands subject to longer periods of inundation (Sniffen 1981: Smock et al. 1992). Reductions in duration of inundation would reduce invertebrate production, with implications for fish production due to a reduced food base. As seen in the analysis above, invertebrate production in the floodplain of Lake Poinsett would be reduced by 14.7% under the worst case withdrawal scenario (Full1995NN).

Based on the data collected in the upper basin in this study, the more critical wetlands for benthic invertebrates are those with a duration of inundation $\geq$ 9 months (see Figure 4–10). The amount of these wetlands affected (acreage) appears to be minimal (Chapter 10. Wetland Vegetation; Appendix 11.E). Based on the Wetland Vegetation Working Group's assessment of the effects on floodplain wetlands, and the benthic community results presented in this chapter, we would characterize effects of withdrawal on floodplain benthic communities in the upper river as minor and in the middle river as negligible. Table 4–12 summarizes the characterization of the effects of the predicted hydrologic changes on freshwater benthic communities in the St. Johns River system.

4.1.6 Characterization of Uncertainty for Freshwater Macroinvertebrate Communities

As noted above, the lack of consistent, long-term benthic community data in the upper and middle river and the high variability seen in the benthic community characteristics in the 2009 field study limited our ability to make quantitative predictions of the effects of predicted hydrologic change on benthic communities. Some reaches of the river (segments 7, 5, and 4) were not assessed in this effort, and our predictions are based on best professional judgment using data collected on benthic communities in upstream segments (segments 8 and 6) in combination with the projected hydrologic changes from the modeling (both HSPF and EFDC models). Overall, the level of uncertainty attached to these predictions is medium to high for most freshwater reaches of the river due to weak or no predictive ecological models and complete lack of data in some river segments (7, 5, and 4), although there is reasonably good support from the literature. This uncertainty is indicated in Table 4–12.

4.1.7 Potential Means of Mitigation of Effects for Freshwater Macroinvertebrate Communities

To minimize effects of water level reductions on benthic communities, the magnitude of those reductions should be minimized where possible through regulation of the withdrawal, particularly in the upper basin, where the effects on water levels appear to be greatest. Larger water level reductions in response to a given rate of withdrawal would likely affect benthic communities to a greater degree, so regulatory measures such as a sliding water withdrawal schedule (reduced withdrawal rates as flow progressively declines) and low flow cutoff of withdrawal could help minimize effects. A low flow cutoff of 194 mgd (8.5 $m^3 s^{-1}$) was incorporated into the HSPF hydrologic model for the upper basin (Chapter 3. Watershed Hydrology). These low flow cut-off criteria could also be considered as permit applications are submitted for other withdrawal points downriver.

4.1.8 Monitoring Needed for Adaptive Management for Freshwater Macroinvertebrate Communities

Confirmation of hydrologic and ecological assessments with respect to benthic communities can only be accomplished through monitoring. As discussed above (Section 4.1.6), the data set for macroinvertebrates, especially in the freshwater reaches, is weak. Establishing an ongoing program of monitoring benthic macroinvertebrate communities in the St. Johns River basin will provide a very useful data set to complement the other long-term data sets that currently exist (hydrology, water quality, and phyto- and zooplankton). Effects of withdrawals on benthic communities cannot be quantitatively assessed without more data. Merritt and Cummins (2011) had several recommendations for a monitoring program:

(A) Collect numerical abundance data (relative abundance) of selected taxa or taxa groups. Although biomass is a better measure of ecosystem function, numerical abundance is a suitable, less expensive surrogate for biomass.

(B) Classify taxa according to FFGs in order to elucidate the nutritional structure of the macroinvertebrate community. This approach also reduces costs because it is effective with lower resolution taxonomy, which could be accomplished in the field. Calculate ratios between FFGs (and other life habit categories) in order to track the status of ecosystem function. Because the ratios are dimensionless numbers, they are relatively independent of sample size and can be obtained from semiquantitative field samples. Several metrics that may be useful are suggested below.

(C) Concentrate on populations of key macroinvertebrate taxa with the highest probability of being disrupted by habitat alterations resulting from water withdrawals. For example, the Odonata that usually require more than 1 year to complete their life cycle, would be susceptible to loss of wetted habitat of $\geq$ 1 yr inundation duration.

(D) Establish long-term sites and sample them at the same seasons and within the same habitats year-to-year. Because of very high variability (overlapping standard errors), year-to-year comparisons should be restricted to the same site and season and standardized to a specific habitat.

Table 4–12. Summary characterization of effects of withdrawals on freshwater benthic communities under Full1995NN and Full2030PN (segments 7 and 8; water level effect); Full1995NN and Full2030PS (segments 4 to 6; water level effect); and FwOR1995NN and FwOR2030PS (segments 2 and 3; salinity effect).

River Segment	Δ FW Benthic Community 1995	Δ FW Benthic Populations 1995	Δ FW Target Taxa Populations 1995	Δ FW Benthic Community 2030	Δ FW Benthic Populations 2030	Δ FW Target Taxa Populations 2030	Overall
1	NA	NA	NA	NA	NA	NA	NA
2	***	***	***	***	***	***	***
3	***	***	****	***	***	***	***
4	*****	*****	*****	*****	*****	*****	*****
5	*****	*****	*****	*****	*****	*****	*****
6	****	****	***	****	****	***	***
7	****	****	****	****	****	****	****
8	****	****	***	****	****	***	****

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

Note:
NA = Not applicable
Hatching = River region or ecological component not directly assessed

Other Metrics for Future Consideration

Three other metrics could be useful in future monitoring using benthic macroinvertebrates, in terms of assessing effects of water level changes due to withdrawal: mobility index, voltinism index, and habitat stability index.

Mobility Index

Calculated by biomass or abundance as the ratio of those invertebrate taxa with low or very low dispersal ability versus those with high dispersal ability. Those taxa with low capability for dispersal cannot readily move to adjacent aquatic (inundated) habitat as wetland areas where they reside dry up. Some of these nonmotile taxa (e.g., amphipods, grass shrimp) are important food items for fish and wildlife, with the potential impacts on food webs. The motile taxa would be largely unaffected by drying due to water withdrawal, as they could follow receding water levels or disperse to adjacent inundated wetland areas (e.g., adult Coleoptera and Hemiptera). All of the aquatic insects (except Collembola) have winged adult life stages and thus have a higher, although variable, potential for dispersal. A preliminary assessment of this metric was conducted looking at the total taxa collected in the 2009 field study (Appendix 11.D). A threshold value of 1.0 is proposed, with a value greater than 1.0 indicating that the community may be susceptible to withdrawal because a majority (> 0.50%) of the community would consist of individuals with limited motility and dispersal capability. A ratio less than 1.0 would suggest that the benthic community would be resistant to withdrawal effects because a majority of the taxa would be able to disperse to new habitat. Of the taxa listed in Appendix 11.D, 15 are classified as having low to very low motility, and 46 as highly motile, giving a preliminary ratio of 0.33. The provisional conclusion would be that, in general, the macroinvertebrate fauna would not be highly vulnerable to water withdrawal.

Voltinism Index

Calculated as the ratio of invertebrate taxa with longer life cycles (semivoltine = >1 yr) to those with shorter life cycles (univoltine or polyvoltine = < 1 yr or multiple generations per year). The length of life cycle, egg to adult, would be an indicator of the rapidity with which a taxon can respond to loss of habitat due to drying. If relative abundance of longer life cycle taxa is reduced, this could also have food web effects both because of loss of food base for fish and wildlife, and because many of these taxa (e.g., odonates) are important intermediate predators in wetland and aquatic food webs. Again, a threshold of 1.0 is proposed, with values > 1.0 suggesting the community could be vulnerable to water withdrawal effects due to a majority of the taxa being longer life cycle organisms. Of the total taxa collected in the 2009 field study (Appendix 11.D), 13 were tentatively classified as semivoltine (very little life history work has been done on aquatic invertebrates in Florida), giving a ratio of 0.2. This indicates a majority of the fauna collected were shorter life cycle taxa that could avoid drying effects or could rapidly recover from them, with the provisional conclusion that the benthic community would not be vulnerable to water withdrawal because of life history characteristics.

Habitat Stability Index

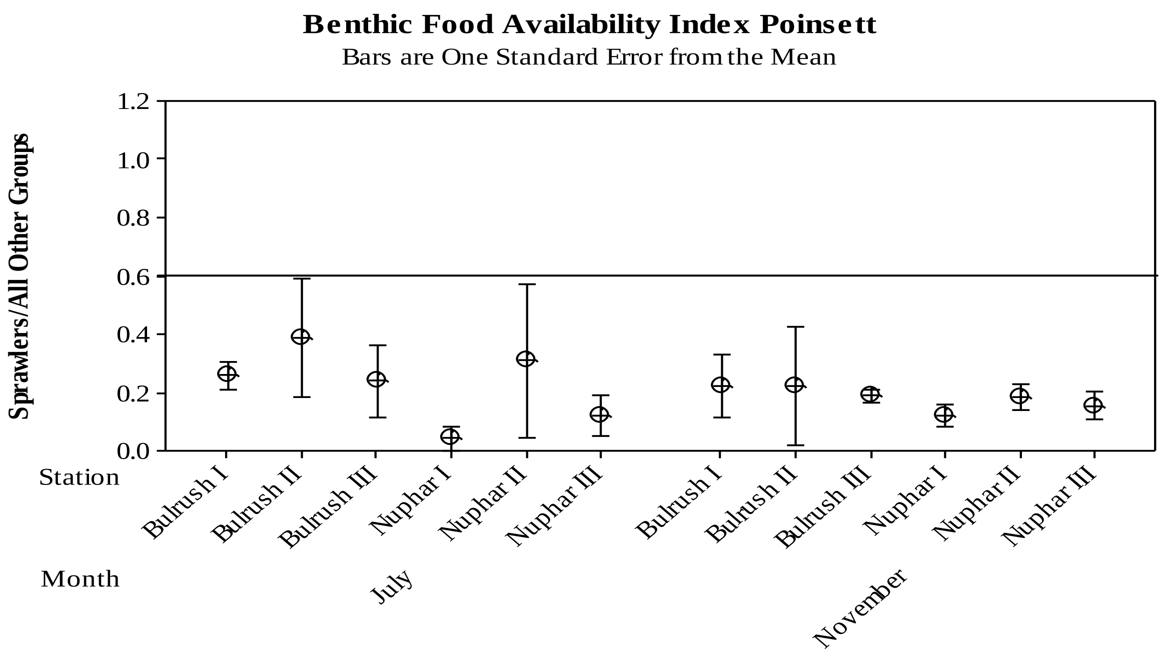
Two versions of this metric were listed in Table 4–2. The second version (Habitat Stability II) appears to be more useful for the St. Johns River fauna, as it relates those taxa that require stable habitat for feeding (e.g., the amphipod *Hyalella* grazing on periphyton) or attachment (filtering collectors) to those that do not require a stable substrate such as swimming, sprawling, and burrowing invertebrates. Many of the former group are important fish food items. As indicated in Table 4–2, a threshold ratio of 0.6 is used from Merritt et al. (1996, 1999) with a ratio greater than this indicating a benthic community dominated by organisms requiring stable substrate. Substantial loss of this substrate, due to declines in water level, would have negative effects on the benthic community. Figure 21 in Appendix 11.C indicates that the benthic community at most of the bulrush and spatterdock sites in Lake Monroe, Lake Poinsett, and St. Johns River near Yankee Lake have ratios < 0.6, with a few sites having a ratio much greater than 0.6. The provisional conclusion is that the benthic communities are not susceptible to water withdrawal because of loss of substrate.

4.1.9 Other Considerations for Freshwater Macroinvertebrate Communities

Two other issues are discussed here. First, during the course of fieldwork conducted in this study, beds of freshwater mussels (Unionidae) were common and abundant in the upper and middle river. Substantial beds of live mussels were found in Lake Poinsett, in the river channel at State Road (SR) 50, in the river channel at SR 46 (upstream of Lake Harney) and in Lake Harney. This group of freshwater clams is among the most imperiled group of aquatic fauna in the continental United States (Master et al. 1998). As noted in Appendix 11.A, a mussel endemic to the St. Johns River, *Elliptio monroensis*, was identified from Lake Harney (J. D. Williams, FWC, pers. comm., 2010) during this effort. Mussels could have been another target freshwater taxon in this study but two factors precluded using them:

- The taxonomy of the mussel fauna of the St. Johns River is not well understood. Recent unpublished genetic work with one of the most common species in the drainage, *Elliptio buckleyi*, indicates that it appears to be a complex of species (J. D. Williams, FWC, pers. comm., 2010). Therefore, we do not know the taxonomic identity of one of the most common mussels of the river.
- There have been no studies of the life history or environmental requirements of mussels in the St. Johns River or in Florida in general. Studies of mussel populations in the Flint River basin in Georgia (Gagnon et al. 2004; Golladay et al. 2004) indicated that some taxa were resistant to reduced flow and DO associated with drought conditions (generalist taxa), while other mussel taxa were sensitive to DO < 5 mg L^{-1}. Overall mussel mortality during drought ranged from 13% to 93%. Effects tended to be greatest in medium-sized streams, with the mussel assemblages of small and large streams being more resistant to drought, due to natural resistance in the former and minimal effects of drought on stream habitat in the latter.
- Increasing dominance by the exotic Asian clam (*Corbicula fluminea*) may result in loss of native bivalve taxa that would not be directly attributable to reduced water flows or levels.
- Until we collect more data on the mussel fauna of the St. Johns River drainage, and work out the taxonomic issues, we cannot assess the effects of the hydrologic changes predicted in this study on mussel populations of the river.

The second issue that requires more study is the linkages between benthic macroinvertebrate communities and fish populations. As discussed in Section 2.2.5, benthic macroinvertebrates are important components of the diets of many sport fish. Lower invertebrate productivity in floodplain wetlands subject to less inundation has been established (Sniffen 1981; Smock et al. 1992), and thus reductions in floodplain inundation could reduce invertebrate productivity and the food base available to fish populations. To the extent that predicted reductions in floodplain inundation appear to be minor (Chapter 10. Wetland Vegetation), the effects on floodplain benthic communities should also be minor, with similar effects on fish populations. Merritt et al. (1996) derived a Benthic Food Availability surrogate index using benthic invertebrates, a ratio of > 0.6 indicating a good food supply for benthic-feeding fish. At the three study locations sampled in 2009 (Lake Poinsett, Lake Monroe, and St. Johns River near Yankee Lake) the index is < 0.6 at most of the sampling stations (Figure 4–15 and Figure 4–16) in bulrush and spatterdock habitat on the lake shoreline. This appears to indicate that fish populations at these locations are, in part, sustained by other food resources, such as zooplankton and drifting invertebrates. Analysis of fish food habits from prior studies in the river indicated that planktonic taxa such as copepods and cladocerans and semi-planktonic ostracods are important components of the diet of many fish (R. A. Mattson, SJRWMD, pers. comm., 2011; summary spreadsheets are available upon request). Subtle changes in benthic communities that may occur with small reductions in hydrology, such as those associated with the evaluated scenarios, may not be a major consequence for fish. This is another issue that could use more data and study in order to strengthen our understanding of the linkages between benthic communities and fish.



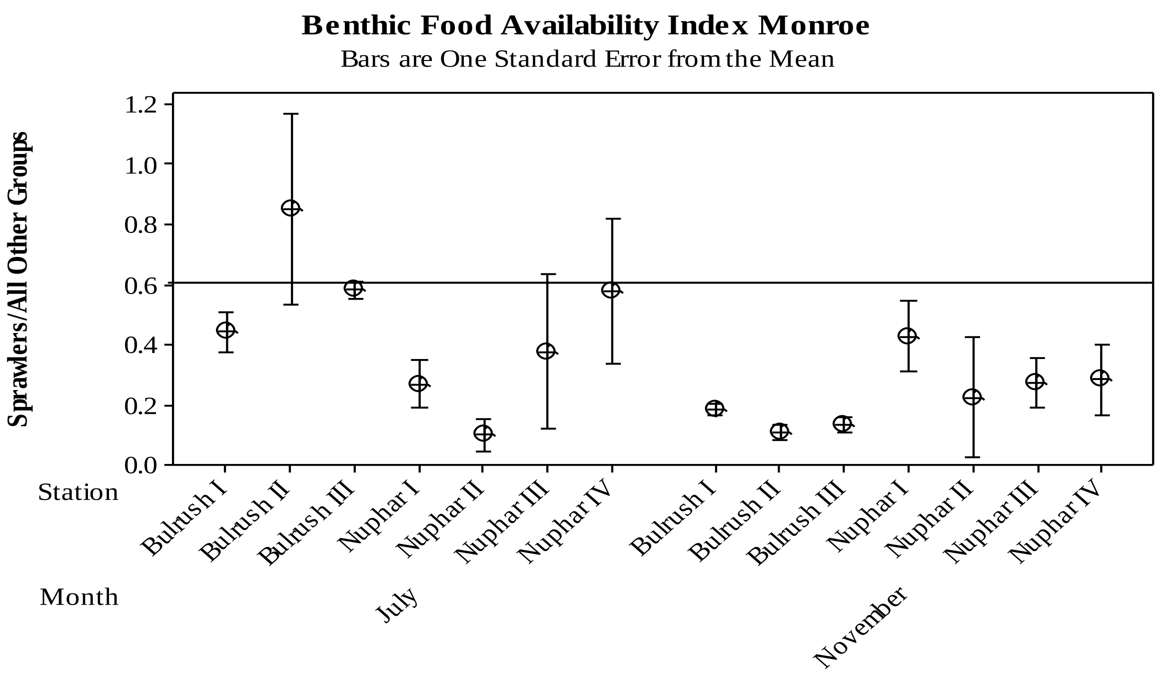


Figure 4–15. Benthic Food Availability Index calculated as the ratio of sprawlers to all other life habit groups (clingers + climbers + burrowers + swimmers). 0.6 threshold is indicated. n = 3 at each station. Bars are one standard error from the mean.

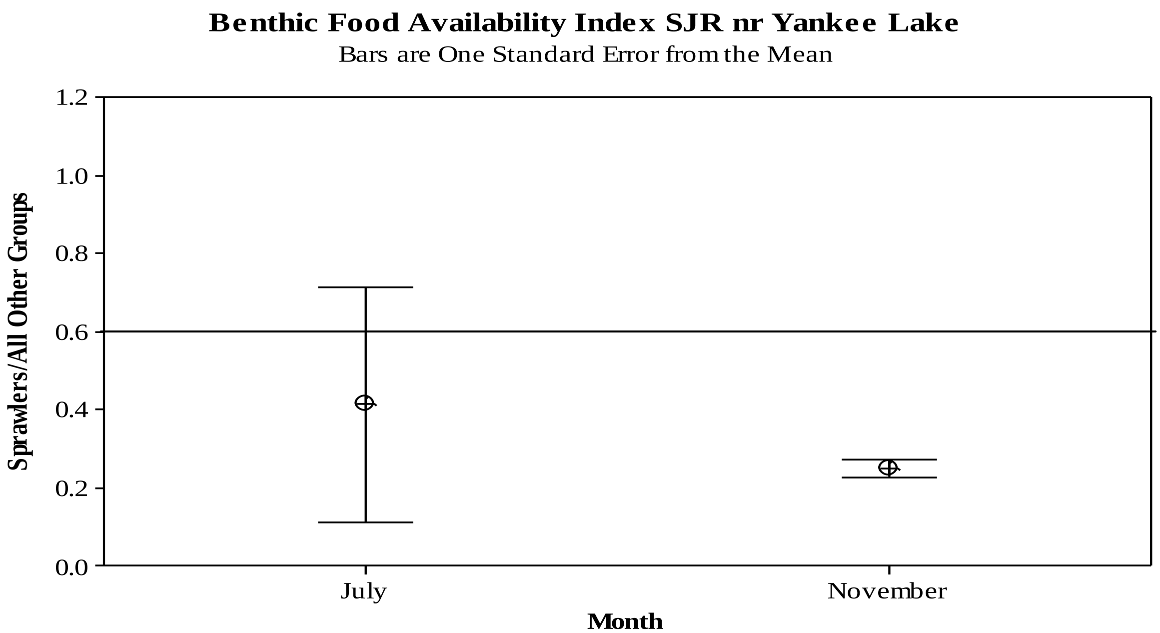


Figure 4–16. Benthic Food Availability Index calculated as the ratio of sprawlers to all other life habit groups (clingers + climbers + burrowers + swimmers). 0.6 threshold is indicated. n = 3 at each station, SJR = St. Johns River. Bars are one standard error from the mean.

4.2 Estuarine Macroinvertebrate Communities

4.2.1 Effects of Salinity on Community Composition and Population Abundance

Effects on Community Structure

Nonmetric multidimensional scaling (MDS) analysis at multiple taxonomic levels (species, genera, family, and phyla) indicated that salinity exerts a major effect on benthic community structure (Figure 4–17). The sampling sites clustered into a group of eight low-salinity sites (mean salinity 0.4 to 5.8 ppt, although one site in this group had a mean salinity of 8.5 ppt) and nine higher salinity sampling sites (mean salinity 13.6 to 25.7 ppt; Montagna et al. 2011). The other water quality variables analyzed with MDS were DO, pH, and water temperature. These, along with salinity, were the only water quality data collected concurrently with the benthic samples at the same locations. None of the other water quality variables accounted for as much variation in community structure as salinity (Montagna et al. 2011). Earlier work with benthic communities of the lower river SAV beds (Montagna et al. 2008a) also indicated that salinity was a major explanatory factor of benthic community structure.

Abundance (as # individuals m^{-2}) was highest at the lowest salinity sites (Figure 4–18). Peak abundance was at a mean salinity of 0.4 ppt and the regression was statistically significant (r^2=0.55, p<0.004). Abundance was higher at salinities below approximately 2 to 3 ppt (Figure 4–18). The primary mechanistic basis for this relationship is likely to be the increase in salinity variation that accompanies increasing mean salinity. The lower reaches of the estuary (segment 1 and the lower portion of segment 2) exhibit the highest amount of salinity variation, over both shorter (days to weeks) and longer (months to years) time scales (Chapter 6. River Hydrodynamics Results). This variation creates a level of physiological stress that reduces abundance. High variation in salinity creates physiological stress even in relatively pristine estuarine systems such as Apalachicola Bay (Livingston 2003). Factors other than salinity may also contribute to the decline in abundance toward the mouth. This area of the estuary is also influenced by high-density urban land uses in the downtown Jacksonville area and industrial land uses associated with the Port of Jacksonville. Contamination by heavy metals and organic toxins (e.g., PAHs, PCBs) has been documented in this area (J. Higman, SJRWMD, pers. comm., 2010). Previous work (Evans et al. 2004) documented widespread sediment contamination and related effects on chironomid populations in these reaches of the estuary. This could also explain the reduced abundance in the higher salinity areas, which are located in these more pollutant-impacted areas of the estuary.

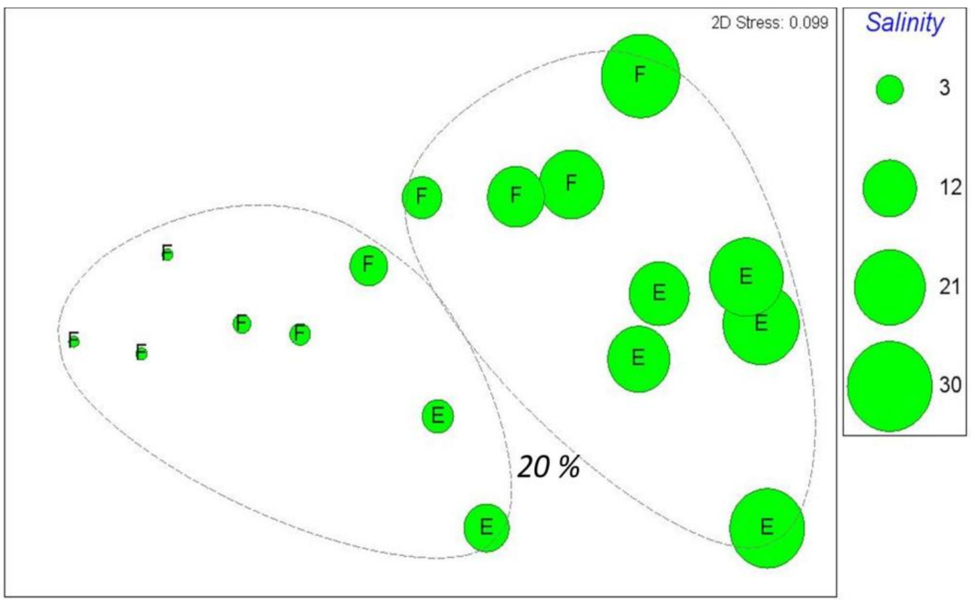


Figure 4–17. Nonmetric multidimensional scaling plot showing the relationship between benthic community structure and mean salinity (adapted from Montagna et al. 2011). F = Florida Department of Environmental Protection (FDEP) sites; E = Environmental Monitoring and Assessment Program (EMAP) sites.

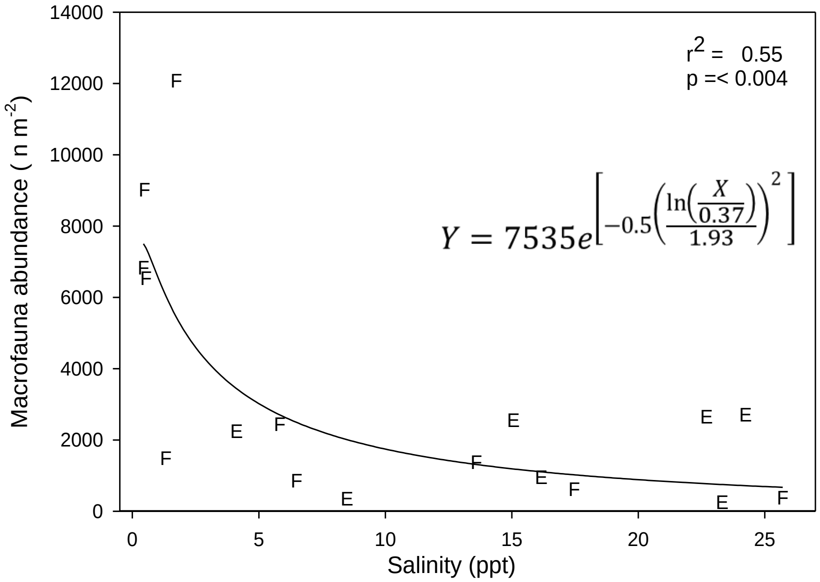


Figure 4–18. Regression plot of mean benthic community abundance versus mean salinity in the St. Johns River estuary (adapted from Montagna et al. 2011). F = Florida Department of Environmental Protection (FDEP) sites; E = Environmental Monitoring and Assessment Program (EMAP) sites.

Effects on Benthic Populations

Montagna et al. (2011) presented the results of nonlinear regression analysis of mean salinity versus abundance of numerically dominant taxa in the estuary. Table 4–13 presents a summary of these analyses. Higher taxa (i.e., phylum, class) were analyzed for linear relationships with salinity; lower taxa (i.e., family, genus) were analyzed for the presence and location of an optimum. Analysis indicated three patterns for the covariance between the abundance of taxa and increasing salinity: increasing abundance, decreasing abundance, or peak abundance at some optimum mean salinity (Figure 4–19). Nine taxa declined with increasing salinity. These were mostly freshwater taxa, such as aquatic insects and oligochaetes. Six taxa displayed increased abundance with increasing salinity; these six were groups with mostly marine taxa (various echinoderms, phoronids, etc.). Fourteen taxa had peak abundance at some mean salinity. Nineteen taxa had no significant relationship with mean salinity, although five of these displayed some type of peak abundance at an optimum salinity. Apparent incongruities in salinity responses (e.g., Mollusks as a group decrease in response to increasing salinity, while the snail family Hydrobiidae and the hydrobiid snail *Littoridinops* sp. display salinity optima) are likely due to the variable responses to salinity exhibited by the array of species constituting the Mollusks, that is, some species decrease in abundance and others increase in response to salinity (Boesch 1977).

Model-predicted Salinity Changes

Analysis presented in Chapter 6, River Hydrodynamics Results, indicates that under the potential future scenario (FwOR2030PS), there will be some upstream encroachment of higher salinities (Figure 4–20). These will mostly occur in segment 2. An earlier analysis (ECT 2008) projected the 5 ppt isohaline (depth integrated) to move upstream 0.48 to 1.13 km at withdrawals of 206.9 to 310.3 mgd (9.06 to 13.59 $m^3 s^{-1}$) respectively. Analysis of spatial changes in salinity was conducted by the Fish Working Group (see Chapter 12. Fish). Under the FwOR1995NN test scenario, decreases in the area of lower salinity zones (< 0.5 ppt and 0.5 to < 5 ppt) and increases in the area of higher salinity zones (5.0 ppt and greater) occur (Figure 4–21). The salinity zones 0 to < 0.5 ppt and 0.5 to < 5 ppt decreased in area by 1.9% to 3.9%, while the zones 5 to < 12 ppt, 12 to < 18 ppt, and 18 to < 30 ppt increased by 0.4% to 7.1%.

Frequency analysis of the model output salinity (Appendix 11.G) indicates that in the upper oligohaline area of the estuary (SR16 at Green Cove Springs), small increases in salinity due to withdrawal (a given salinity present for a greater number of days at a 2-year recurrence interval) will occur on an average annual basis. Withdrawals will have greater effects on salinity at drought conditions (5 to 10-yr recurrence). Figure 4–22 (analysis in Appendix 11.G) shows that bottom salinities at this location will increase up to a maximum of 4 ppt for about 10 days during a 1 in 10-year drought under the projected future scenario (FwOR2030PS). Interestingly, this is about the same increase even in the absence of the upper basin projects and historic rate of sea level rise (FwOR1995NN), suggesting that withdrawals from the Ocklawaha River will have a greater effect on salinity dynamics in the estuary. Downstream in segment 2 (Figure 4–23; Appendix 11.G –JAXSJR40 station), lower salinities will display larger increases, with the higher salinities (≥ 15 ppt) not changing substantially. In segment 1 (Figure 4–24; Appendix 11.G–JAXSJR17 station) salinities will change very little across all flow conditions. Chapter 6, River Hydrodynamics Results, likewise indicates that there will be salinity increases in the estuary over the base condition (Base1995NN), although the scenario indicates the proposed deepening of the commercial navigation channel in the estuary will have a greater effect on estuarine salinity regimes than withdrawals.

Table 4–13. Summary of nonlinear regression analyses of taxa population responses to increasing salinity in the St. Johns River estuary (adapted 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.

* Statistically significant regressions ($p < 0.05$).

Populations increase **Populations decrease** **Population exhibits optimum**

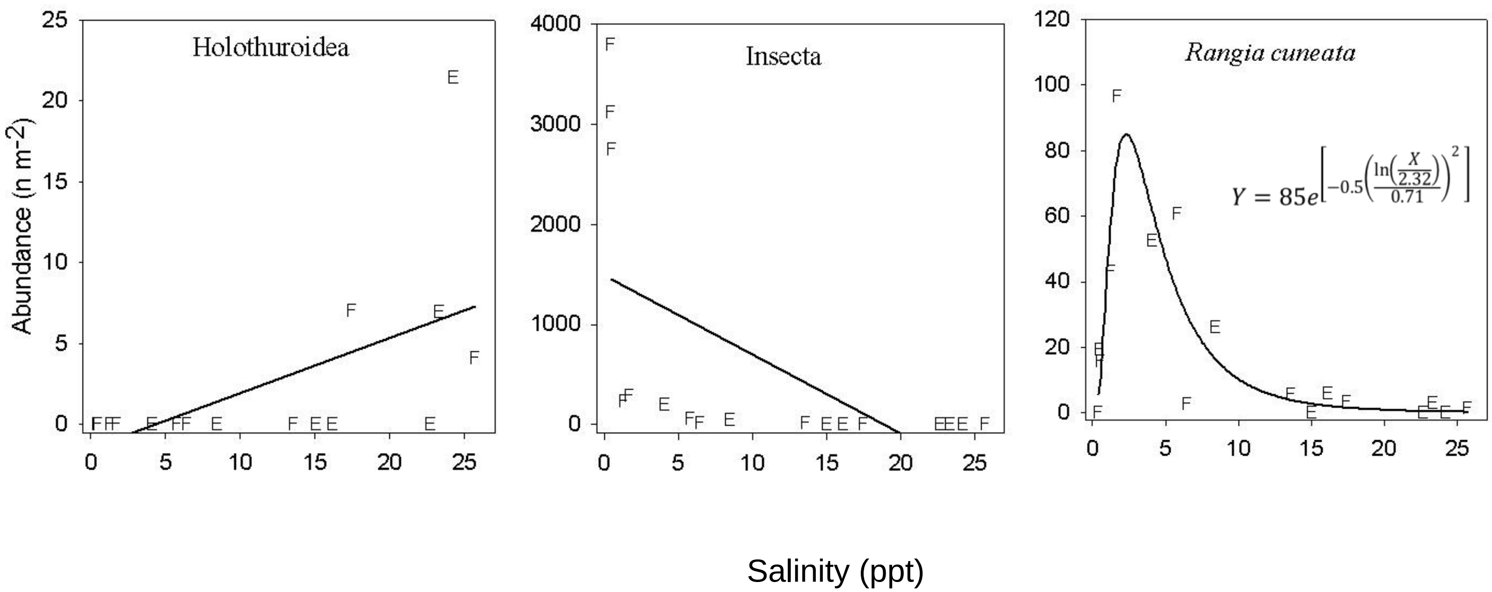


Figure 4–19. Regression plots of mean benthic community abundance versus increasing mean salinity in the St. Johns River estuary. (adapted from Montagna et al. 2011). F = Florida Department of Environmental Protection (FDEP) sites; E = Environmental Monitoring and Assessment Program (EMAP) sites.

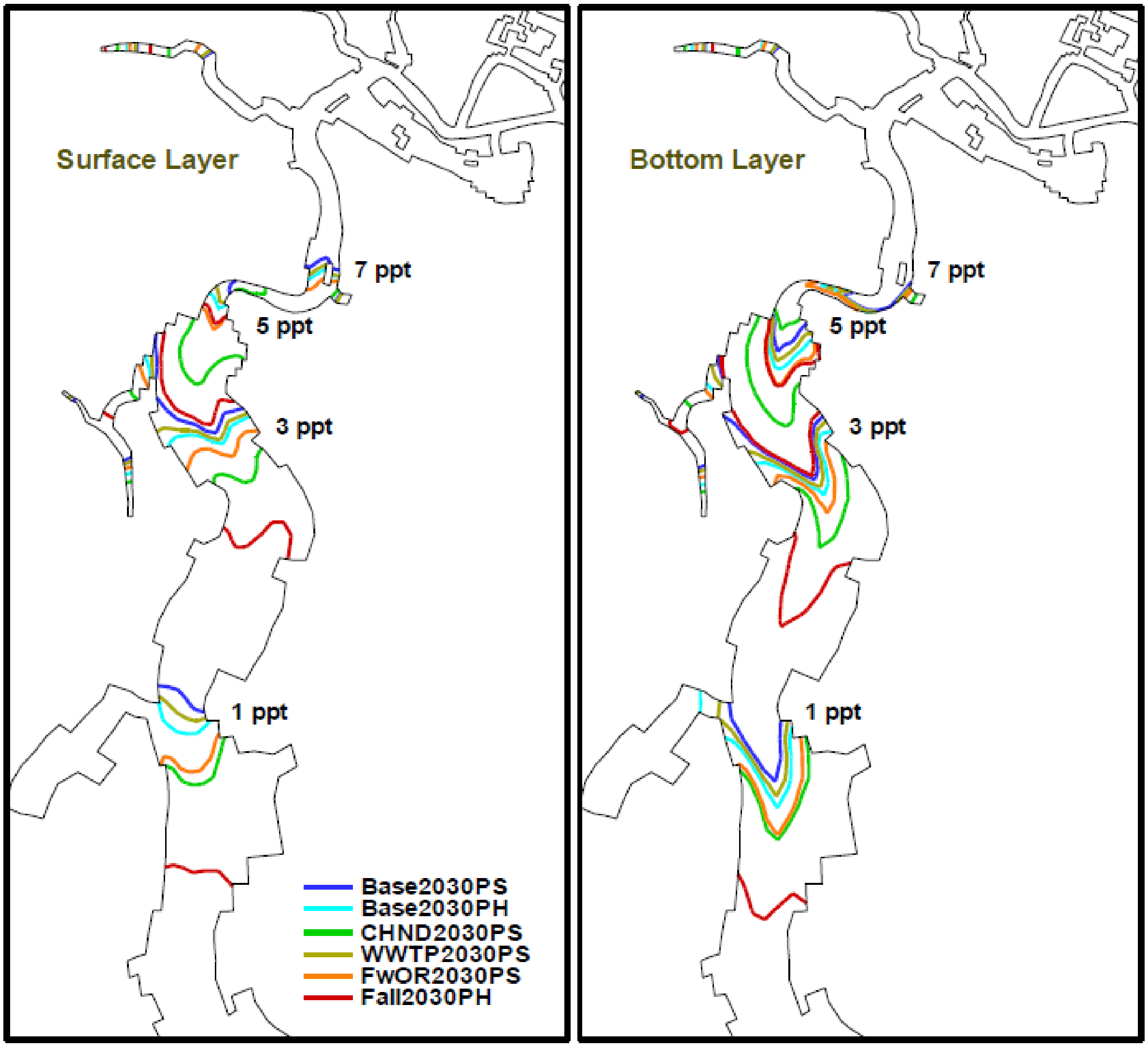


Figure 4–20. Maps illustrating upstream movement of 1, 3, 5, and 7 ppt isohalines for scenario FwOR2030PS (orange isohaline line) from the hydrodynamic model output.

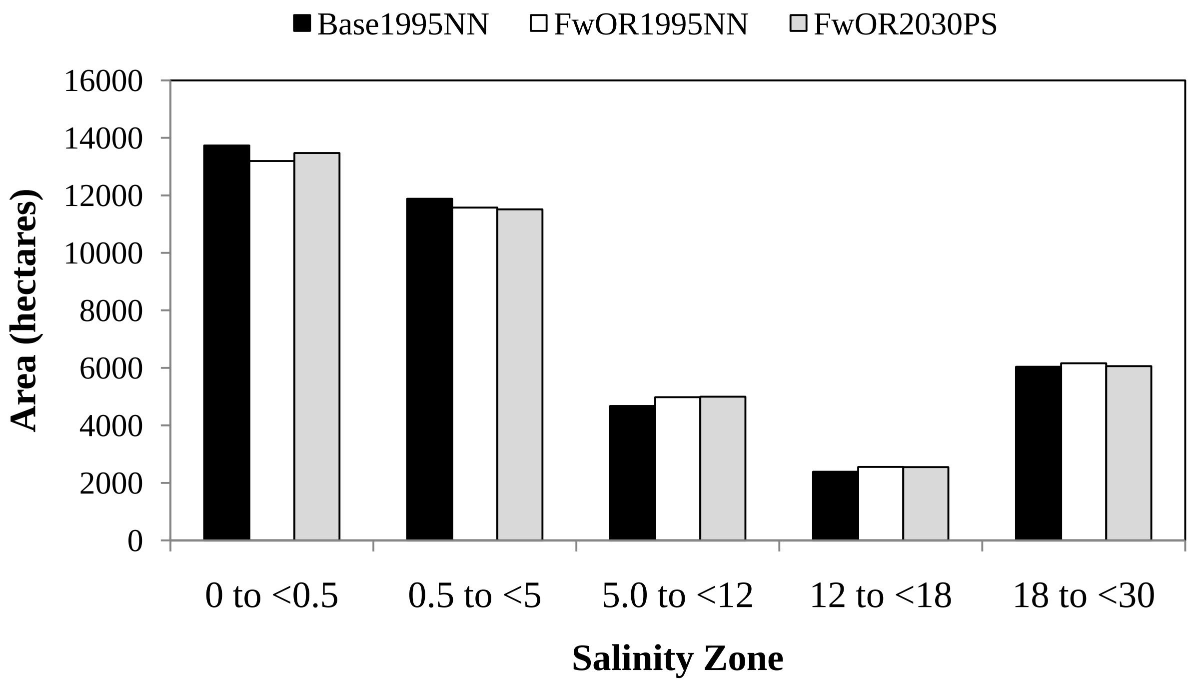


Figure 4–21. Changes in area of salinity zones defined by the Fish Working Group for the Base1995NN, FwOR1995NN, and FwOR2030PS scenarios.

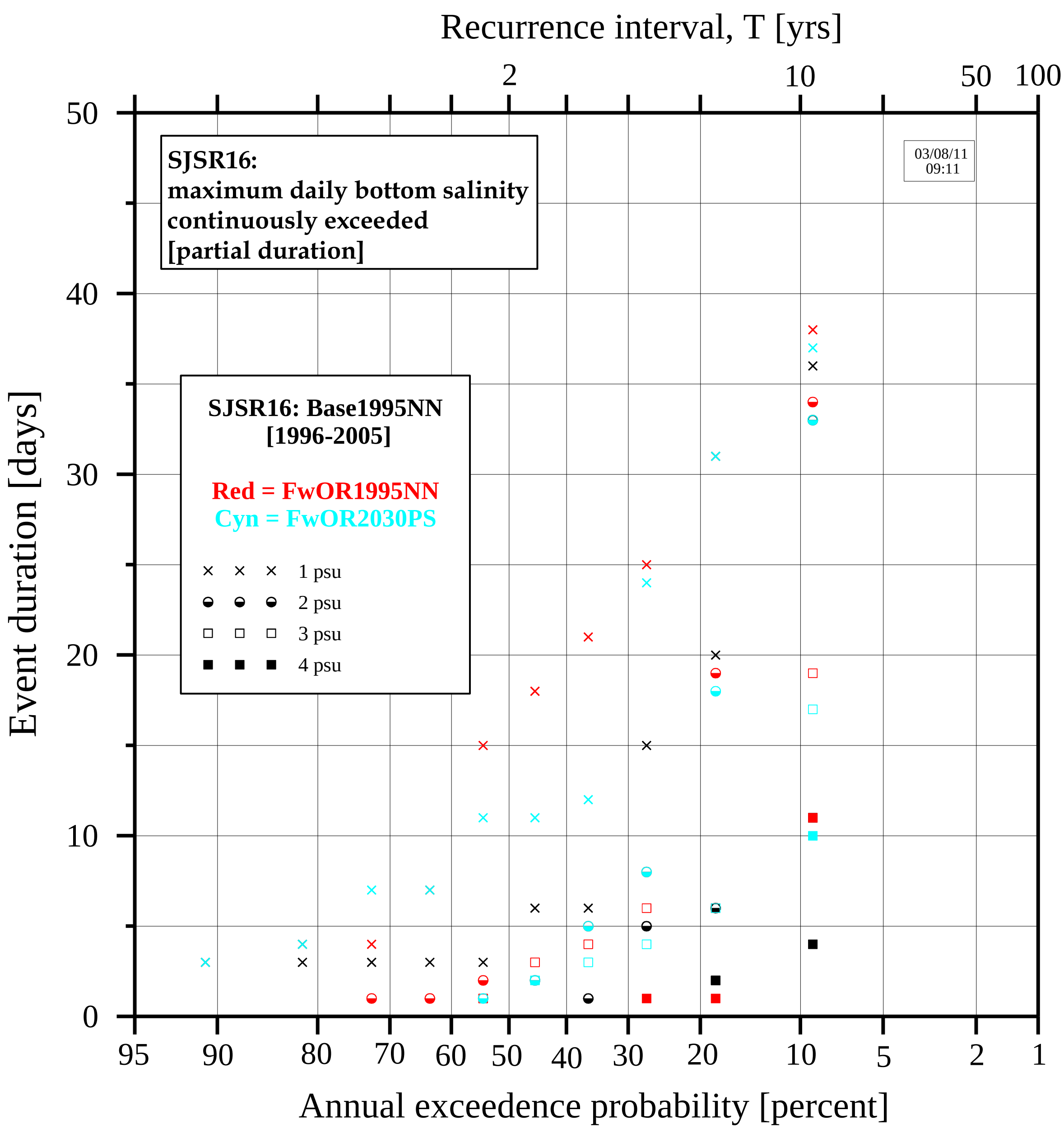


Figure 4–22. Partial duration frequency analysis of salinity at SR16 at Green Cove Springs (segment 3). Methods described in Appendix 11.G.

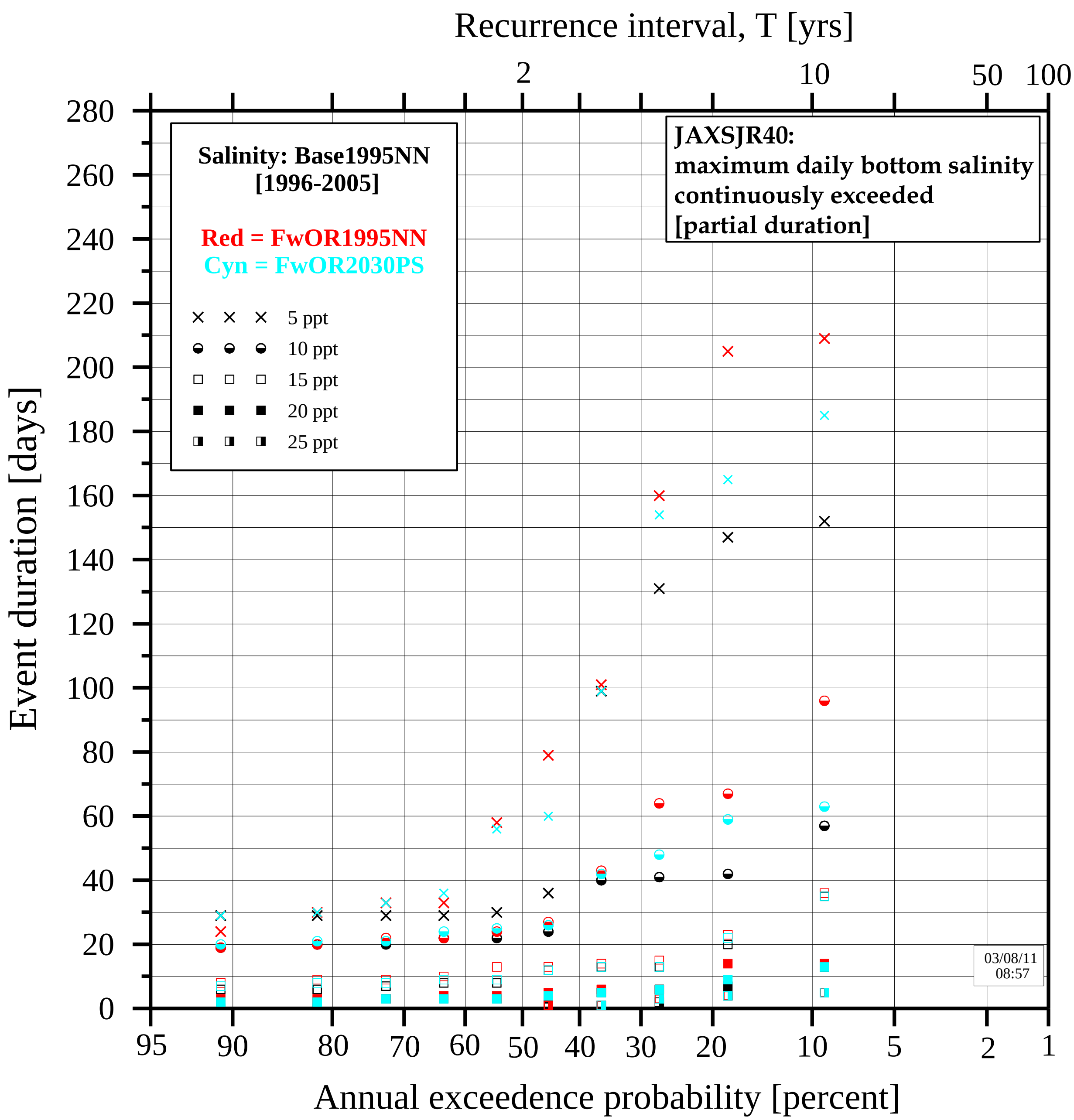


Figure 4–23. Partial duration frequency analysis of salinity at JAXSJR40 (segment 2). Methods described in Appendix 11.G.

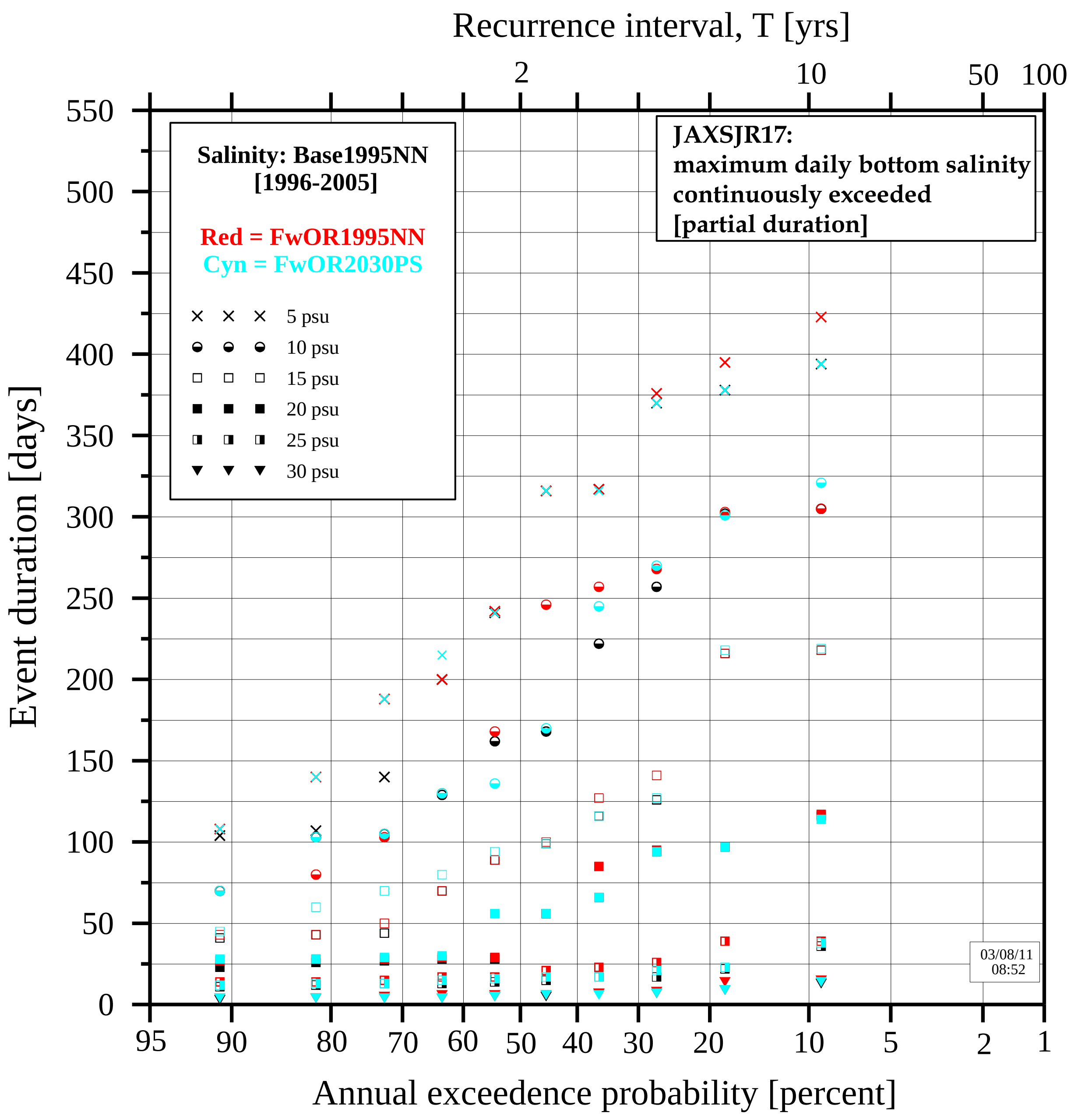


Figure 4–24. Partial duration frequency analysis of salinity at JAXSJR17 (segment 1). Methods described in Appendix 11.G.

Effects of Salinity Change on Estuarine Benthic Communities

Because overall benthic community abundance was highest in the lowest salinity zones (Figure 4–18), and because many of the benthic taxa and taxa groups examined by Montagna et al. (2011) exhibited peak abundance at or below 5 ppt, we maintain that loss of low salinity habitat (≤ 5 ppt) due to water withdrawals would be the principal concern for estuarine benthic communities. Modeling generally indicates that the largest increases in salinity will be in segment 2, where the benthic communities are already dominated by estuarine taxa with a fairly broad salinity tolerance. Thus, the effects here are projected to be minor to negligible. At the SR 16 at Green Cove Springs location, modeling indicates that higher salinities will be seen for a few days longer during droughts as a result of upstream withdrawals. Such salinity increases would affect sensitive taxa, such as aquatic insects, which decline with increasing salinity (Figure 4–19).

A spreadsheet model was constructed using the equation in Figure 4–18. This was used to evaluate changes in total benthic population density based on the projected changes in area of the different salinity zones indicated in Figure 4–21. The changes in population density were derived by applying the percent change in area of a zone (negative or positive) to the mean density in the zone. Results are shown in Figure 4–25. The population density changes mirror the area changes, as would be expected; abundance in the two lower salinity zones (0 to < 0.5 and 0.5 to < 5) declines up to 3.9%, while abundance in the three higher salinity zones (5 to < 12, 12 to < 18, and 18 to < 30) increases up to 7.1%.

Many freshwater invertebrates do exhibit a fairly wide degree of tolerance to salinity (Table 4–14; Merritt and Cummins 2008), and thus would not be greatly affected by the projected salinity increases. More sensitive taxa include mayflies (Ephemeroptera), caddisflies (Trichoptera), and stoneflies (Plecoptera). In laboratory experiments, DeSalvo (2000) found that damselfly nymphs (*Enallagma* spp.) in the lower St. Johns River could tolerate up to 20 ppt salinity with no significant effects, and that the main effect of salinity on populations of damselfly nymphs was due to loss of preferred SAV habitat. Williams and Williams (1998), also using laboratory studies, found that larvae of selected caddisfly taxa could survive 4 hrs of immersion in full-strength seawater (35 ppt).

Effects of projected changes in salinity on benthic macroinvertebrate habitat, primarily the extensive beds of SAV (principally eelgrass, *Vallisneria americana*) do not appear to be of concern. Overall loss of SAV habitat will be minimal (Chapter 9. Submersed Aquatic Vegetation). Earlier work by Montagna et al. (2008a) showed that infaunal benthic community structure will not be affected by loss of SAV habitat (Figure 4–26). Selected epifauna taxa or taxa groups would be affected by loss of SAV. Soulen (1998) demonstrated that grass shrimp in the genus *Palaemonetes* were almost exclusively associated with SAV cover, with minimal abundance in unvegetated areas (Figure 4–26). Through laboratory experiments, she attributed this to predation pressure on shrimp from largemouth bass. Other epifauna dependent upon SAV include the Conrad's false mussel (*Mytilopsis leucophaeata)*, which uses the grass blades for attachment (SJRWMD unpubl. data; and R. A. Mattson, SJRWMD pers. obs., 2009), and various aquatic insect groups (DeSalvo 2000). In the evaluated scenario, because loss of SAV is projected to be minimal, impacts to SAV-associated epifauna would likewise be minor.

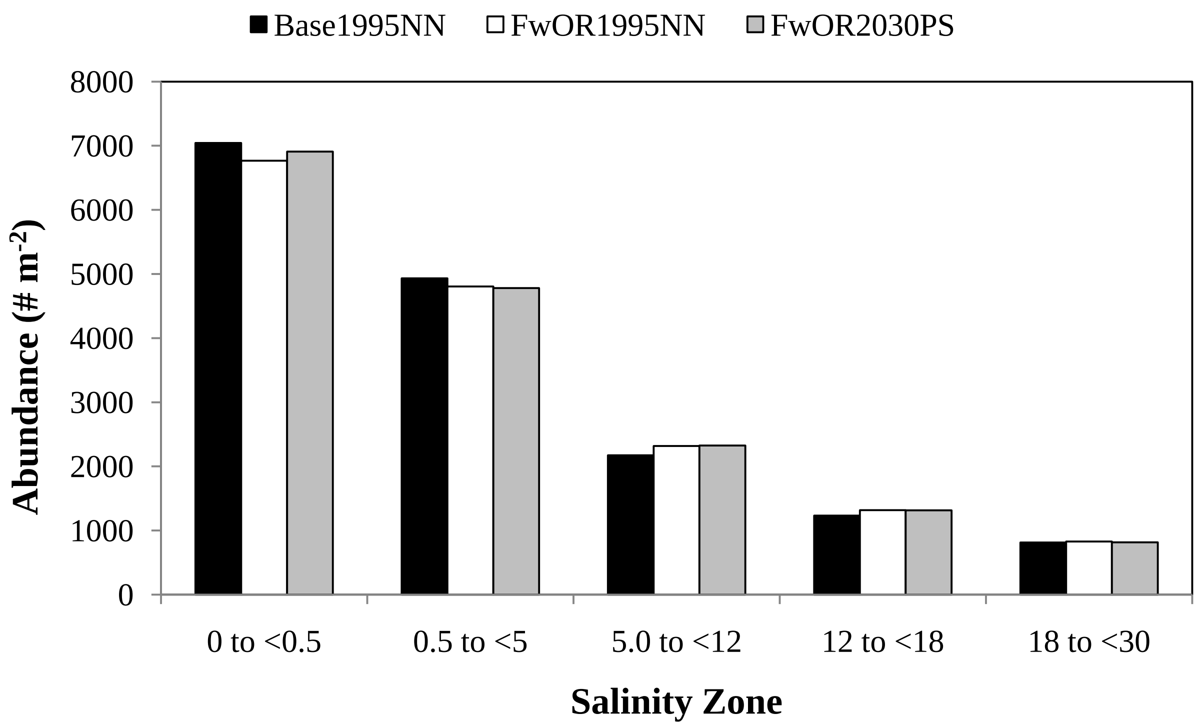


Figure 4–25. Estimated changes in mean benthic macroinvertebrate population density in the five salinity zones defined by the Fish Working Group, based on the projected changes in area indicated in Figure 4–21 and for the Base1995NN, FwOR1995NN, and FwOR2030PS scenarios.

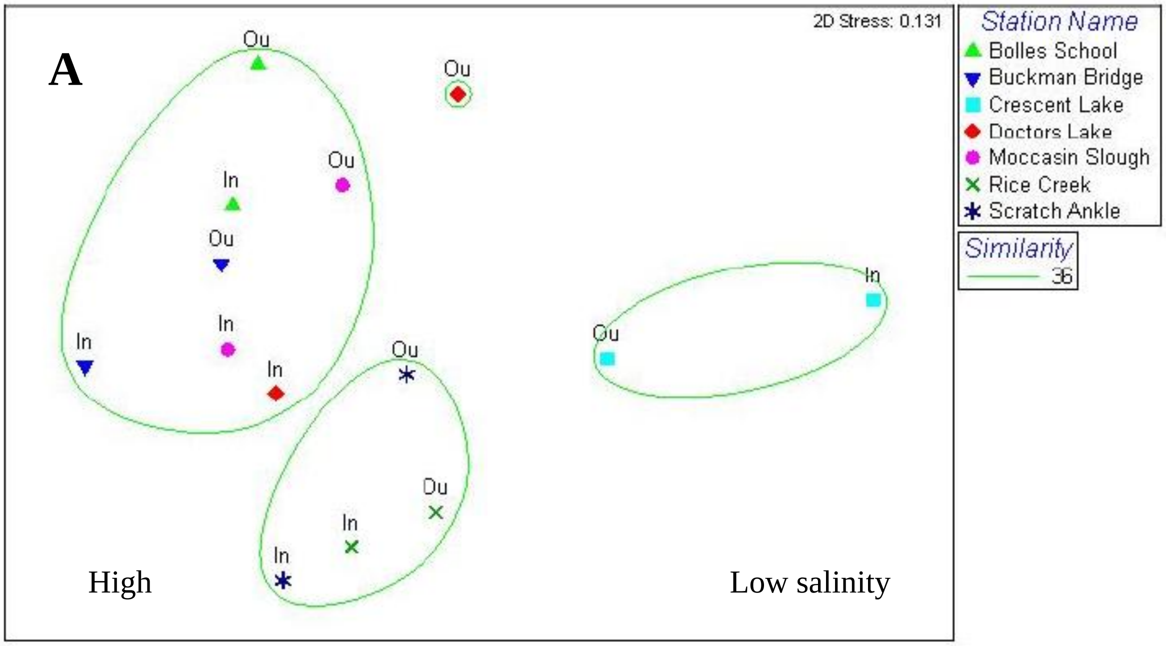
Table 4–14. Summary of salinity tolerance of aquatic insects and selected other freshwater invertebrate groups. (adapted from Merritt and Cummins 2008)

Major Taxon	Reported Ionic Concentration Range	Estimated Salinity (Concentration per 1,000)
Diptera	1,000 to > 50,000 mg L^{-1} NaCl or other salts	1 to 50 ppt
Odonata	10,000 to 45,700 mg L^{-1} NaCl or field measurements of TDS	10 to 45.7 ppt
Hemiptera	9,000 to 13,000 mg L^{-1} various salts or field measurements of TDS	9 to 13 ppt
Trichoptera	50 to 30,000 mg L^{-1} NaCl, other salts, or field measurements of TDS (most < 10,000 mg L^{-1})	0.05 to 30 ppt (most < 10 ppt)
Ephemeroptera	50 to 8,000 mg L^{-1} NaCl, marine salts, or field measurements of TDS	0.05 to 8 ppt (some reports up to 25 ppt)
Plecoptera	50 to 12,000 mg L^{-1} NaCl, marine salts, or field measurements of TDS	0.05 to 12 ppt
Coleoptera	50 to 149,000 mg L^{-1} NaCl or field measurements of TDS	0.05 to 149 ppt
Amphipoda (freshwater)	800 to 35,374 mg L^{-1} NaCl or marine salts	0.8 to 34.4 ppt
Oligochaeta (freshwater)	204 to 15,000 mg L^{-1} field measurements of TDS	0.2 to 15 ppt
Gastropoda (freshwater)	1,500 to 30,000 mg L^{-1} NaCl, other salts, or field measurements of TDS	1.5 to 30 ppt (some reports 0 to 6.8 ppt)
Decapoda (freshwater)	8,100 to 25,850 mg L^{-1} TDS field measurements	8.1 to 25.9 ppt
Isopoda (freshwater)	2,000 to 10,000 mg L^{-1} TDS field measurements	2 to 10 ppt
Hydracarina	26,530 mg L^{-1} TDS field measurement	26.5 ppt

Note:
TDS = Total Dissolved Solids

<u>Effects of Other Water Quality Change on Estuarine Benthic Communities</u>

Other than changes in salinity, reductions in DO due to reduced mixing could be the other water quality change of concern in the estuary (ECT 2008). Reduced DO affects benthic community structure and function. Diaz and Rosenberg (1995) provided an extensive review of the effects of hypoxia on marine benthic communities. They indicate that DO levels < 3 mg L^{-1} generally represent the threshold at which adverse changes in benthic communities begin to occur. Continuous monitoring of DO in the lower river at Federal Point (Table 4–6) indicated that periods of low DO (< 2 mg L^{-1}) do occur in the lower river. DO below 2 mg L^{-1} occurred 10% of the time, and DO below 5 mg L^{-1} occurred 19% of the time. DO < 3 mg L^{-1} thus occurs somewhere within this range (10% to 19%). Of the 10 most abundant taxa analyzed by Montagna et al. (2011), five may be characterized as tolerant taxa that can withstand periods of hypoxia. These include the freshwater oligochaete *L. hoffmeisteri* (fourth most abundant) and the estuarine polychaete *Streblospio benedicti* (fifth most abundant). Mason (1998) indicated that 84% of the taxa he collected in the lower river and estuary were eutrophic and/or pollution tolerant. Thus, similar to conditions in the benthic community upstream in the unvegetated river channel, the benthic communities of the estuary indicate a somewhat stressed or impaired system. ECT (2008) predicted that DO saturation would be reduced < 0.01 to 0.02 mg L^{-1} throughout the lower river and estuary by a reduction in freshwater inflow of 206 mgd (9.1 cms). Therefore, it appears DO levels in the estuary are relatively insensitive to changes in flow. Most of the benthic fauna in the river channel would not be affected by this DO reduction.



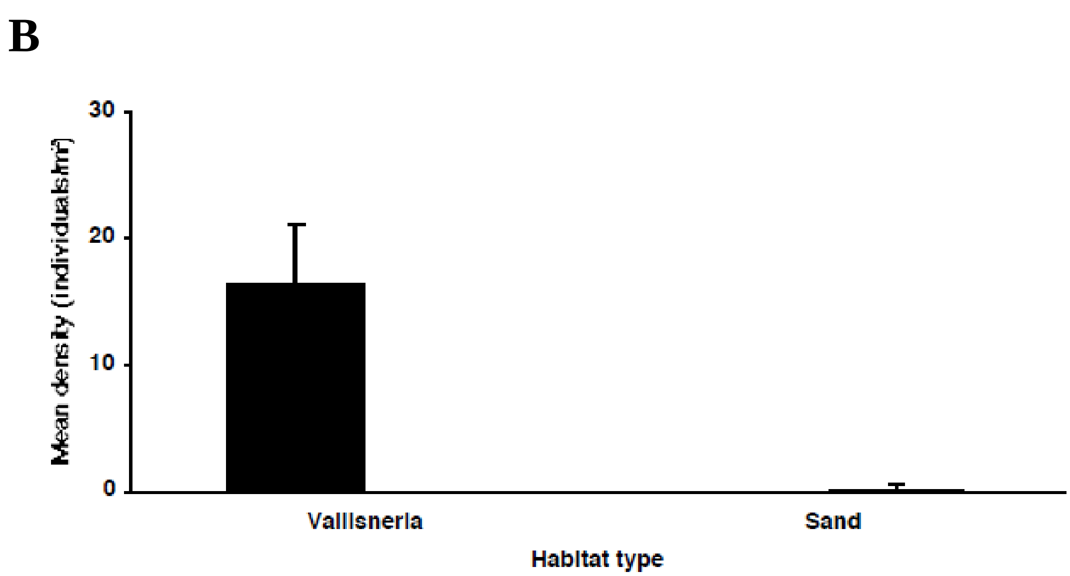


Figure 4–26. (A) Nonmetric multidimensional scaling plot of infaunal benthic community structure versus salinity and habitat; In = samples taken inside submersed aquatic vegetation (SAV) bed, Ou = samples taken outside SAV bed (adapted from Montagna et al. 2008a). (B) Plot showing abundance of *Palaemonetes* spp. in SAV and unvegetated habitats in the lower St. Johns River and estuary (adapted from Soulen 1998).

4.2.2 Effects of Hydrologic Change on Estuarine Target Taxa Populations

Blue crab

Blue crabs, *Callinectes sapidus*, are ecologically and economically important elements of the lower St. Johns River and estuary. Ecologically they are important estuarine predators and scavengers and a staple food item in the diets of many estuarine fishes (Van Den Avyle and Fowler 1984). Blue crabs are a major commercial fishery in the lower St. Johns River and have been for decades (Tagatz 1968). Blue crab commercial landings in the four counties bordering the St. Johns estuary (Table 4–15) totaled more than 800,000 pounds of hardshell crab in 2009 (the most recent year for which final data are available). This was about 17% of total state landings. The estimated dockside value of these landings was about $1.04 million. Total softshell blue crab landings in these four counties (23,713 lbs) constituted about 30% of the total statewide landings in 2009, with a value of $180,456 (Table 4–15). Appendix 11.H presents a literature review of the relationships between salinity and life history characteristics of blue crabs.

Effects of Predicted Changes in Salinity and River Inflow on Blue Crab

Surface water withdrawals have the potential to affect blue crab populations in two ways: alteration of spatial and temporal salinity patterns and/or alteration of critical habitats (especially SAV) or food resources. Evaluation of the potential for effects also requires consideration of the complex life history of the species, with both planktonic (larval) and benthic (juvenile and adult) phases.

A detailed review of salinity effects on blue crabs is presented in Appendix 11.H. Many life stages of this crab exhibit a fairly wide degree of salinity tolerance, particularly juvenile and adult crabs, which are the main benthic life stages that occur within the river mouth in the lower river and estuary. Based on this review of the literature, in conjunction with the EFDC model-predicted salinity changes described above, salinity changes due to upstream water withdrawals will be unlikely to have an adverse impact on populations of blue crab in the St. Johns River (Appendix 11.H).

Table 4–15. Blue crab landings (hard- and softshell) in 2009 in the four counties bordering the St. Johns River estuary.

County	2009 Landings (lbs)	Dockside Value
	Hardshell crab	**Hardshell crab**
Clay	42,920	$52,362
Duval*	443,584	$541,173
Putnam	131,433	$160,348
St. Johns*	234,966	$286,658
Total Hardshell	852,903	$1,040,541
	Softshell crab	**Softshell crab**
Clay	4,927	$37,494
Duval*	692	$5,266
Putnam	6,137	$46,703
St. Johns*	11,957	$90,993
Total Softshell	23,713	$180,456

Source: FWC Fish and Wildlife Research Institute.
* Some of the reported landings may come from the Atlantic coast of the county, rather than the St. Johns River estuary.

The FWC supplemented the data of MacDonald et al. (2009) with two additional years of data from the Fisheries Independent Monitoring Program. Detailed descriptions of the analysis of these data are presented in Chapter 12, Fish, but a brief summary is presented here. The data for various size classes of the finfish and shellfish species collected (pseudospecies), for the various gear types, were screened using the nonparametric Spearman's rho for relationships between abundance and lagged freshwater inflow. Only statistically significant relationships were retained for analysis. The data from these were appropriately transformed, and regression relationships were determined between abundance and inflow. Only regressions with an r^2 of > 0.25 were retained for further analysis. These were entered into an SAS program developed to evaluate the effects of various upstream withdrawal scenarios on population abundance. Results are presented in Table 4–16. Comparison of the base scenario (Base1995NN) with the worst case full withdrawal test scenario (FwOR1995NN) indicates that blue crab abundance would increase, thus, no adverse effects on blue crab are expected from the upstream withdrawals. Future scenarios that address increased freshwater inflows to the St. Johns River estuary (from the USJRB Projects and 2030 land use) result in reduced blue crab abundance (Table 4–16; FwOR2030PS), possibly due to downstream movement of areas of preferred salinity into areas of less-than-desirable habitat (downtown Jacksonville and the Port of Jacksonville). Imposition of withdrawals on these future conditions results in increased crab abundance (Table 4–16).

Other Factors in Blue Crab Abundance

Blue crab abundance varied with bottom habitat type, with higher abundance in Some oyster and Other habitat categories in 21.3 m seine collections (MacDonald et al. 2009, Figure 40). Higher abundance was found in areas with no vegetation versus Some vegetation in these seine collections. In the 183 m seine collections, higher crab abundance was found in the Other habitat category and higher abundance in Some vegetation versus none. McMichael and Tsou (2003) found that habitat was a significant explanatory factor in abundance of blue crab in the Suwannee estuary, with tidal creek habitat appearing to be important. Results of MacDonald et al. (2009) somewhat corroborate this in that they found higher crab abundance in their backwater habitats, which are largely tidal creeks. Model-predicted salinity changes in areas of tidal creeks and SAV habitat (Chapter 6. River Hydrodynamics Results; Chapter 9. Submersed Aquatic Vegetation) appear to be minor and would not affect blue crab habitat or populations.

Tagatz (1968) studied food habits of blue crab in the estuary, and Laughlin (1982) studied the feeding habits of blue crab in Apalachicola Bay. Both studies found that bivalves were an important food source, along with various small crustaceans (shrimp, amphipods and small crabs) and plant material and organic detritus. Younger crabs consumed a greater volume of plant and detrital material in both studies, and both studies showed that a considerable degree of spatial and temporal variability in diet was seen in both estuaries. Blue crabs appear to exhibit a high degree of plasticity in their diets, driven by food resource availability (Laughlin 1982). This diet plasticity may compensate for the subtle changes in benthic macroinvertebrate communities that may accompany salinity alterations driven by flow reductions.

Penaeid Shrimp

Three species of shrimp in the family Penaeidae occur in the lower St. Johns River and estuary: white shrimp (*Litopenaeus setiferus*), brown shrimp (*Farfantepenaeus aztecus*), and pink shrimp (*Farfantepenaeus duorarum*). Of the three, white shrimp are most abundant in this northeast Florida region, and pink shrimp least abundant (Joyce 1965). The ecological importance of shrimp in the St. Johns estuary stems from their abundance and role as key consumers of plant material and small benthic fauna and as important food items in the diets of many estuarine fishes. Their economic importance stems from their value in commercial and recreational fisheries and their importance as key food items in the diets of many finfish of high recreational interest in the St. Johns estuary, mostly sciaenids (e.g., red and black drum, spotted sea trout, croaker).

The commercial shrimp fishery in the St. Johns estuary harvests penaeid shrimp both for food and for bait for recreational fishing. There is also a productive recreational fishery for white shrimp (primarily using cast nets) in the lower St. Johns River during the late summer. The landings from this recreational sector are not quantified in Florida, but surveys in other southeastern states suggests it could be substantial (Muncy 1984). Commercial landings of penaeid shrimp for food and bait are summarized in Table 4–17 for 2009 (the most recent year for which final landings data are available) for Duval County. It is assumed that the majority (if not all) of the shrimp landings in this county are based on populations supported by the St. Johns River. Joyce (1965) noted that “. . . the St. Johns River system is probably the most important single geographical feature affecting the shrimp populations of the northeast coast of Florida.” Therefore, this assumption appears reasonable. More than 2.5 million pounds of shrimp (all three species) were harvested for food in Duval County in 2009, with a dockside value of over $4.8 million. White shrimp accounted for 84% of the total food shrimp landings (Table 4–17). Again, it is likely that most of this harvest was dependent on the habitats and freshwater flows of the St. Johns River. A review of the literature on the direct and indirect effects of salinity and flow on penaeid shrimp populations is presented in Appendix 11.H.

Effects of Predicted Changes in Salinity and River Inflow on Penaeid Shrimp

Analysis of shrimp abundance and distribution versus lagged freshwater inflow using the results of MacDonald et al. (2009) plus two additional years of data collected in the Fisheries

Independent Monitoring program indicated no significant relationships between monthly or annual shrimp abundance and freshwater inflow. There was a significant effect of flow on distribution of the center of abundance (Table 4–16) with the center of abundance moving upstream with lower flows and higher salinities and generally moving downstream with higher flows and lower salinities due to land use changes and the addition of the upper basin projects. White shrimp populations are not expected to be affected by the predicted alterations in freshwater inflow, because the salinity zones they prefer will not be reduced in size, but rather move spatially within the estuary. Consideration of the salinity tolerances of various life stages of white shrimp (Appendix 11.H) likewise concluded that the predicted salinity alterations will be unlikely to have adverse effects on shrimp populations.

Table 4–16. Results of regression analyses relating monthly abundance of blue crab and median center of abundance of white shrimp versus lagged freshwater inflow under various scenarios of withdrawal, land use, upper basin project restoration, and sea level rise. All scenario comparisons are versus base scenario (Base1995NN).

Callinectes sapidus								
Response type	**Gear Type (time period)**	**Lag (days)**	**r^2 of regression**	**Base median center of abundance (river km)**	**FwOR1995NN (% change from base)**	**FwOR1995PN (% change from base)**	**Base2030PS (% change from base)**	**FwOR2030PS (% change from base)**
Monthly (trip) abundance	183 m seine (April to October)	180	0.256	—	+24.92	+22.67	-11.50	+9.20
Monthly (trip) abundance	6.1 m trawl (June to December)	180	0.469	—	+15.11	+13.15	-12.08	+2.60
Litopenaeus setiferus								
Response Type	**Gear Type**	**Lag (days)**	**r^2 of regression**	**Base median center of abundance (river km)**	**FwOR1995NN (Δ km)**	**FwOR1995PN (Δ km)**	**Base2030PS (Δ km)**	**FwOR2030PS (Δ km)**
Center of abundance	21.3 m seine (August to November)	30	0.481	34.92	+1.25	+1.09	-1.53	-0.26
Center of abundance	21.3 m seine (June to July)	90	0.657	32.59	+3.75	+3.22	-2.11	+1.64
Center of abundance	6.1 m trawl (June to September)	30	0.417	57.96	+3.24	+2.77	-2.63	+0.49

Note:
+ = % increase in abundance or movement upstream in km
- = % decrease in abundance or movement downstream in km

Table 4–17. Penaeid shrimp landings in 2009 in Duval County.

Shrimp	2009 Landings (lbs)	Dockside Value
White shrimp	2,132,720	$4,222,786
Brown shrimp	416,669	$629,170
Pink shrimp	3,707	$6,895
Total Food Shrimp	2,553,096	$4,858,851
*Total Bait Shrimp	16,419	$70,930

Source: FWC Fish and Wildlife Research Institute 2011
* Bait shrimp are reported collectively, not broken out by species

4.2.3 Characterization of Key Effects for Estuarine Macroinvertebrate Communities

Freshwater inflow and related changes in salinity influence benthic macroinvertebrate community structure and function in estuaries (Browder 1991; Bulger et al. 1993; Montagna et al. 2008a). Changes in salinity are associated with changes in benthic community structure (taxa composition and abundance) in the St. Johns River estuary (see Figure 4–17 and Figure 4–18) and in other estuaries in Florida (Montagna et al. 2008b). As with the freshwater reach, the critical question is: will the predicted changes in salinity create changes in the benthic community in ways and magnitudes sufficient to cause management concern?

Table 4–18 summarizes the predicted benthic community changes that would occur due to reduced freshwater inflow (and increased salinity upstream). We used the same five descriptors of level of effect (negligible, minor, etc.) for estuarine benthic communities as was used for freshwater benthic communities. Relatively small increases in salinity are predicted to occur in segments 1 and 2 for the evaluated scenario (Chapter 6. River Hydrodynamics Results). These alterations will most likely have negligible effects on benthic communities in these segments because most of the benthic community species at these sites are already tolerant of moderate and variable levels of salinity. In these segments, changes in DO caused by flow reductions are predicted to be slight (ECT 2008), and would not affect benthic communities in these segments. For the evaluated scenario (FwOR2030PS), our assessment of the potential effects is that they would be negligible to minor (Table 4–19). The upper reaches of segment 3 will not experience alterations in salinity, but the lower reaches of this segment will see slightly higher salinities during low flow and drought events (see Figure 4–22). Reductions in populations of sensitive freshwater taxa (e.g., insects, freshwater mollusks) (see Figure 4–18 and Figure 4–19) would likely occur in these areas during these periods of elevated salinity. Overall, the area affected would be small (see Figure 4–21), and many of the taxa affected are short life cycle species (e.g., chironomid midges and oligochaete worms) that would quickly recover following cessation of the drought.

Table 4–18. Summary of predicted effects on benthic communities in segments of the lower St. Johns River and estuary with reduced freshwater inflow under scenario FwOR2030PS.

Segment	Physical and Chemical Changes	Biological Changes
Segment 3	• Smaller salinity increase on average annual basis (Chapter 6. River Hydrodynamics Results; Figure 4–20) • Larger increases in salinity during droughts in lower reaches (Figure 4–21	• Minimal changes in benthic communities or populations (Figure 4–18; Figure 4–19; Figure 4–25.) • Some changes in benthic community structure (Figure 4–17) and abundance (Figure 4–18); reduction in abundance of populations sensitive to increased salinity (Figure 4–19; Table 4–13 and Table 4–14) • No effects on infauna due to loss of SAV (Figure 4–26A); loss of SAV dependent epifauna (Figure 4–26B) • No effects on white shrimp populations; no adverse effects on blue crab abundance (Table 4–16)
	• Slight decreases in DO concentration and saturation (ECT 2008)	• Negligible or minor changes in community structure due to loss of very sensitive (likely rare) taxa (Diaz and Rosenberg 1995)
Segment 2	• Smaller salinity increase on average annual basis (Chapter 6. River Hydrodynamics Results; Figure 4–20) • Larger increases in salinity during droughts	• No or very slight changes in benthic communities or populations (inferred due to existing composition of the benthic community – Appendix 11.A) • Minimal changes in benthic community structure due to most benthic organisms being tolerant of salinity • No effects on infauna due to loss of SAV (Figure 4–26A); loss of SAV dependent epifauna (Figure 4–26B) • No effects on white shrimp populations; no adverse effects on blue crab abundance (Table 4–16)
	• Slight decreases in DO concentration and saturation (ECT 2008)	• Negligible or minor changes in community structure due to loss of very sensitive (likely rare) taxa (Diaz and Rosenberg 1995)
Segment 1	• Minimal increases in salinity across most flow conditions (Chapter 6. River Hydrodynamics; Figure 4–20)	• No or very slight changes in benthic communities or populations (inferred due to existing composition of the benthic community – see Appendix 11.A) • No effects on white shrimp populations; no adverse effects on blue crab abundance (Table 4–16)
	• Slight decreases in DO concentration and saturation (ECT 2008)	• Negligible or minor changes in community structure due to loss of very sensitive (likely rare) taxa (Diaz and Rosenberg 1995)

Note:
DO = Dissolved oxygen
SAV = Submersed aquatic vegetation

4.2.4 Characterization of Uncertainty for Estuarine Macroinvertebrate Communities

Because we had more benthic data to work with in the lower river and estuary, levels of uncertainty are reduced due to the ability to develop reasonable quantitative relationships between salinity and benthic communities and populations (see Figure 4–17 and Figure 4–18) and due to good agreement in the scientific literature on the mechanistic relationships between salinity and benthic community structure (Montagna et al. 2008a; Kinne 1971). Table 4–19 indicates the levels of uncertainty associated with effects on freshwater and estuarine benthic communities in the lower river and estuary. Most sources of uncertainty are from limitations on the data collected from a segment (e.g., segment 1 had relatively little existing data), or from the moderate levels of certainty associated with the predictive models (e.g., Figure 4–18, r^2=0.55). Overall, the levels of uncertainty associated with assessment of effects of benthic communities of the lower river and estuary are low to medium due to the development of moderately strong predictive models, strong supporting evidence from the literature, and good understanding of mechanisms.

4.2.5 Potential Means of Mitigation of Effects for Estuarine Macroinvertebrate Communities

The main way to mitigate the effects of altered salinities due to freshwater inflow reductions would be to implement some type of system to scale withdrawal volumes to ambient flows, with reduced rates of withdrawal at critically lower flows. This approach is used by the Southwest Florida Water Management District (Flannery et al. 2002) and is specifically designed to protect ecological structure and function in estuaries.

4.2.6 Monitoring Needed for Adaptive Management for Estuarine Macroinvertebrate Communities

As with the upstream, freshwater benthic communities, establishment of an ongoing, long-term program of monitoring of benthic communities of the lower river and estuary will generate a valuable set of data to evaluate the long-term effects of salinity and freshwater inflows. This was shown by the data set generated and used by the FWC Fisheries Independent Monitoring Program, which should be maintained in its current form to continue monitoring important finfish and shellfish populations in the lower river and estuary.

Establishment of monitoring at those sites studied by Cichra and Adicks (1998) and Mason (1998), along with two or three other sites further downstream between Jacksonville and the river mouth, will enable us to include the historical data collected at these sites, along with the data from nearby sites sampled by FDEP. One important consideration for any benthic monitoring in the lower river is to concurrently monitor water quality and sediment along with the collection of benthic community samples. This was not done in any prior effort except for the one time EMAP-E effort in 2000. Lack of concurrent physical-chemical data limits the usefulness of benthic community data for evaluating the effects of environmental change.

Table 4–19. Summary characterization of effects of withdrawals on estuarine benthic communities in the lower river and estuary under scenarios FwOR1995NN and FwOR2030PS.

River Segment	Δ EST Benthic Community 1995	Δ EST Benthic Populations 1995	Δ EST Target Taxa Populations 1995	Δ EST Benthic Community 2030	Δ EST Benthic Populations 2030	Δ EST Target Taxa Populations 2030	Overall
1	**	**	**	**	**	**	**
2	**	**	***	**	**	***	***
3	**	**	***	**	**	***	***
4	NA	NA	NA	NA	NA	NA	NA
5	NA	NA	NA	NA	NA	NA	NA
6	NA	NA	NA	NA	NA	NA	NA
7	NA	NA	NA	NA	NA	NA	NA
8	NA	NA	NA	NA	NA	NA	NA

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

Note:
NA = not applicable to these river segments.

4.2.7 OTHER CONSIDERATIONS FOR ESTUARINE MACROINVERTEBRATE COMMUNITIES

Effects of Sediment Composition on Benthic Communities

It is well established that the quality and composition of the bottom sediment is a major influence on benthic community structure and function (Rhoads 1974; Day et al. 1989; Mannino and Montagna 1997; Kennish et al. 2004). Evaluation of the effects of sediment composition on benthic community structure in this study was hampered by the lack of quantitative sediment data collected concurrently with the benthic invertebrate samples. Although the SJRWMD has an extensive network of sediment quality sites in much of the lower St. Johns River and estuary, none of these sites were concurrent with or even adjacent to the benthic community sites used in this analysis. Montagna et al. (2011) compared the SJRWMD sediment network (300 sampling sites) with the benthic sites they analyzed and found that few of the sediment sites came even within 1 km of the benthic data sites. They used kriging to generate a generalized map of sediment composition on the lower river and estuary (Figure 8 in Montagna et al. 2011) and compared it with the locations of the benthic community sites used in their ecological analyses. Some of the changes in benthic community structure could be associated with a gradual decrease in % silt and increase in % sand in upstream reaches of segment 3. Sediment data were lacking in the lowest reaches of the river (essentially no sediment data in the river main stem, segment 1), which further limited the ability to derive inferences about the effects of sediment on benthic communities.

Effects on Meroplankton

Another issue that was not considered in this study was the potential effects of the EFDC model-predicted salinity changes on the meroplanktonic larvae of many estuarine benthic macroinvertebrates. In part, this was due to the lack of data on the distribution and abundance of larger zooplankters such as zoea and megalops larvae of crabs, veliger larvae of mollusks, polychaete larvae, and the planktonic larval phases of penaeid shrimp. SJRWMD operates a zooplankton monitoring program in the lower river, but this consists of collection of vertically integrated water samples with a 5-cm diameter length of pipe. This mainly collects microzooplankters, such as cladocerans, copepods, and rotifers. The larger, more motile forms are missed by this method.

Because the meroplankton are somewhat motile and can essentially move with a mass of water of preferred salinity, it may not be the salinity changes that affect meroplankton, but the change in area or volume of the zones of preferred salinity. Peebles et al. (1991) found negative relationships between preceding 3-day average flow and abundance of polychaete, bivalve, and decapods larvae in the Little Manatee River estuary in Tampa Bay, indicating reduced abundance at higher flows. This reduction appeared to be due to a downstream movement of the center of abundance of these meroplankton at increased flows. Polychaete larvae were generally most abundant in the bay, outside the river mouth, decapod larvae were most abundant at the river mouth, and bivalve larvae were most abundant inside the river about 4 km above the mouth. Peebles et al. (1991) suggested that a limiting factor for meroplankton at low flows might be limits on the amount of habitat at a particular salinity in terms of the volume ($m^3 \times 10^6$) of habitat, and that relatively small reductions in freshwater inflow might result in large reductions in available habitat volume.

A final issue with effects of salinity changes on meroplankton could be changes in the salinity cues or hydrodynamic circulation patterns used to initiate vertical migration to take advantage of inflowing or outflowing currents (Day et al. 1989; Epifanio and Garvine 2001). This could affect abundance and recruitment of larvae to the estuary. The lack of data on meroplankton abundance and distribution in the estuary precludes consideration of this potential effect. This would be an area for additional study and monitoring.

Effects of Benthic Community Changes on Fish Populations

Benthic invertebrates are known to be important components of the diets of many estuarine fish (Day et al. 1989; Livingston 2003). Changes in benthic invertebrate communities could alter food resources for particular estuarine fish taxa. Estuarine fish diets have been shown to be highly variable and influenced by resource availability (Day et al. 1989; Livingston 2003). Fish feeding habits have been shown to be influenced by ontogeny (life stage), salinity gradients, environmental variability (spatial and temporal), and habitat (Livingston 2003). Subtle alterations in benthic community structure may not be a major consequence to fish populations of the estuary, as they may take advantage of the food resources available (e.g., Figure 2–2 for largemouth bass in the lower river). This is another issue that would benefit from further study and monitoring.

5 Summary and Conclusions

5.1 Freshwater Macroinvertebrate Communities

Because of lack of existing data, a short-term field study was conducted in July and November 2009 comparing the benthic macroinvertebrate communities in bulrush and spatterdock habitats in Lake Poinsett (upper basin), Lake Monroe and the St. Johns River near Yankee Lake (middle basin). Some samples were also collected in shoreline and floodplain emergent wetlands in both lakes and in hydrilla habitat in Lake Monroe. Water levels declined (0.75 m) between July and November at Lake Poinsett but remained mostly stable at Lake Monroe and St. Johns River near Yankee Lake. Water quality in both of the lakes was similar based on long-term monitoring data.

A number of metrics describing the benthic macroinvertebrate communities were calculated, including taxa richness, percent composition of various taxa, functional feeding group status, and diversity. High variability among habitats and sampling dates precluded detection of patterns in the benthic community that could be related to hydrology or other environmental drivers.

Samples collected in various shoreline and floodplain wetland habitats in Lake Poinsett in July 2009 displayed patterns that could be related to hydrology; habitats inundated for longer durations had lower overall taxa richness (possibly related to higher predation rates), but had higher percentages of long-lived invertebrate taxa (e.g., Odonata) and those indicative of more permanent aquatic habitat (e.g., Amphipoda, Ephemeroptera). Habitats inundated for shorter periods had higher percentages of more motile taxa (Coleoptera, Diptera) and those with shorter duration life cycles (Oligochaeta, Diptera).

Model scenarios that looked only at withdrawal (those ending in "NN") generally predicted declines in water level across the entire hydrologic regime. These scenarios would be expected to have more detrimental effects on benthic communities due to loss of aquatic habitat and declines in aquatic macroinvertebrate production. Addition of the upper basin projects, land use changes projected to 2030, and sea level rise ("2030PN" and "2030PS" scenarios) all reduced the decline in water level or resulted in increases in water level over current conditions. These changes would be expected to reduce and mitigate detrimental effects on benthic communities or possibly have a positive effect.

Shoreline and floodplain wetland habitats with a duration of inundation of ≥ 9 months supported benthic communities with larger proportions of invertebrate taxa indicating more permanent aquatic habitat. Analyses by the Wetland Vegetation Working Group indicated that these wetland habitats would be minimally affected (in terms of acreage experiencing reduced inundation) by the proposed water withdrawals, with corresponding minor to negligible effects on the benthic communities.

Any changes in water quality due to reduced flows (particularly reductions in DO) are not expected to have detrimental effects on in-channel benthic communities because many of the taxa are already adapted to low DO conditions. Changes in water quality are not expected in shallow shoreline habitats.

Two target taxa of freshwater invertebrates were evaluated: crayfish and the apple snail. These are known to be important food items for many key species of fish and wildlife. Modeled withdrawals would result in some reduction of preferred marsh habitats of these taxa at lower lake and river levels, but these effects were judged to have minor effects on populations of these taxa. Addition of the upper basin projects and 2030 land use changes ("1995PN" and "2030PS" scenarios) would decrease the number of critical drying events for the target taxa, which would have negligible or possibly positive effects.

Overall projected impacts of the proposed surface water withdrawals on freshwater benthic communities are characterized as minor to negligible, with projected future hydrologic conditions (upper basin projects, land use changes) reducing detrimental impacts on benthic communities to largely negligible. Because of data limitations, uncertainty for these conclusions is characterized as high.

5.2 Estuarine Macroinvertebrate Communities

A greater amount of existing data in the lower St. Johns River and estuary permitted quantitative analysis of benthic communities and populations in relation to salinity. Benthic infaunal data sets from the FDEP and USEPA were analyzed with multivariate and regression tools, looking at community composition and abundance.

Two target taxa were examined — blue crab and penaeid shrimp (primarily white shrimp). Both of these support important commercial and recreational fisheries in the St. Johns River estuary. Juveniles of these have been sampled for approximately 9 years by the FWC Fisheries Independent Monitoring Program.

Salinity influenced benthic community structure and appeared to be the strongest driver affecting community structure (as opposed to other water chemistry variables and habitat). The abundance of populations of various numerically dominant macroinvertebrate taxa exhibited differing responses to increases in salinity. Populations of nine taxa (e.g., aquatic insects) decreased in abundance with increasing salinity; abundance of six marine taxa (e.g., echinoderms, phoronids) increased with increasing salinity; 14 taxa displayed peak abundance at some optimum salinity; and abundance of 19 taxa exhibited no statistically significant relationship with salinity.

Upstream withdrawals from the St. Johns and Ocklawaha rivers (FwOR1995NN) resulted in upstream movement of higher salinities. Land use changes projected to 2030, addition of the upper basin projects, and projected sea level rise (FwOR2030PS) still exhibit upstream salinity increases in response to withdrawals, in many cases not much different from the scenario that only looked at withdrawal effects (FwOR1995NN). Withdrawals from the Ocklawaha River appear to have more of an effect on estuarine salinity than upstream St. Johns River withdrawals.

Projected salinity changes in the lower reaches of the estuary (segments 2 and 1) are expected to have negligible effects on benthic communities because they are largely composed of taxa well adapted to salinity. Encroachment of higher salinities in upper reaches (segment 3) during droughts may have temporary effects on sensitive taxa (freshwater insects and mollusks), many of which should be able to recover following the return of more normal river flows. Loss of low-salinity habitat (mean salinities $\leq$ 5 ppt) will have the most pronounced effects on benthic communities of the estuary, but loss of this habitat is projected to be minimal.

Loss of SAV habitat would have no effect on benthic infauna populations, but would have detrimental effects on epifauna that are dependent on SAV habitat. Loss of SAV is projected to be minor to negligible, with corresponding effects on SAV-associated benthic communities.

Any changes in other water quality variables (primarily DO) will have minimal or negligible changes on benthic communities in the unvegetated river channel bottom because many of the taxa are adapted to hypoxic conditions.

Analysis of freshwater inflow and abundance of white shrimp indicated that populations of this target taxon appear to be insensitive to flow changes, and thus would be unaffected by the projected changes due to withdrawal. Blue crab abundance overall appeared to also be unaffected by the projected withdrawal changes.

Overall projected impacts of the proposed surface water withdrawals on estuarine benthic communities are characterized as minor to negligible, with projected future hydrologic conditions (addition of upper basin projects, land use changes) reducing detrimental impacts on benthic communities. Effects on white shrimp are characterized as negligible, while effects on blue crab are characterized as moderate due to the projected changes in relative abundance as a result of flow alterations. Because of uncertainties in the biological data, uncertainty for these conclusions is characterized as moderate.

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8 APPENDICES (SEPARATE DOCUMENT)

APPENDIX 11.A. DESCRIPTION OF BENTHIC COMMUNITIES OF THE ST. JOHNS RIVER

APPENDIX 11.B. GEOGRAPHIC COORDINATES OF THE SITES SAMPLED IN THE 2009 FIELD STUDY

APPENDIX 11.C. MINITAB PLOTS OF METRICS CALCULATED FROM THE DATA OF THE 2009 FIELD STUDY

APPENDIX 11.D. MERRITT AND CUMMINS FINAL REPORT FROM 2009 FIELD STUDY

APPENDIX 11.E. GIS ANALYSIS OF CHANGES IN WETLANDS IN LAKE POINSETT WHERE BENTHIC SAMPLING WAS CONDUCTED IN 2009 FIELD STUDY

APPENDIX 11.F. LITERATURE REVIEW OF HYDROLOGIC AND HABITAT REQUIREMENTS OF CRAYFISH AND APPLE SNAIL

APPENDIX 11.G. FREQUENCY ANALYSIS OF HYDROLOGIC AND SALINITY CHANGES DUE TO WITHDRAWALS

APPENDIX 11.H. LITERATURE REVIEW OF SALINITY AND FRESHWATER INFLOW EFFECTS ON BLUE CRAB AND PENAEID SHRIMP