CHAPTER 7. BIOGEOCHEMISTRY

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2012

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ACRONYMS, ABBREVIATIONS, AND CONVERSION FACTORS

Al	Aluminum
ArcMAP	GIS application
BOD	Biochemical oxygen demand
C	carbon
C:N	Carbon to nitrogen ratio
Ca	Calcium
Cfs	Cubic feet per second
Chl-a	Chlorophyll-a
DEM	Digital Elevation Model
DF	Degrees of Freedom
DO	Dissolved oxygen
DOC	Dissolved organic carbon
DP	Dissolved phosphorus
EPA	U.S. Environmental Protection Agency
F.A.C.	Florida Administrative Code
FDEP	Florida Department of Environmental Protection
Fe	Iron
GIS	Geographic Information System
ha-m	Hectare-meter
HCl	Hydrochloric acid
LiDAR	Light Detection and Ranging technology
LOI	Loss on ignition
LPC	Lake Poinsett center (sampling location)
LPI	Lake Poinsett inlet (sampling location)
LPO	Lake Poinsett outlet (sampling location)
LWI	Lake Winder inlet (sampling location)
LWO	Lake Winder outlet (sampling location)
MBC	Microbial biomass carbon
MCA	Marsh Conservation Area
MFLs	Minimum flows and levels
Mg	Magnesium
MGD	Million gallons per day
MSE	Mean Standard Error
Mt	Metric ton
N	Nitrogen
N	Number of observations
NAT FWS MAR	Natural freshwater surface flow marsh hybrid systems
NGVD29	National Geodetic Vertical Datum, 1929
NH_4-N	Ammonium nitrogen
NO_x	Nitrate and nitrite
NPP	Net primary production
NRCS	Natural Resources Conservation Service
OC	Organic carbon

OEW	Orlando Easterly Wetlands
OM	Organic matter
P	Phosphorus
P	Probability
PCU	Platinum cobalt units
PLRGs	Pollutant load reduction goals
PM	Predictive model
PO_4	Dissolved orthophosphorus
r^2	Coefficient of determination
RI	Return interval
RT	Recovery time
Se	Standard error
SE	Supporting evidence
SJR	St. Johns River
SJRWMD	St. Johns River Water Management District
SOM	Soil organic matter
SRP	Soluble reactive phosphorus
TIS	Tanks-in-Series Model
TKN	Total Kjeldahl nitrogen
TMDL	Total maximum daily load
TN	Total nitrogen
TOC	Total organic carbon
TP	Total phosphorus
U.S.C.	United States Code
UM	Understanding of causal mechanisms
USDA	US Department of Agriculture
USGS	United States Geological Survey
USJRB	Upper St. Johns River basin
WSIS	Water Supply Impact Study

1 ABSTRACT

In warm climate wetlands, organic matter and associated nutrients are alternately sequestered and released from wetland soils through the wet and dry season cycles of inundation and exposure. The ratio of the time exposed to the time inundated, influences the degree of oxidation and therefore the total mass release of organic matter and nutrients. Because anthropogenic alteration of hydrology as a result of water withdrawals can increase the duration of exposure of organic soils, we examined the potential for increased release of organic matter. We estimated annual release rates per additional day of exposure for total organic carbon (TOC), nitrogen (N), and phosphorus (P) by measuring changes in soil inventories and net releases to overlying water for soils subjected to varying periods of exposure. Carbon and nitrogen annual release rates increases were 18.7 and 2.28 mg m^{-2} d^{-1} of exposure, respectively while total phosphorus ranged between -0.3 and 2.4 mg m^{-2} d^{-1} of exposure. These incremental increases in the annual release rates were much lower than expected and reflected the refractory character and moderate percentage of organic matter in the wetland soils associated with this portion of the river.

We used the daily increase to the annual release to estimate the increases in annual mass releases associated with various water withdrawal scenarios that reflected differing levels of withdrawal, land use, and completion of regional water projects in the Upper St. Johns River. The increase in constituent annual mass release was the product of the additional hectare-days of exposure in each simulated year and the daily release rates. The median increase in annual mass releases for the 33-year model period under different scenarios ranged from 0 to 63 metric tons (mt) for organic carbon (OC), 0 to 2.3 mt for phosphorus (P) and 0 to 8.9 mt for nitrogen (N). The increased wetland soil releases were used in a model simulation of mass reductions to estimate the amount of the released constituent that would reach the river. Median values for reduction rates under the most extreme scenario (Full1996NN) calculated that median annual increases in loadings of up to 29 mt of OC, 0.5 mt of P, and 2.1 of N reach the river. The maximum simulated increases in loadings to the river were small relative to baseline loadings (<2% for OC & P and <4% for N) and would lead to quite small increases in constituent concentrations.

The greatest potential for deleterious effects would be decreases of dissolved oxygen (DO) concentrations resulting from increased loadings of dissolved organic carbon (color) and nutrients. We predicted the relative change of DO in the river using a multiple regression with water level and concentrations of TOC and TP ($r^2 = 0.415$; $p < 0.0001$) as predictors of DO. Results suggested that the effect of additional wetland soil exposure in the most extreme scenario could culminate in reducing median wet season concentrations of DO by 0.005 mg L^{-1}. We assessed these effects by considering their strength, persistence, and the diversity of species and functions affected. We concluded that ecological effects for the most extreme scenario on the most effected section of the river would be negligible. However, if this same extreme scenario was assessed using soil release rates from the Blue Cypress Marsh Conservation Area upstream, effects were much greater (median DO declines of 2.47 mg L^{-1}).

Our analysis indicates that the effects of anthropogenic alteration of wetland hydroperiod and subsequent release of organic carbon and nutrients will depend upon the lability of soil organic matter. Given the remaining scientific uncertainties, it appears prudent to employ a phased, adaptive management approach to additional water withdrawals which includes careful monitoring.

2 INTRODUCTION

The St. Johns River is a low-gradient, blackwater river, with abundant riverine and floodplain wetlands, especially in its upper reaches. It is the longest river wholly within Florida; and unlike many large rivers in the United States, it flows northward. It is a coastal plain river that begins in the low elevation freshwater marshes of St. Lucie and Indian River counties, just inland of Vero Beach (Cox and Vosatka 1976; Demort 1990). It drains an area of approximately 24,000 km^2 (9,000 mi^2) but drops only 8 m (26 ft) in surface water elevation, or level, over its 512 km (318 mi) length (Demort 1990), yielding an average slope of 1.6 cm km^{-1} (0.63 in. mi^{-1}) (McGrail et al. 1998). Most of the water elevation drop occurs within the first third of the river. The St. Johns River is generally divided into three basins (upper, middle, and lower) based on water quality and tidal influence (Bass and Cox 1988). Note that we use water level in a relative sense and use water elevation when it is in relation to sea level (National Geodetic Vertical Datum, 1929 [NGVD29]); water stage or level is used by many of the chapters in this report whereas when we use water elevation, it is always in reference to NGVD29.

Because the character of a river is strongly affected by its adjacent wetlands (Schlosser and Karr 1981**;** Burt and Pinay 2005; National Academy of Sciences 2002), the health of the St. Johns River reflects the health and functions of its associated riverine and floodplain wetlands. The river–wetland interaction is especially strong in the southern reaches (headwaters) of the river, where there is very little topographic relief and there are extensive peat-based, freshwater marshes (Demort 1990). The low relief has also resulted in the formation of wide slow-flowing areas of the river, which are the lakes of the St. Johns River. These lakes are generally highly colored and mildly acidic, with high levels of nutrients, but low primary productivity, so they do not fit a typical trophic classification scheme (Hanson 1962; Hutchinson 1975); instead, they require classifications that recognize their dystrophic nature (Hanson 1962; Shannon and Brezonik 1972; Carpenter and Pace 1997).

Hydrology is the prime determinant of wetland form and function (Holden 2005; Martin et al. 1997; Reddy and Delaune 2008). Changes in a river's hydrology can cause hydrologic and ecologic changes in its floodplain wetlands (Baldwin and Mitchell 2000; Burt and Pinay 2005). For example, prolonged inundation of wetlands inhibits organic matter (OM) oxidation and promotes the accumulation of OM (Craft and Richardson 1993; Reddy et al. 1993; Qualls and Richardson 2003; Aldous et al. 2005; Graham et al. 2005). The accumulation of OM adds to the storage compartments for carbon (C) and nutrients, such as nitrogen (N) and phosphorus (P) (Dunne et al. 2007; Reddy and Delaune 2008), and is often cited as the single most important long-term sink for both C and nutrients (Craft and Richardson 1993; Reddy et al. 1993). Conversely, exposure of wetland soils stimulates OM oxidation, reduces OM accumulation, and can lead a loss of OM and release of C and nutrients (Stephens 1956; Aldous et al. 2005; Reddy and Delaune 2008).

Surface water withdrawals from the St. Johns River could change the river's hydrology and consequently cause reduced storage of OM, thus C and nutrients, in the associated floodplain wetlands. The goal of the Biogeochemistry Working Group was to assess these potential effects. We based our efforts on the simulated hydrology (see Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results) developed by the Hydrology and Hydrodynamics

Working Group for selected points along the St. Johns River for various hydrologic scenarios (Table 2–1).

Table 2–1. Scenarios used in the Biogeochemistry Analysis. See Chapter 6. River Hydrodynamics Results for a discussion of the scenarios.

Scenario Name	Withdrawal Rate	Land Use	USJRB Projects	Sea Level Rise
Base1995NN	0 MGD	1995	Not complete	No accelerated rate
Half1995NN	77.5 MGD	1995	Not complete	No accelerated rate
Full1995NN	155 MGD	1995	Not complete	No accelerated rate
Half1995PN	77.5 MGD	1995	Complete	No accelerated rate
Full1995PN	155 MGD	1995	Complete	No accelerated rate
Half2030PS	77.5 MGD	2030	Complete	Accelerated rate
Full2030PS	155 MGD	2030	Complete	Accelerated rate
FullOR2030PS	262 MGD	2030	Complete	Accelerated rate

2.1 HYDROLOGIC AND BIOGEOCHEMICAL CONTROLS

Floodplain wetlands are important interfaces between upland and aquatic ecosystems. They buffer river discharges from upland runoff during high rainfall events, and they can receive discharges from rising river water (Wetzel 1984). Wetlands have high primary production rates and low decomposition rates, relative to upland ecosystems, and are often net sinks for C and nutrients such as P and N (Kadlec and Wallace 2008; Reddy and Delaune 2008) but can also export large quantities of dissolved organic carbon (DOC) (Reddy and Dangelo 1997; Schiff et al. 1998).

Wetlands develop organic soils because primary production rates are high but decomposition rates of OM are low under saturated and inundated conditions, (Reddy and Delaune 2008; Wetzel 1984). Consequently, OM deposition usually exceeds decomposition. The rate of accumulation of organic wetland soils and the rates of storage of C and nutrients depends on the net difference between production and decomposition (Holden 2005). If organic wetland soils are drained (i.e. not inundated or saturated) for sufficient time, decomposition rates increase; unless production is also stimulated, the balance may shift towards decomposition. Thus, the primary process for accumulating organic matter may cease and the soils may become a net source for C and nutrients rather than a net sink (Holden et al. 2004). Water level drawdown or drainage of a wetland contributes to soils becoming more oxic rather than anoxic. The shift toward oxic conditions increases aerobic respiration, decomposition rates, microbial activity, and nutrient cycling and release. Other changes can include a decrease in soil surface elevation and soil depth as OM decomposes to carbon dioxide (Snyder 2005).

The physical and chemical stability of organic wetland soils in the USJRB requires extended periods of inundation (long hydroperiods) (DeBusk and Reddy 2003; Reddy et al. 2007).

Reduced hydroperiods in some portions of the USJRB have caused soil losses of 3 cm yr^{-1} during the last 10 years (SJRWMD, unpublished data). Because surface water withdrawals decrease hydroperiods and increase exposure of organic wetland soils, additional total organic carbon (TOC), N, and P may be released from the wetlands to the St. Johns River when these area are reflooded. The increased mass loads of TOC and nutrients may result in ecological impacts to the river. A direct effect we addressed in our analysis is the potential for these loads to decrease DO concentrations ([DO] brackets denote concentration) in the river. Our estimates of potential increases in mass loading of TOC and nutrients, if any, and our estimates of the potential changes in lake [DO] were provided to other working groups to use in their assessments of other potential ecological effects from surface water withdrawals from the St. Johns River.

We hypothesized that surface water withdrawals could increase exposure of wetland soils in terms of both duration (days) and area (hectares). These increases in exposure could, in turn, increase TOC and nutrient releases and increase TOC and nutrient loads to the St. Johns River. An increase in mass loads has implications for river water quality and ecosystem stability. We examined results from the Environmental Fluid Dynamics Computer code (EFDC) hydrodynamic modeling and the Hydrologic Simulation Program- FORTRAN (HSPF) hydrologic modeling (see Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results) along with the distribution of organic wetland soils and concluded that the area around Lakes Winder and Poinsett would be most sensitive to hydrologic changes and concentrated our biogeochemistry analysis on these areas. This process is described later in Section 3.3.

2.2 PURPOSE AND OBJECTIVES

The purpose of this chapter is to assess whether surface water withdrawals from the St. Johns River could affect wetland biogeochemistry, which in turn could affect river water quality and the river ecosystem.

The objectives and tasks for the biogeochemistry analysis were:

(1) Identify the water quality constituents that could affect the river due to changes in wetland soil biogeochemistry caused by the modeled water withdrawals.

(2) Determine the number of potential additional days of wetland soil exposure by evaluating modeled changes in water elevation at various locations and under various modeled water withdrawal and land use hydrologic scenarios.

(3) Quantify the potential areal extent of increased wetland soil exposure by comparing various modeled water withdrawal and land use hydrologic scenarios against Base1995NN (the base scenario) using hypsographic information obtained from the Digital Elevation Model (DEM).

(4) Generate experimental data to quantify TOC, P, and N releases from organic wetland soils from predicted increased days of soil exposure.

(5) Use a Wetland Constituent Release Model to estimate increase in mass load contributed by wetland soils due to predicted increased days and area of exposure based on model hydrology.

(6) Use a Wetland Constituent Reduction model to estimate how much TOC, P, and N reach the river.

(7) Determine the empirical relationships between [TOC], [N], and [P] and [DO] in lake water.

(8) Use these empirical relationships to estimate the potential decrease in lake [DO] under various model scenarios.

(9) Convey relevant findings to other working groups, such as the Plankton and Fish Working Groups, to aid in their assessments of modeled water withdrawals.

2.3 Exchanges with Other Working Groups

The Hydrology and Hydrodynamics Working Groups (Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results) provided simulated hydrology at selected points along the St. Johns River for the various hydrologic scenarios (see Table 2–1). The Wetland Vegetation Working Group provided a Digital Elevation Model (DEM) of the land surface, which we used to calculate the additional area of exposed wetland soil due to the modeled surface water withdrawals (Chapter 10. Wetland Vegetation). Benthic Macroinvertebrates and Fish Working Groups provided information on thresholds for effects in [DO] that were ecologically important within their various areas of expertise.

The Biogeochemistry Working Group provided information to other working groups. Estimated changes in DOC and nutrient loads to the river were provided to the Plankton and Submersed Aquatic Vegetation Working Groups. Estimates of potential changes in [DO] were supplied to the Benthic Macroinvertebrates and Fish Working Groups.

2.4 Conceptual Model

We developed a conceptual model to illustrate the pertinent hypothesized causal linkages between surface water withdrawals and effects on wetland organic soils (Figure 2–1). Hydrologic effects were inputs from the Hydrology and Hydrodynamics Working Groups. These inputs allowed calculation of predicted hydrologic changes from the Base1995NN (baseline) scenario for different scenarios of water elevations and discharge at various locations along the river (see Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results). A reduction in water levels can elicit biogeochemical effects by decreasing inundation and, therefore, increasing the period of exposure of floodplain wetland organic soils. By facilitating aerobic decomposition, the increased exposure of organic soils may cause increased rates of mineralization and a subsequent release of C and nutrients upon inundation. As the conceptual model illustrates, increased mineralization of nutrients can lead to increased nutrient loading to adjacent or downstream water bodies. The modeled increases of the load produced by deviation from base release rates of C and nutrients were given to the Plankton and the Submersed Aquatic Vegetation Working Groups as part of their assessment of water withdrawals. These working groups then reciprocated by providing the Biogeochemistry Working Group with information on the effects of these increased loads.

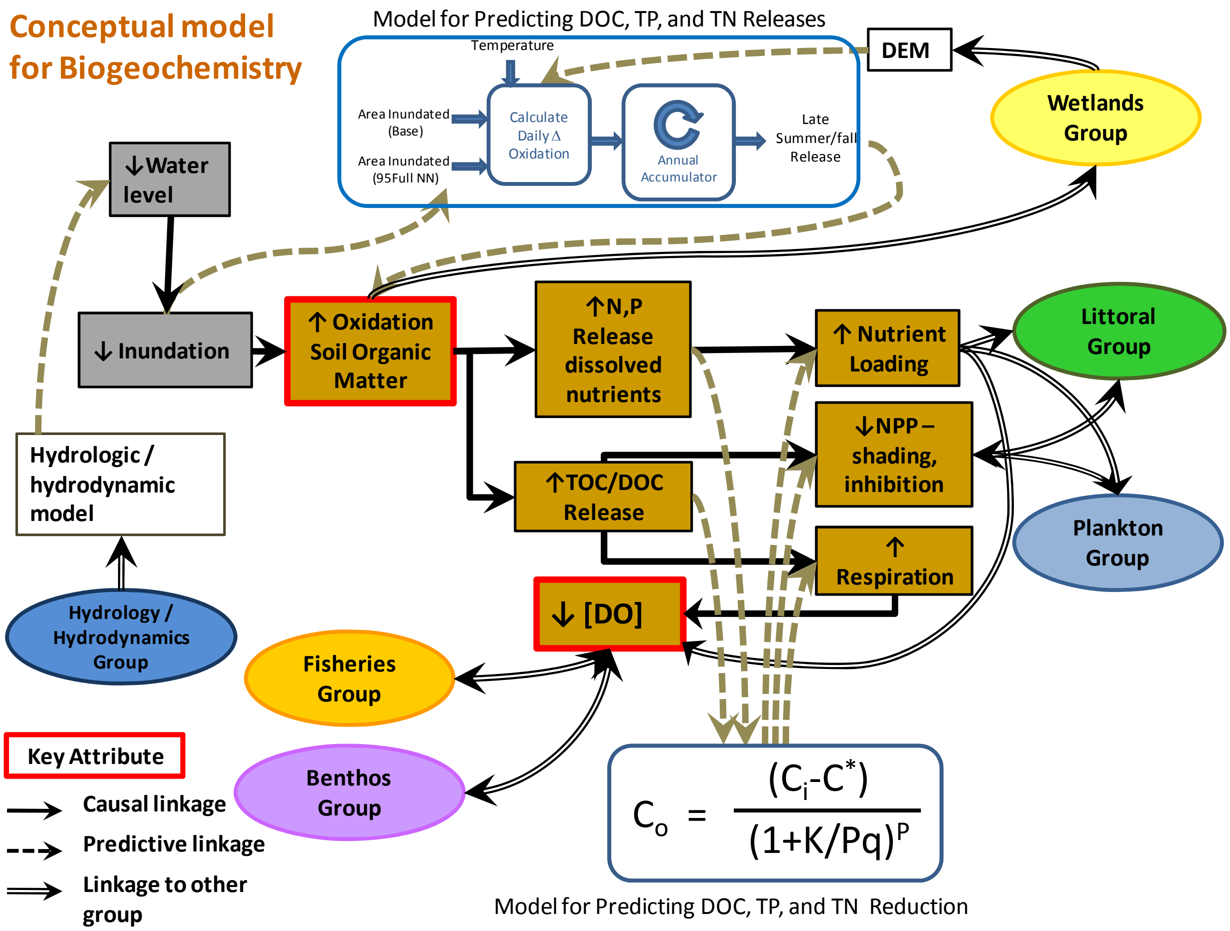

Figure 2–1. Conceptual model showing causal and predictive interactions with the indicated biological groups of biogeochemistry of wetland organic soils due to water withdrawals. DOC = dissolved organic carbon, TP = total phosphorus, TN = total nitrogen, DEM = Digital Elevation Model, N = nitrogen, P = phosphorus, TOC = total organic carbon, DO = dissolved oxygen, NPP = net primary production.

The conceptual model also indicates pathways for causation and information that lead to changes in lake [DO]. OC is more available in its dissolved state; therefore, the microbial community is expected to metabolize many of these dissolved organic carbon (DOC) compounds and consume oxygen in the process. The less labile DOC is also degraded by ultraviolet light, which releases more labile subgroups to the microbial community. The DOC and UV interactions can also consume DO through iron oxidation and reduction reactions (Miles and Brezonik 1981). All these processes contribute to an increase in the oxygen demand and result in reduced [DO]. Information on any predicted reduction of [DO] was conveyed to the Benthic Macroinvertebrates and Fish Working Groups for use in their assessments of the water withdrawal scenarios (see Chapters 11 and 12).

2.5 Assessment of River-Wetland Hydrology

The magnitude and direction of the river–wetland interaction varies along the river's length and also varies with season, local terrain, soils, and macro- and micro-topography of the floodplain.

In segment 9, which includes the headwaters of the St. Johns River around Blue Cypress Lake, rain generally drains from the wetland into the lake or river. Further downstream, water may drain laterally from the wetlands to the river or, with rising river stage, may flow from the river into the wetlands. In the middle and lower reaches of the river, wetlands are flooded primarily by rising river stage caused by upstream runoff. For approximately the first 250 km (155 mi) from the mouth, the river's elevation is constrained by sea levels (Figure 2–2).

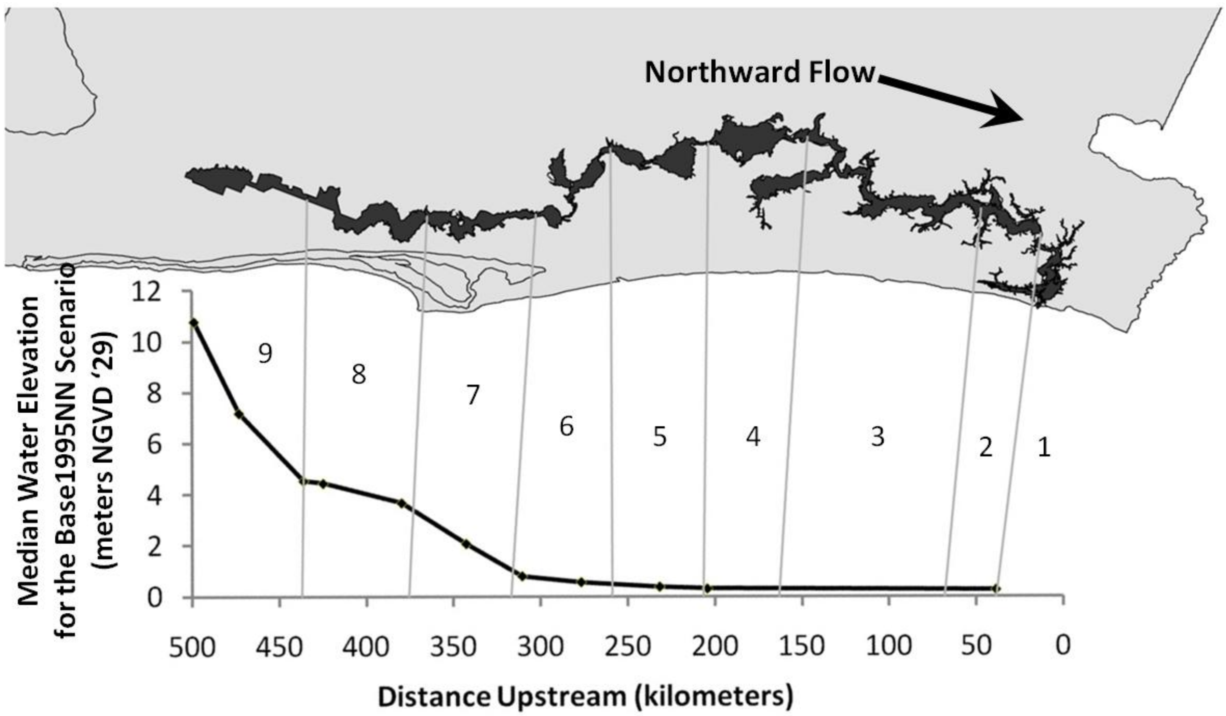

Figure 2–2. Median water elevation along the St. Johns River with river kilometers and river segment numbers (1 to 9). Data is from the Base1995NN scenario. Note that the environmental assessment for the water withdrawal ends at a weir in segment 8, where more than half of the river elevation drop has occurred.

Water levels in the wetlands and the river often become decoupled because of the hydrologic resistance (or roughness) of vegetation, the wetland surface, and micro- and meso-topography. The degree of decoupling is inversely related to wetland water levels, and it increases greatly as water levels approach the soil surface (Kadlec and Wallace 2008). These effects are well known and can be modeled. A two-dimensional model, preferably linked to a groundwater model, is required. Such modeling requires detailed information on soil, vegetation, and meso-topography (Restrepo et al. 1998). These data and models were not available for this analysis. Consequently, we used a simple flat-water surface assumption; that is, the water elevation in the wetlands was assumed equal to the water elevation in the adjoining river or lake.

The flat-water surface assumption overestimates wetland drainage rates, so it also overestimates the soil exposure and underestimates wetland inundation. We dampened the effect of overestimation by using the differences in duration of soil exposure between modeled scenarios rather than of absolute durations of exposure. All other things are essentially constant, including evapotranspiration and rainfall. While the error in the flat-water surface assumption is biased to underestimating inundation, the difference between any two scenarios has bias or error because it removes the larger part of the bias common to both scenarios. For that reason, we based all of the models and analyses on the differences between the scenarios.

2.6 Hydrologic Status and Trends

The Upper St. Johns River Basin (USJRB) extends from the headwater marshes near Vero Beach, 130 km (80 mi) to Lake Harney. There is no river channel in the St. Johns River headwater marshes. Once the natural river channel begins, it flows through a series of river-run lakes (i.e., Lakes Hell 'n' Blazes, Sawgrass, Washington, Winder, and Poinsett). Water has a very short residence time of several days through these lakes when water elevations are high. The long-term median discharge for Lake Poinsett is 17 $m^3 s^{-1}$ (600 cfs), which equates to a long-term median hydraulic residence time of only 19 days.

The USJRB contains river segments 7, 8, and 9. Our studies concentrated on the northern end of segment 8, downstream of the Lake Washington weir. This area is dominated by Lakes Winder and Poinsett. Segment 8, like much of the upper basin, is comprised of a mosaic of floodplain marshes, lakes, river, and stream channels (Fisher et al. 2009; Goolsby and McPherson 1970).

The wetlands of the USJRB are freshwater marshes dominated by herbaceous, emergent marsh vegetation such as maidencane (*Panicum hemitomon*), Pickerelweed (*Pontederia cordata*), and Yellow Water Lily (*Nuphar luteum*). In marshes north of Lake Poinsett, species such as *Spartina bakeri* dominate large wetland areas that are intermittently flooded (Dermort 1990). The northern part of segment 8, which includes Lakes Winder and Poinsett, was the focus area for the Biogeochemistry Working Group analysis. Lake Winder is about 6 km^2 (1,480 ac) in area and is located 8 km (5 mi) upstream of the larger (18 km^2; 4,450 ac) Lake Poinsett. The river exits the northeastern portion of Lake Winder and flows northward toward Lake Poinsett. Lake Poinsett is largely open water, but extensive areas of vegetation occur along the western shore of the lake (Demort 1990). Between the lakes, the river is channelized and has a sand bottom (Demort 1990).

Water elevation data for Lake Poinsett's outlet (1953 through 2009 U.S. Geological Survey data for SR 520) indicate that there has been no long-term trend in water elevation. Although simple regression analysis indicates a downward trend in water elevation through time (up to 0.008 $m\ y^{-1}$), water elevation varies cyclically (Figure 2–3 A, B). Testing of temporal trends in such cases requires transformation of the data to remove the cyclic component of variation. We used the unobserved components models module in Statistical Analysis Software (SAS Institute, Inc., Cary, NC, USA) to separately test the significance of the linear and cyclic trends. In this case, once the significant periodicities were removed, the linear slope was insignificant (i.e., it is not statistically different from zero; Figure 2–3 C).

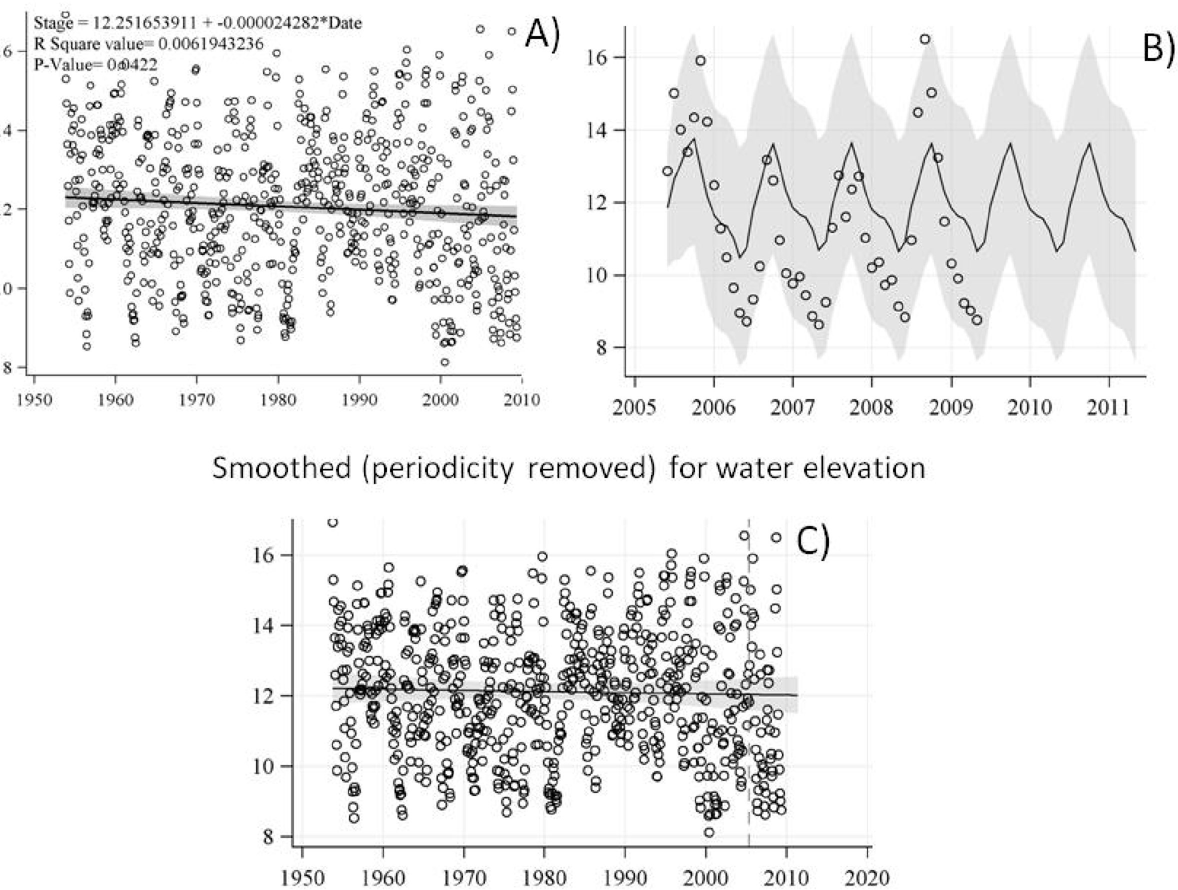

Figure 2–3. Time series analysis for observed water elevations at Lake Poinsett. (A) Simple regression of monthly water elevations indicates a significant decline in water elevations (NGVD29) at Highway 520 on Lake Poinsett. (B) Repeating cycle revealed by the unobserved components models module of Statistical Analysis Software. (C) Transformed data with no linear trend.

Based on vegetation patterns evident in SJRWMD GIS rasters of aerial photography taken in 2004 (St. Johns River Water Management District 2004) and in the early 1940s (USDA. 1938-1944), there is little evidence for changes in the extent of wetland communities due to anthropogenic changes during this interval. Further upstream, there were major short-term changes in the hydrologic regime caused by removal of natural river obstructions (i.e., dense aggregations of floating mats of vegetation, known as jams) at Lake Washington, followed by installation of a weir. Apparently, these modifications did not affect the long-term wetland hydrology of Lakes Winder and Poinsett.

Upstream of the Lake Washington weir, extensive hydrologic and biogeochemical changes have occurred in the floodplain wetlands. Much of the original floodplain was diked and drained and converted to farms or urban areas. As part of the upper St. Johns River Basin (USJRB) Projects, large areas of previously drained wetlands are being reflooded. However, restoration is incomplete in one wetland area within the 2,400-ha (6,000-ac) St. Johns Marsh Conservation Area (St. Johns MCA). The area is still severely overdrained, and portions are subsiding at a rate of approximately 3 cm y^{-1} (0.1ft y^{-1}) with concomitant release of C and nutrients. The extensive loss of upstream wetlands and continuing oxidation of wetland soils in this area have likely had significant water quality effects in Lakes Winder and Poinsett. We believe that water quality in upstream lakes has been degraded by these conditions. A primary goal of the USJRB projects is to remediate these problems (Keenan et al, 2002).

2.7 Water Quality Status and Trends

Lake Poinsett and other USJRB water bodies are considered impaired for DO and nutrients by Florida Department of Environmental Protection (FDEP) and U.S. Environmental Protection Agency (EPA). Rule 62-302.530(30) Florida Administrative Code (F.A.C.) sets surface water quality standards that require [DO] of not less than 5.0 mg L^{-1} for Class III freshwater bodies. Normal daily and seasonal concentrations must also be maintained above this level. Nutrient loads and concentrations must not cause violations of other listed criteria or cause an imbalance in natural populations of flora and fauna (Rule 62–302.530(47) (a) (b) F.A.C.). Many water bodies in this area, including Lake Poinsett, are listed for development of total maximum daily loads (TMDLs) to comply with section 303(d) of the Clean Water Act codified in 33 United States Code (U.S.C.) §1313.

The SJRWMD previously set pollutant load reduction goals (PLRGs) for lakes of the upper basin (Keenan et al. 2002). These PLRGs were based on achieving a maximum monthly mean concentration of total phosphorus [TP] of 0.09 mg L^{-1} and which would require reduction of P loads by 27% to 48%. The PLRGs were the basis for TMDLs initially adopted by EPA that were later apparently rescinded, perhaps because of procedural errors. The PLRGs remain the goals for SJRWMD and match well with EPA's currently adopted TMDLs for Lake Sawgrass and the river above Lake Poinsett (Gao 2005).

Water quality dynamics in the headwaters and tributaries of the upper St. Johns River seem to fit the river continuum theory (Vannote et al. 1980) with nutrients (Ensign and Doyle 2005; Webster and Patten 1979) and with allochthonous organic carbon (OC) supplied both from drainage basin leaf fall and from upper river segments. The five river-run lakes, Lakes Hell 'n' Blazes, Sawgrass, and Washington, Lakes Winder and Poinsett, as well as the main stem of the upper basin appear also to follow the flood pulse theory, with allochthonous C supplied by lateral floodplain wetlands at and during the draining of adjacent wetlands following a flood pulse (Benke et al. 2000; Junk et al. 1989; Tockner et al. 2000). This would predict significant allochthonous OC loads from the adjacent, lateral wetlands, which should be sensitive to increased exposure periods, as a result of surface water withdrawals. In contrast, the middle and lower St. Johns River seem to conform to the river production theory (Thorp 1994), in which the primary labile OC source is autochthonous and primarily from phytoplankton. This implies that the USJRB and its lakes are likely more sensitive to wetland biogeochemical dynamics than middle or lower basin river and lakes.

The lakes of the USJRB, which include Lakes Winder and Poinsett, are dystrophic, heterotrophic systems that have low primary productivity (Belanger et al. 1985; Fisher et al. 2009; Goolsby and McPherson 1970) and muted primary production responses to the addition of nutrients (Alderidge and Schelske, 2000). The cause of dystrophy is high levels of colored dissolved OM; average color for Lake Poinsett between 1979 and 2010 was 192 platinum cobalt units (PCU). Dystrophic systems tend to be mildly to highly acidic, and are colored or stained with different mixtures of tannins, humic substances, and organic acids (Wetzel 1992). Many of these organic compounds are resistant to further biological degradation but are slowly broken down by photolysis. Algal primary production is generally low such that most of the chemical energy flowing through the ecosystem comes from allochthonous OM, carried into the receiving system from the watershed, and from quasi-autochthonous OM, supplied by floodplain wetlands (Wetzel 1992).

Episodic hypoxia (low DO) or anoxia (no DO) occurs in many dystrophic rivers. The Amazon (Junk 1997), the Okavango Delta in Botswana (Mladenov and McKnight 2003), and the rivers of the Pantanal (Hamilton et al. 1997) all exhibit hypoxia or anoxic events at least occasionally and often annually. With low primary productivity and high [OM], [DO] in the St. Johns River is also generally low, often < 5 mg L^{-1} (Belanger et al. 1985; Cox and Vosatka 1976; Goolsby and McPherson 1970; Lowe et al. 1984). Nearly every year, USJRB lakes experience hypoxic events in late summer or early fall. These events are contemporaneous with inundation of adjacent wetlands (Figure 2–4) and are often severe enough to cause fish kills (Keenan et al. 2008) (Figure 2–5). Although these events occur naturally, there are concerns that the frequency and extent of hypoxia events have been exacerbated by anthropogenic impacts to the basin. In this assessment, we evaluated whether increased surface water withdrawals have the potential to contribute to this degradation.

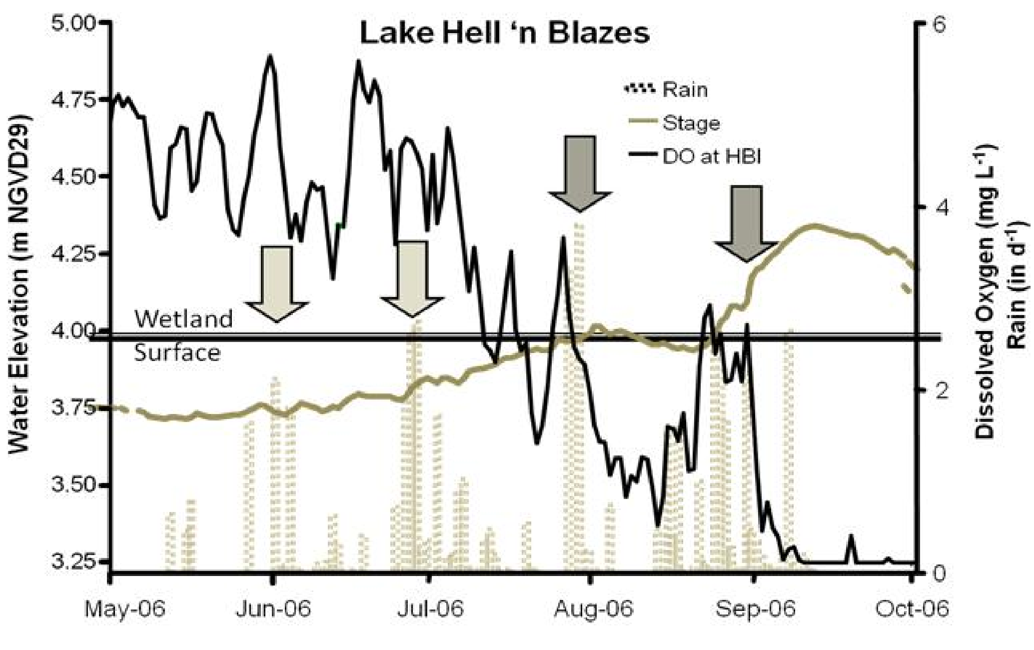

Figure 2–4. Relationships among water elevation, wetland surface, rainfall, and dissolved oxygen (DO). Light arrows depict large rainfall events with little DO response, and dark arrows depict rainfall events when water elevation is above wetland surface with large DO reductions.

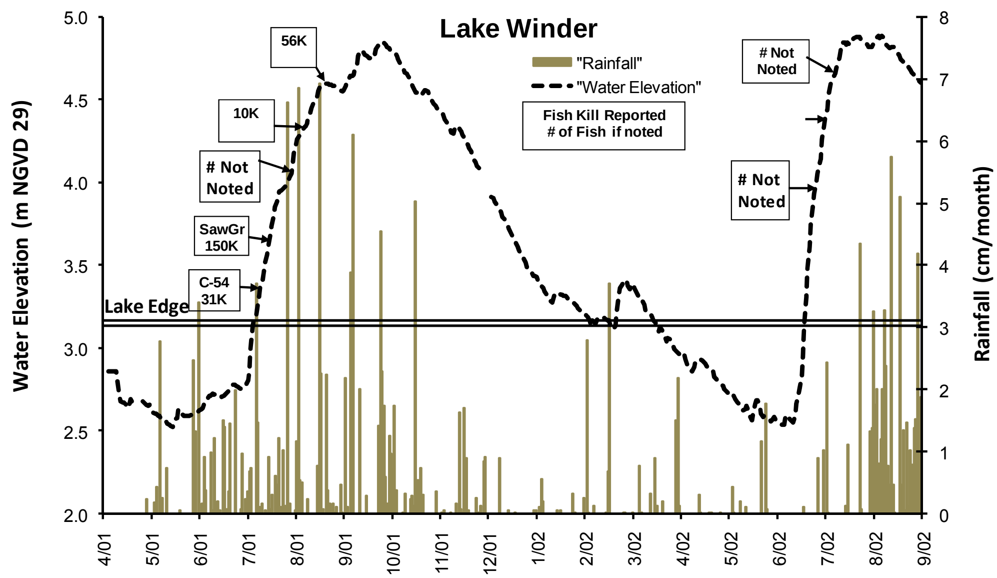

Figure 2–5. Water elevation, rainfall, and lake edge elevation (wetland surface), and their association with fish kills. Fish kills correspond to rainfall events during the rising limb of the hydrograph (June through October wet season) and when the water elevation begins to inundate the wetlands.

It is not obvious what factors lower the [DO] in Lakes Winder and Poinsett, but some possibilities are release of nutrient limitation of the microbial community, the influence of plankton, inflow of low DO wetland floodplain waters to the lakes (Gao, 2005), and an increase in the loading of labile OC from the floodplain wetlands. The total nitrogen (TN) to total phosphorus (TP) ratio for Lake Poinsett between 1983 and 2010 is 25, indicating a N and P co-limitation. Because sediment oxygen demand (DB Environmental 2007) and phytoplankton numbers are generally low in these lakes (Belanger et al. 1985; Fisher et al. 2009), it is reasonable to conclude that the oxygen demand that causes hypoxia is allochthonous. Over drainage of upstream, floodplain wetlands could contribute to pulsed releases of labile TOC and nutrients (and therefore increased oxygen demand) upon reflooding (Tank et al. 2010; Wilson et al. 2011). It is expected that completing the USJR projects will reduce over drainage and decrease the release of labile TOC and nutrients and therefore also reduce the frequency and magnitude of hypoxic events.

Much of this chapter focuses on C and its potential loss from wetland soils as a result of increased exposure. We typically used [DOC] as a measure of [TOC] in water; however, we often measured [TOC]. In this work, we have used these two parameters interchangeably, depending on available data. As part of method development and prior to reporting results, we established a relationship between [DOC] and [TOC] in Lake Poinsett water (Figure 2–6). These data show that, for this system, [TOC] is a good estimator of [DOC] because nearly all of the TOC is in the for of DOC (average of 98%).

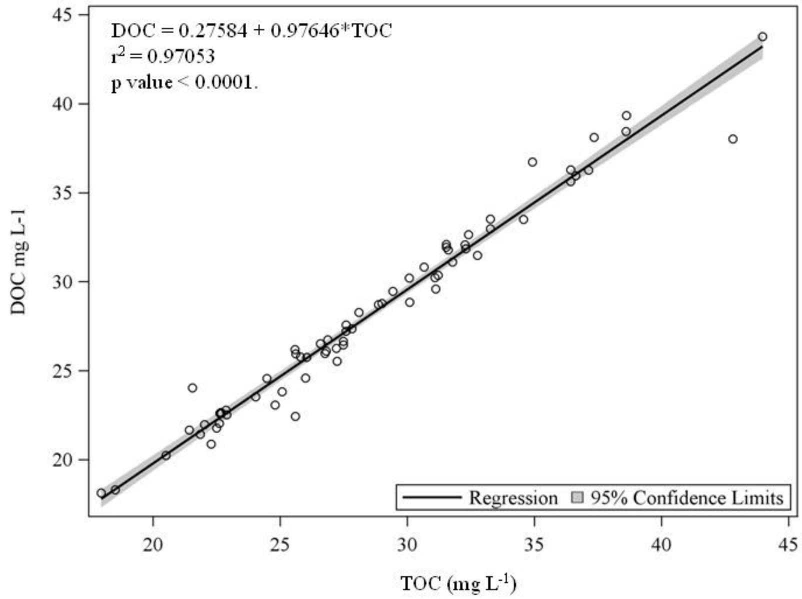

Figure 2–6. Regression between total organic carbon (TOC) and dissolved organic carbon (DOC) in Lake Poinsett (1979 to 2008). r^2 = coefficient of determination, p = probability.

Long-term [DO] trends in Lake Poinsett suggest that concentrations are decreasing (Figure 2–7), whereas concentrations of total phosphorus [TP], total organic carbon [TOC], dissolved organic carbon [DOC], chlorophyll-a [Chl-a], and total nitrogen [TN] are increasing (Figure 2–8).

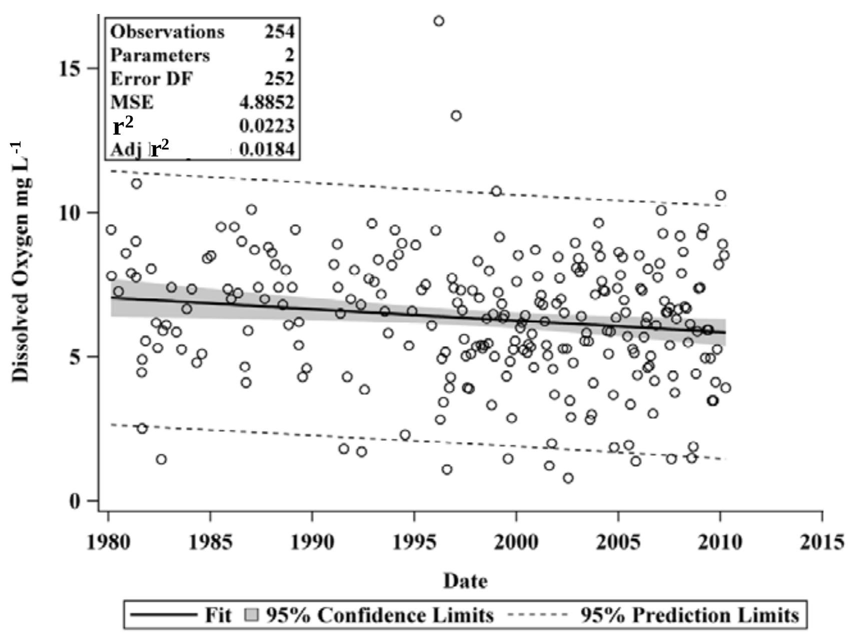

Figure 2–7. Monthly dissolved oxygen values with linear regression and 95% probability of slope shaded with 90% prediction lines (dashed lines). The period of record is 1979 to 2010. DF = degrees of freedom, MSE = mean square error, r^2 = coefficient of determination.

Generally, the historic trend for most water quality constituents in Lake Poinsett is deteriorating as shown in Figure 2–8, but [Chl-a] has remained quite low. For example, [Chl-a] averaged 11 µg L^{-1}, and showed little change, during the period of record 1979 through 2010.

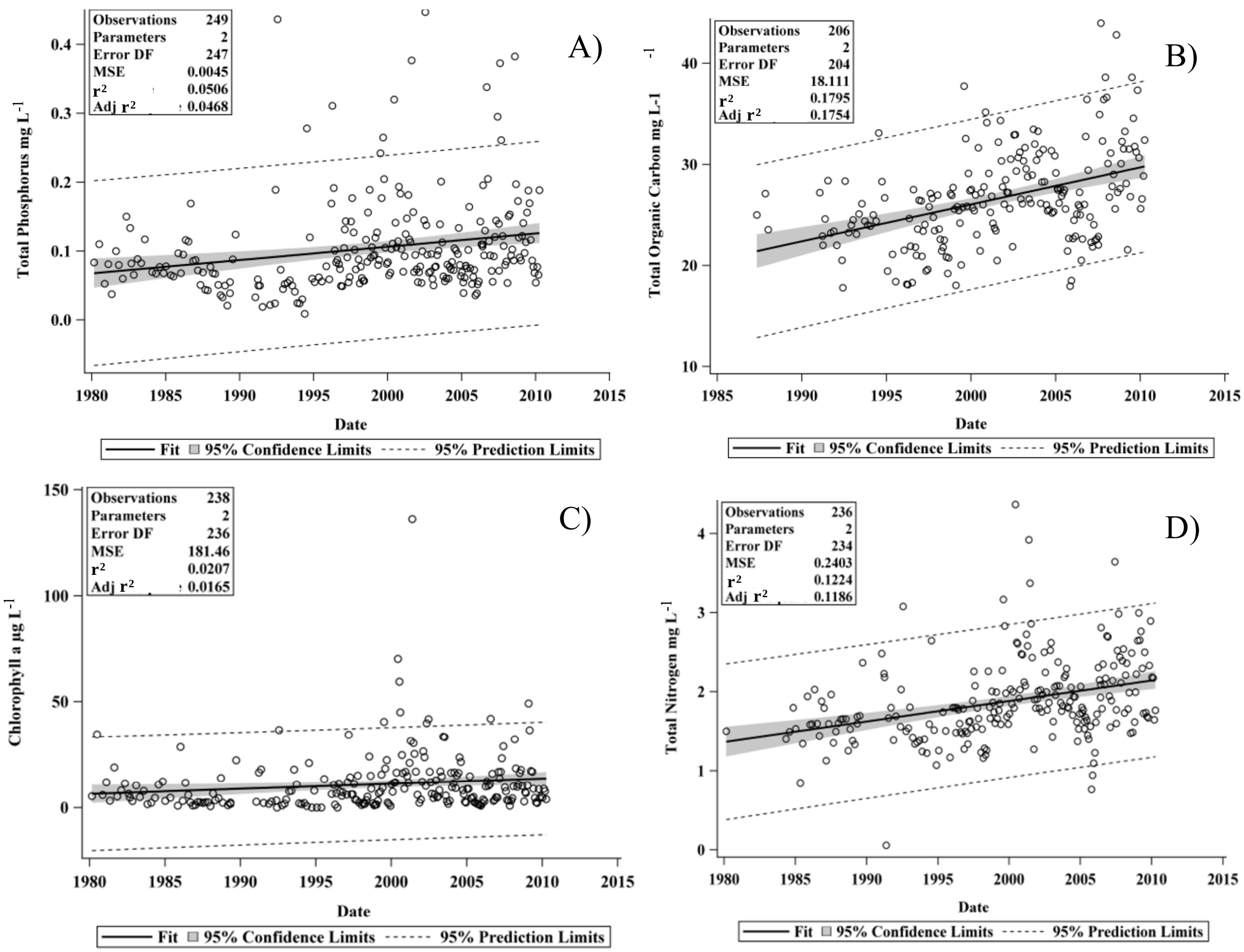

Figure 2–8. Monthly values with linear regression and 95% probability of slope shown shaded and with 90% prediction lines (dashed lines) for (A) total phosphorus. (B) total organic carbon, (C) Chlorophyll-a, and (D) total nitrogen. The period of record is 1979 to 2010. TP = total phosphorus, TOC = total organic carbon, DF = degrees of freedom, MSE = mean square error, r^2 = coefficient of determination.

Within a given year, [DO] in Lake Poinsett decreases from January through August and then increases for the remaining four months. Mean monthly values are generally above the 5 mg L^{-1} threshold, except during the warmer summer and fall months (July through October; Figure 2–9), when rainfall is greatest and wetland water levels are highest.

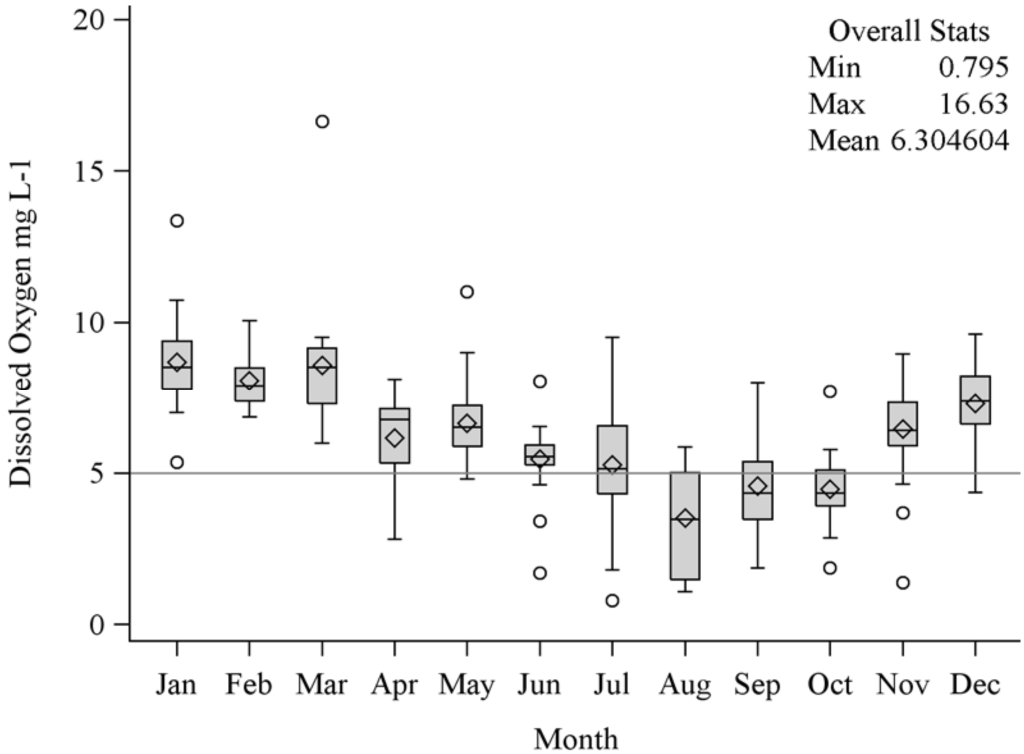

Figure 2–9. Monthly statistics of dissolved oxygen in Lake Poinsett for the period of record 1979 to 2010. The "box" shows the range of data between the 1st and 3rd quartiles (25th and 75th percentiles), with the lower end of the box representing the 1st quartile and the top of the box representing the 3rd quartile. The line inside the box shows the 50th percentile value or median while the diamond symbol is the mean or average. The "whiskers" show the highest or lowest value that falls within 1.5 times the "spread" of the high and low end of the box (where spread is defined as the difference between the 1st and 3rd quartile values). Values that are outside of 1.5 times the spread from the upper or lower end of the box are called "outside values" and are plotted as points.

3 MATERIALS AND METHODS

3.1 DIGITAL ELEVATION MODEL

The Wetland Vegetation Working Group developed a Digital Elevation Model (DEM) for river segment 8 using Light Detection and Ranging (LiDAR) data and ground elevations from several transects previously surveyed by the Minimum Flows and Levels (MFLs) section of SJRWMD (see Chapter 10. Wetland Vegetation). The data from the MFL transects were used to correct the LiDAR ground elevations for vegetation interference. This resulted in a much more accurate DEM (see Chapter 10. Wetland Vegetation). Using this model and ArcMAP, a Geographic Information Systems (GIS) application, we developed hypsographic curves for Lakes Winder and Poinsett, and the sub-basins draining into them. These curves are elevation above sea level, (NGVD29) plotted against cumulative area at or below a given elevation.

3.2 ASSESSMENT OF RIVER–WETLAND HYDROLOGY

Prior to assessing the environmental effects of surface water withdrawals on the St. Johns River, the river was divided into nine segments based on relative ecological and hydrologic characteristics within each segment (see Figure 2–2 and Chapter 2, Appendix 2B). Areas upstream of the Lake Washington weir were not part of our assessments. This weir hydrologically virtually hydrologically isolates the southern part of segment 8 and all of segment 9 from the southernmost surface water withdrawal point an so were not expecte3d to be affected by withdrawals (see Chapter 3. Watershed Hydrology).

We determined the area of wetlands with organic soils directly influenced by the hydrologic regime of the river (riverine wetlands). The total area of wetlands for each segment was determined from land use maps (SJWMD 2011). We then subtracted forested wetlands, since they are nearly always associated with tributaries or seeps, and therefore highly unlikely to be directly affected by river water elevations (see Chapter 10. Wetland Vegetation), leaving only those wetlands most likely to be affected by riverine hydrology. Although, Natural Resources Conservation Service soil maps show most of segment 7 as having mineral soils (SJRWMD 2007), MFL transect data reported at least a few centimeters of organic epipedon (organic sediments) on the surface of nearly all soils sampled in both segments 7 and 8 (Mace 2006, 2007a, 2007c). Consequently, we included all of the riverine wetlands as having organic soils in both segments for our analysis.

We assessed the different segments based on their magnitude of water elevation decrease with withdrawal under the Full1995NN scenario, riverine wetland area and soils, and projected discharges (see Table 2–1).

3.3 SCENARIO ANALYSES AND RANKING

The WSIS scenarios were developed in cooperation with District management and members of the WSIS team to correctly reflect the past hydrology, to encompass a range of possible future conditions, and to represent extreme conditions that are unlikely but would stress the system. The hydrology and hydrodynamic results from analysis of the different scenarios are presented in Chapter 3. Watershed Hydrology and Chapter 6. River Hydrodynamics Results. See Table 2–1 for the scenarios used in the Biogeochemistry analysis.

We ranked the scenarios in order of modeled deviation from the baseline scenario for relevant hydrologic parameters. We did not rank any calibration or base scenarios. We also did not rank any scenarios with sea level rise because the effects would not reach segment 8, and the effects were not modeled that far upstream (south). Under the same reasoning, we did not rank scenarios with additional Ocklawaha River withdrawals because these only affect areas where water level is unresponsive to surface water withdrawals.

Our ranking was based on the average difference in water elevations between the 20th and 80th percentile inundation exceedence levels because this is the range of inundation where the vast majority of wetlands exist (Figure 3–1). We first analyzed the scenario that ranked as having the greatest decrease in average water elevation. If greater than negligible ecological effects were projected from this analysis, we evaluated and documented those effects and then analyzed the next-ranked scenario. We did this iteratively on successive scenarios, until only negligible ecological effects were predicted or all scenarios were analyzed. Because our focus was on Lakes Winder and Poinsett of river segment 8, we applied the process to those lakes and used the entire period of record (1976 to 2008).

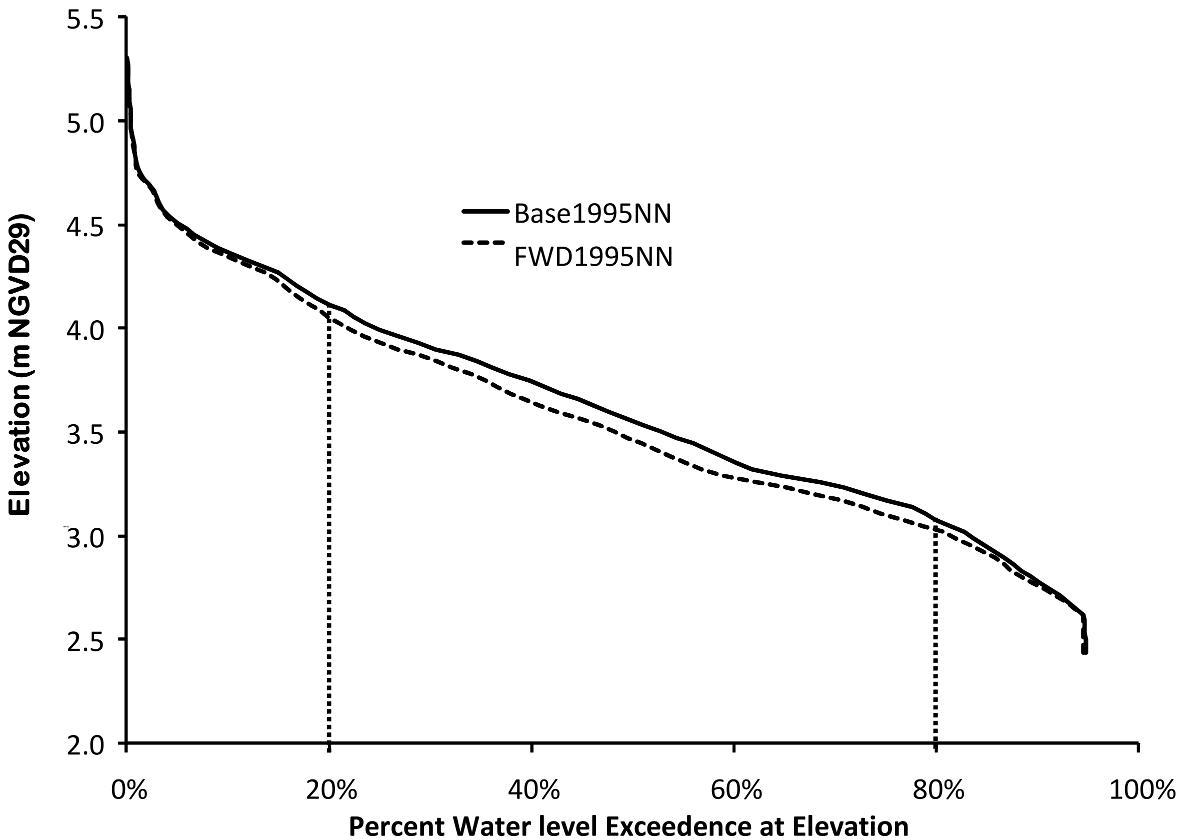

Figure 3–1. Depiction of exceedence curves showing how we developed comparisons for ranking scenarios. We took the average difference (between the lines) from the 20% to 80% exceedence levels.

3.4 SYNOPTIC SURVEY OF WETLAND SOILS

A total of 86 sites were sampled along the river using two methods: (1) surface and subsurface sample method and (2) surface sample plot method (See Appendix 7A for maps and locations). With the first method, samples were collected by inserting a soil core to a depth greater than 30 cm. Soil cores were then extracted, and the cores were transported to the laboratory for extrusion and sectioning into 0 to 10 cm and 10 to 30 cm sections, which were analyzed separately. In the laboratory at the Soil and Water Science Department, University of Florida, soil samples were analyzed for the various physicochemical soil characteristics listed in Table 3–1.

Table 3–1. Soil sample analytes for sectioned cores.

Analytes*
Bulk density
Moisture content
pH
Organic matter (OM) as loss on ignition
Total phosphorus [TP]
Total carbon (TC)
Microbial biomass carbon (MBC)
Total Kjeldahl nitrogen (TKN]
Inorganic fractions of P
Available phosphorus (Mehlich III (acid extractable) method)
Iron (Fe)
Aluminum (Al)
Calcium (Ca)
Magnesium (Mg)

* Laboratory analyses were performed by the Soil and Water Science Department, University of Florida (Analytical methods are listed in Appendix 7B).

For the second method, only the 0 to 10 cm layer of wetland surface soil was sampled. Both sampling methods were used at many sites. At each surface sample plot site, survey plots ranging in area from 25 m^2 to 200 m^2 (depending on vegetation community type) were set up within specific vegetation communities. Within each plot, five surface soil samples were collected. Soils were then composited and transported back to the laboratory on ice, and stored prior to analyses. Soils were analyzed for pH, conductivity, [OM], [TC], [TN], and [TP] at the Soil and Water Science Department, University of Florida. (analytical methods are listed in Appendix 7B).

3.5 LAKE MASS BALANCE BASED ON FULL1995NN HYDROLOGY

To evaluate additional biogeochemical loadings due to water withdrawals, we estimated the mass balance of carbon, nitrogen, and phosphorus for Lakes Winder and Poinsett. Discharge data were collected at one location for each of the lakes. For Lake Winder, discharge data were collected at station SJR LK Washington while for Lake Poinsett discharge data were collected at station SJR near Cocoa at Hwy 520. Daily discharges were recorded and total monthly discharges were calculated from those. We then examined monthly discharges categorized into three seasons based on historic water quality, water elevation, and discharge data in the USJRB. The seasons were: November through January (very low algal productivity, high water discharges, and high water color), February through May (moderate algal productivity, low water discharges, and low water color), and June through October (low algal productivity, high water elevations, high nutrient levels, and low water color). For our analyses, we used the wet season (June through October) only, as this was during the rising limb of the hydrograph when nearly all hypoxic events occurred. This is also the time when wetlands become inundated following the dry season and when most runoff and downstream loading occurs. Thus, any effects due to extending soil exposure time should manifest themselves during this time.

Water quality was sampled at four locations: Lake Winder inlet, Lake Winder outlet, Lake Poinsett inlet, and Lake Poinsett outlet. Each location was typically sampled once a month from 2005 through 2010. Water was sampled, prepared, preserved (if needed), transported to the laboratory, and stored prior to analyses. Water samples were analyzed for [TOC], concentration of ammonium nitrogen [NH_4-N], concentration of soluble reactive phosphorus [SRP], [TP], and [TKN]. (Analytical methods are listed in Appendix 7B)

Using both discharges and water quality data for each lake, we calculated mass loads into and out of Lakes Winder and Poinsett during calendar months June through October from 2005 through 2010. We calculated Lake Winder mass loads from 2006 through 2009, and for Lake Poinsett, from 2005 through 2010. Periods differ due to availability of data.

3.6 Effect of Water Loss From Soil Air-drying) on Carbon and Nutrient Release

We collected a bulk surface soil sample of approximately 0.04 m^3 of (0 to 10 cm depth) from wetlands on the floodplain of Lake Poinsett by scraping a 0.4 m area 10 cm deep. Three replicate subsamples were placed as a thin layer in plastic trays and exposed to air, at room temperature, in the laboratory. Subsamples were obtained from each of the trays at days 0, 2, 4, 8, 16, and 32.

Soil moisture, dissolved OM, and dissolved nutrient content were then measured. To determine [N], [P], and [C], soils were suspended in deionized water at a soil to water ratio of 1:10. Soil suspensions were then equilibrated for 1 hr under continuous shaking on a mechanical shaker and then centrifuged at 6,000 rpm. The supernatant liquid was removed and filtered through 0.45 µm filter. Filtered solutions were analyzed for NH_4-N, SRP, DOC, and TKN (analytical methods are listed in Appendix 7B). This study and associated analyses were performed by the Soil and Water Science Department, University of Florida.

3.7 Carbon, Nitrogen, and Phosphorus Releases

Additional soil samples were collected at various times for a series of carbon and nutrient release studies as described below. Figure 3–2 shows the sampling sites around Lakes Winder and Poinsett for these release studies.

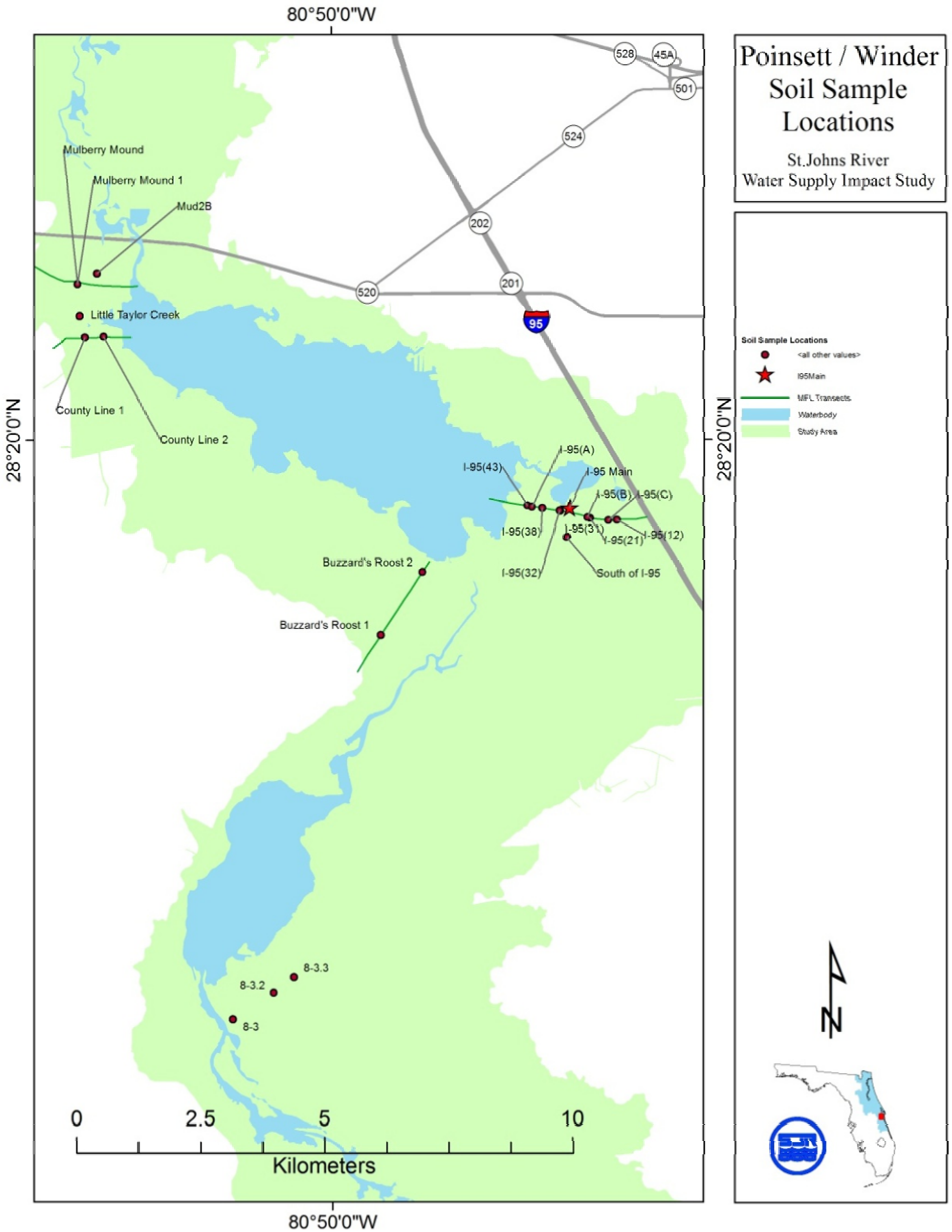

Figure 3–2. Sample collection sites for Lakes Winder and Poinsett showing minimum flows and levels (MFLs) transects.

3.7.1 Effect of Intact Soil Core Diameter on Carbon and Nutrient Release

Intact wetland soil cores were collected from the Little Taylor Creek site, northwest of Lake Poinsett. Soil cores with the following diameters were sampled in triplicate from the site: 5.1, 10.2, 15.2, 20.4, and 24.8 cm (2, 4, 6, 8, and 9.5 in.). Core tubes were pushed from the soil surface down to a soil depth of 30 cm (1 ft). Core tubes were then removed, capped, and transported back to the laboratory. In the laboratory, soil cores were flooded with site water (diluted by 50% with distilled water) to a depth 20 cm (7.9 in). Floodwater was then slowly bubbled with ambient air to ensure water column mixing during an initial 4-day flooding cycle. All soil cores were incubated in the dark at laboratory room temperature. Small aliquots (approximately 20 ml) of floodwater were collected, filtered through 0.45 µm filters, and prepared for analyses. Water samples were obtained at days 0, 1, 2, and 4 and analyzed for [NH_4-N], [SRP], and [TOC]. Additional floodwater samples were collected at days 0 and 4 for measurement of [TP], [DOC], electrical conductivity, and pH. At the end of 30 days, floodwater was removed, and the soil was reflooded again to a depth of 20 cm (7.9 in) with site water (diluted by 50% with distilled water and bubbled) Sampling and analyses followed the same methods as in the first flooding cycle.

At the end of the two flooding cycles, soil cores were sectioned into 0 to 10 and 10 to 20 cm (0 to 4 and 4 to 8 in.) depth increments. Soils were extracted with deionized water at a soil to water ratio of 1:10 for 1 hr under continuous shaking. Soils were then centrifuged and the supernatant liquid was filtered through 0.45 µm filters, and analyzed for [NH_4-N], [SRP], [DOC], [TKN], [TP], and pH. All experiments, laboratory sampling and analyses were performed by the Soil and Water Science Department at the University of Florida. (Analytical methods are listed in Appendix 7B). N, P, and OC releases from soil to overlying water were calculated as the difference between the mass areal storage (mg m^{-2}) in the water column on days 0 and 4 of the first flooding cycle. To estimate the release rates per day of additional soil exposure to air (mg m^{-2} d^{-1}), we assumed the release would be constant with time. With the first day the soil in the setland was exposed in this area representing day zero with a potential release rate due to oxidation from exposure of zero. Because release rates could decline through time as labile C is oxidized, this assumption could overestimate, but not underestimate later release rates.

3.7.2 Field Core Soil Analysis to Determine Carbon and Nutrient Releases

We collected triplicate intact soil cores (10.2 cm (4 in.) in diameter) from several locations at approximately 2-week intervals from 9 February 2010 to 9 March 2010—a dry period. All cores were transported to the laboratory for immediate analysis.

The cores were sectioned to collect the top 0 to 10 cm (0 to 4 in.). Soils were extracted with deionized water at a soil to water ratio of 1:10. The resultant slurry was mixed for a period of 1 hr under continuous shaking on a mechanical shaker, followed by centrifugation. Supernatant liquid was removed and filtered through 0.45 µm filters and analyzed for [DOC], [NH_4-N], [SRP], [TKN], and [TP]. The bulk inventory of each analyte for each period was calculated on a mg m^{-2} basis (measured mass divided by core surface area) to depth of 10 cm (4 in.) (analytical methods are listed in Appendix 7B).

3.7.3 FIELD CORE FLOODING TO DETERMINE CARBON AND NUTRIENT RELEASES

Another set of triplicate intact field cores (10.2 cm (4 in.) in diameter) were collected to a depth of approximately 30 cm (12 in.) at the same sites and times but were transported to the laboratory in ice and immediately flooded as laboratory microcosms. We considered the dry period for these cores to have begun at the time they were first exposed (drained of surface water) in the field prior to collection. An early spring storm flooded the site from 14 March 2010 until 24 May 2010. During the second drying event, which began 25 May 2010, we collected cores on 22 June 2010, and treated them by the same methods.

The mass, concentration times volume, of each nutrient released by the drying event was calculated as the difference in water column mass (mg m^{-2}) between days 0 and 4. The patterns of change in mass over the experimental period indicate that the day 4 values adequately represented the total release for each nutrient (Figure 3–3). Yurova et al. (2008) also found 2 to 4 days adequate for sorption/desorption equilibrium of [DOC]. The mass released for each analyte was plotted against the days of exposure prior to flooding to determine the rate of release rate for each day of exposure (slope of line).

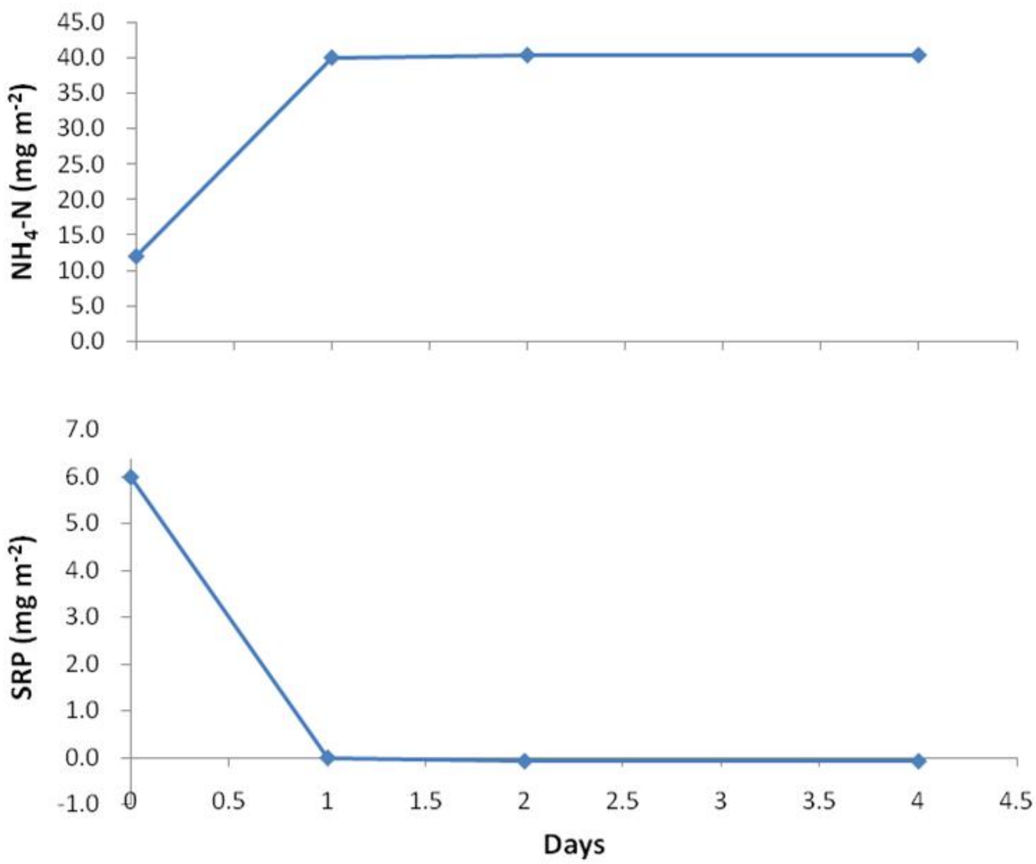

Figure 3–3. Variation in soluble reactive phosphorus (SRP) and ammonium nitrogen (NH_4-N) concentrations following flooding a field core collected after a period of exposure of the wetland. Each constituent 4-day flux curve was verified before use.

3.7.4 Site Water Correction and Dissolved Organic Carbon Release Rates

The release of DOC from these soils could not be determined in the same way as the release of NH_4-N, SRP, TP, and TKN. All the site water needed for the release rate studies was collected on 9 February 2009 and then diluted, by 50%, with deionized water. It was then stored in the dark at approximately 22°C until needed to flood the cores. A significant decline in [DOC] was observed in the site water from day 0 onward. This decline resulted in a variable release rate for DOC. An error in the data recording for two periods required correction using a second nonlinear regression (r^2 – 0.9995, $p < 0.0001$) to allow calculations of DOC release for field cores.

3.8 Models

3.8.1 Wetland Constituent Release Model

The Biogeochemistry Working Group developed the Wetland Constituent Release Model (see Appendix 7C), a Microsoft Excel® spreadsheet model that calculates potential additional C or nutrient mass release due to the difference between the base scenario (Base1995NN) and any other hydrologic scenario. The model uses a daily time step and a 1-cm water level resolution to calculate the additional exposed area of organic soil for a withdrawal scenario relative to the base scenario (Base1995NN).

The daily difference in area exposed is multiplied by the experimentally determined areal release rate for the specific constituent and by a temperature correction factor. The model sums the additional daily releases for each year. The Wetland Constituent Release Model, therefore, calculates a potential annual release due to the additional hectare-days of wetland exposure for each withdrawal scenario analyzed.

The areal release rates used in the model are constituent-specific and are taken from the work described in this chapter. To calculate total mass release, we used the largest statistically significant release rates (soil mass areal releases per day of exposure) that we measured.

3.8.2 Wetland Constituent Reduction Model

SJRWMD staff also developed the Wetland Constituent Reduction Model (see Appendix 7D), a Microsoft Excel® spreadsheet model that calculates the potential reduction of a released constituent while en route to the river. Reduction pathways include settling, sorption, chemical breakdown, biological uptake, and volatilization. Often multiple processes occur simultaneously. In the model, a removal coefficient lumps all of the potential reduction pathways into a single reduction rate coefficient. The model is derived from a Tanks-in-Series (TIS) Model as in Kadlec and Wallace (2008). The Wetland Constituent Reduction Model estimates the fraction of the mass of a released constituent from the Wetland Constituent Release Model that reaches the river (i.e. loading).

Because we had no onsite data, we examined values for C and nutrient reduction rate coefficients for various wetlands presented in EPA's North American Treatment Wetland Database (Knight et al. 1994) and in Kadlec and Wallace's (2008) analysis of 123 constructed and natural treatment wetlands. We compared the distribution of rate coefficients for all wetlands in Kadlec and Wallace (2008) to the distributions for natural wetlands in the EPA database, for natural marsh and pool systems (Knight 1994) including the Orlando Easterly Wetlands project, formally known as Iron Bridge based on its source water from the Iron Bridge waste water treatment facility.

We used biochemical oxygen demand (BOD) as a surrogate for DOC, because DOC is seldom measured in treatment wetlands. Additionally, we assumed that the DOC is mostly labile based on ongoing inhibition and growth experiments (Andrew V. Ogram, Soil and Water Science, University of Florida, pers. comm. 2009), that is, a good portion of the DOC was a BOD load. Our BOD reduction calculation also differs from other calculations because it was calculated from TIS model with one tank, whereas all other rates used three tanks (see Appendix 7D for discussion of TIS model). Kadlec and Wallace (2008) recommended using three tanks in series when there was no information to indicate otherwise, except for BOD, where most systems are better predicted using one tank.

Potential loading to the river is the mass of a constituent remaining after reduction in mass in transit to the river. Because the Wetland Constituent Reduction Model has nonlinear aspects, larger mass releases yield disproportionately higher mass reduction rates. Loading estimates for quite different masses, therefore, converge toward the minimum background concentration asymptotically. Thus, the loading estimates may vary less among scenarios than do the release estimates because this nonlinearity operates to diminish differences.

The output of the model is the portion—or load—of the released mass that reaches the lake or river. This load is distributed in a discharge-weighted manner across the season when water levels typically rise (June through October). By dividing the loading by the discharge volume through that same period, one can project the increase in concentration caused by the added load for each constituent.

3.9 Lake Water Quality

Lake Poinsett has three long-term water quality sampling sites located at the inlet (LPI), center (LPC), and outlet (LPO) of the lake. We used data collected from the center and outlet sites for water quality data analyses. Lake Winder has two water quality sampling sites: the inlet to Lake Winder (LWI) and the outlet (LWO). We used data from these two sites for our water quality analyses. The period of record that was used in water quality data analyses for both lakes was 1979 through 2010. However, for some water quality trend analyses, we used a period of record of 1995 through 2010.

Water samples from these sites were analyzed for various water quality parameters including [DO], [Chl-a], [DOC], [NH_4-N], nitrate and nitrite [NO_x], pH, soluble reactive phosphorus orthophosphate [SRP], Secchi depth, [TOC], [TP], and water temperature. All samples were collected, preserved, and/or filtered (as necessary), stored, and analyzed according to standard methods (see Appendix 7B).

3.10 Projected Changes in Dissolved Oxygen Concentration

Regression and multiple regression analyses were performed on the observed data to determine the best relationship for use in predicting potential changes in [DO] when using water elevation data and loading results from the Wetland Constituent Release and Reduction models.

4 RESULTS

4.1 DIGITAL ELEVATION MODEL RESULTS

The Wetland Vegetation Working Group developed a DEM for river segment 8 using corrected LiDAR data. The southern portion of the segment 8 DEM contains Lake Winder and the sub-basins draining into it, while the northern half contains Lake Poinsett and its sub-basins (Figure 4–1). Hypsographic relationships between area and elevation were determined for each of these halves (Figure 4–2). The elevation to area calculation gives the area at or below each centimeter of elevation above sea level (NGVD29).

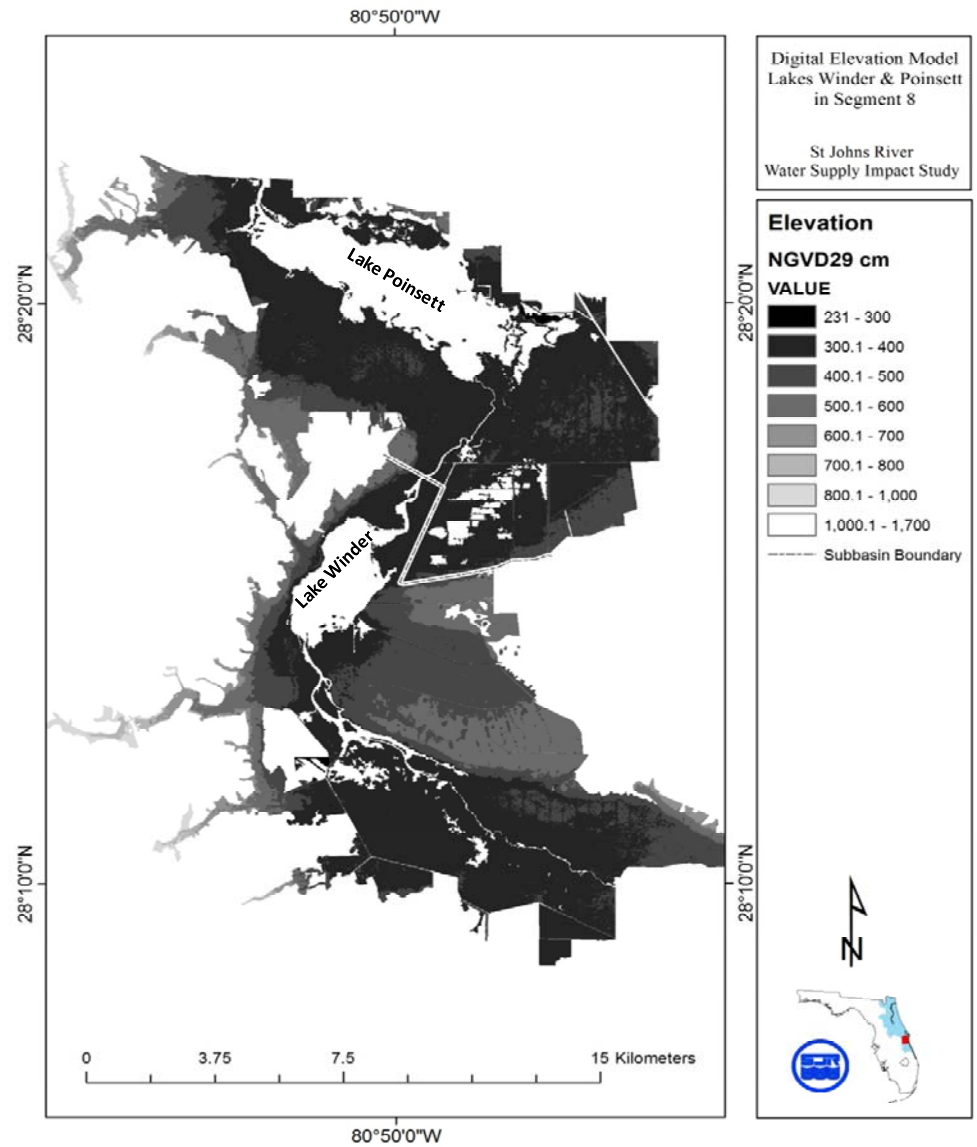

Figure 4–1. Digital Elevation Model showing wetland elevations for Lake Winder, Lake Poinsett, and the open water, which is indicated in white. The sub-basin boundary used to separate the different lakes and their associated wetlands for modeling purposes is also depicted.

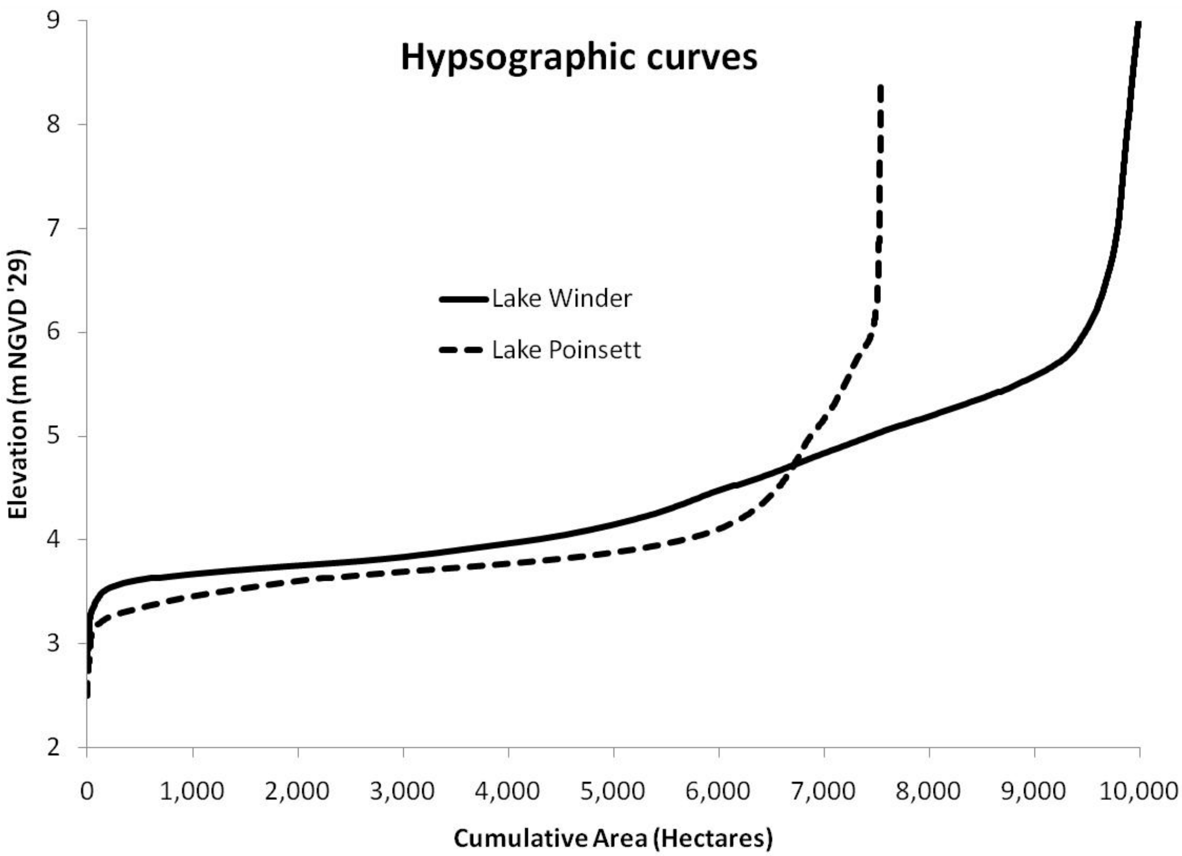

Figure 4–2. Hypsographic curves for Lakes Winder and Poinsett showing cumulative area below an elevation or area that would be inundated at a specific water elevation.

4.2 Assessment of River-Wetland Hydrology Results

As discussed in Section 2.2, for the purpose of assessing the environmental effects of surface water withdrawals on the St. Johns River, the river was divided into nine segments. Figure 4–3 shows the segments with their associated wetland areas and drainage basins.

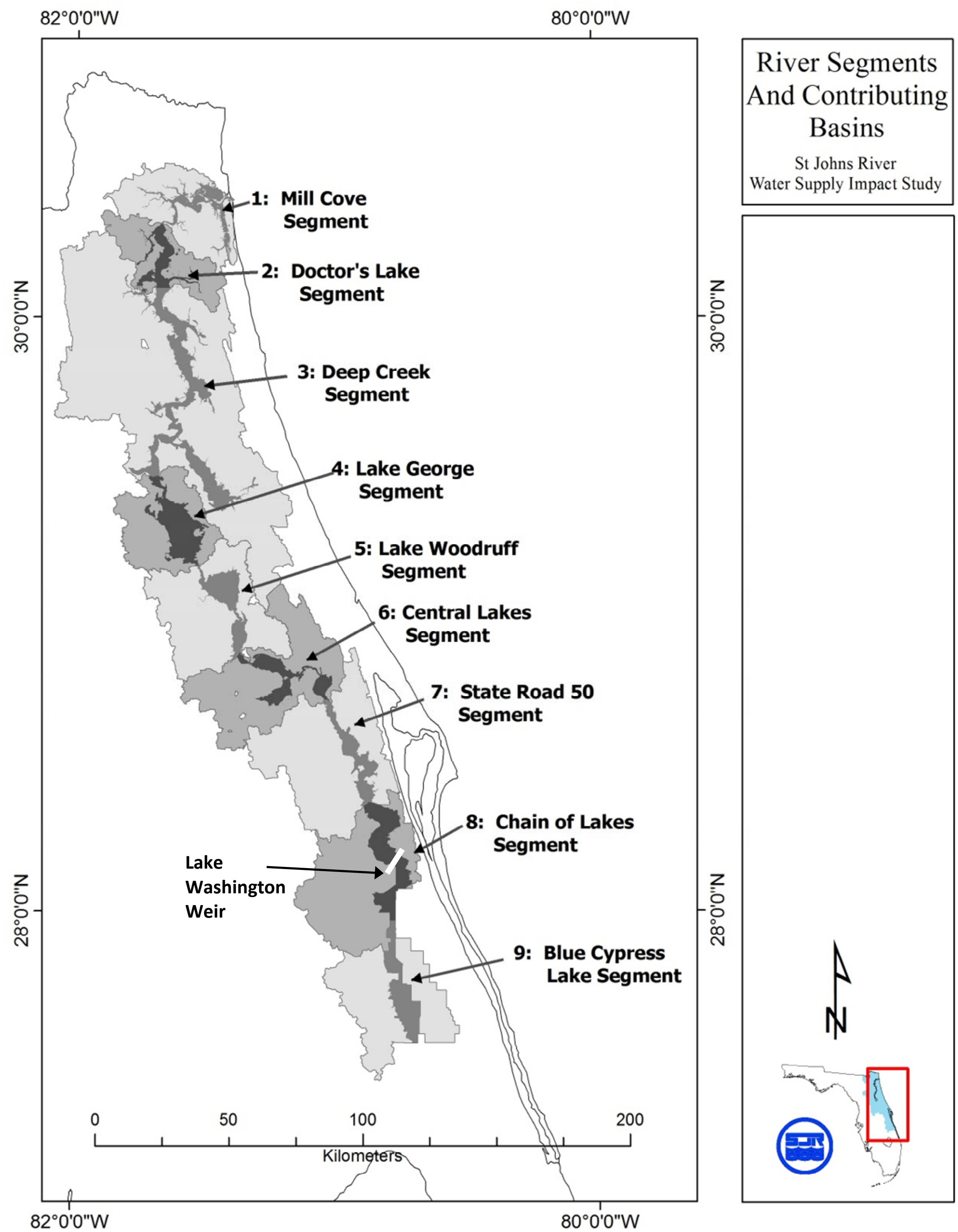

Figure 4–3. River segments in the St. Johns River with their associated wetland areas (shaded in dark grays) and drainage basins (shaded in light grays). The white line in segment 8 shows the approximate location of the Lake Washington weir, which is the southern terminus of the analyses of possible effects from water withdrawals.

The northern part of segment 8, downstream of the Lake Washington weir contains Lakes Winder and Poinsett. This portion of the segment has a large wetland area. In addition, it has a lower discharge and larger decrease in water elevation than any of the downstream segments. It also has similar C:N ratios with the segments 7 and 6. The C:N ratios are an estimator of how easily (rapidly) the SOM will decompose with lower numbers decomposing slower (Aitkenhead and McDowell 2000, Berg 2000, Brady and Weils 2008). Presumably segments 6, 7, and 8 would have similar release rates..

Segment 7 has 9% more wetland area although its water elevation reduction is only 80% of segment 8 and its discharge is 30% greater (Table 4–1). These differences are even more apparent for river segments 6 and 5. Normalizing by area to discharge ratio to river segment 8 (Lake Poinsett) makes the relative differences between the potential carbon and nutrient loading endpoints of all segments more apparent. River segment 8 has the largest ratio of wetlands (loading) to discharge (dilution), so exposure of its organic soils is most likely to affect river concentrations of OC and nutrients. All downstream segments are proportionally less influential. For these reasons, segment 8 should exhibit the greatest effects of water withdrawals.

Table 4–1. Relationship between herbaceous wetland area and discharge for river segments 8 through 5. Average carbon to nitrogen (C:N) ratios and average decrease in water elevation from Base to Full1995NN (most extreme) are presented.

Down-stream Discharge Site	River Segment	Area of Herbaceous Wetlands (ha)	Average Annual Discharge ($m^3 s^{-1}$)	Area to Discharge Ratio (ha $m^3 s^{-1}$)	Normalized to Poinsett	Average C:N Ratio (Synoptic Survey)	Average Decrease in Water Elevation (m)
Poinsett (Cocoa)	8	15,764	40	528	1.00	12.9	0.050
Harney Center	7	17,287	56	382	0.72	13.2	0.040
Sanford	6	6,816	73	114	0.22	13.1	0.029
Astor	5	6,374	107	69	0.13	16.0	0.015

The riverine organic wetland area to rising-limb discharge ratio followed the expected pattern: segment 8 had the highest ratio with steadily decreasing ratios moving downstream through segments 7, 6, and 5. Wetlands in each successive segment had a progressively smaller ability to affect the C and nutrient concentrations in the river because concentrations resulting from a given load diminish as river discharge increases. This type of elementary analysis allowed us to predict lesser effects from segment 8 through successive segments to 5. This provides the framework for applying an iterative approach to segments 8 through 5.

Continuing the iterative approach to segments 4 through 1, the water elevations predicted in the different scenarios are largely uncorrelated with discharge since water levels are mostly controlled by sea level in these segments (Figure 2–2). Consequently, these river segments are biogeochemically insensitive to the magnitude of surface water withdrawals since water elevation, the hydrologic environmental driver, for the relevant biogeochemical processes was not materially influenced by withdrawals.

4.3 HYDROLOGIC ANALYSES OF SCENARIOS RESULTS

Among the scenarios, an unrealistic (test) scenario (Full1995NN) showed the largest hydrologic deviation from the base scenario (Base1995NN). All realistic model scenarios included completion of USJRB projects and showed lesser reductions in water levels or increases in water level (Figure 4–4).

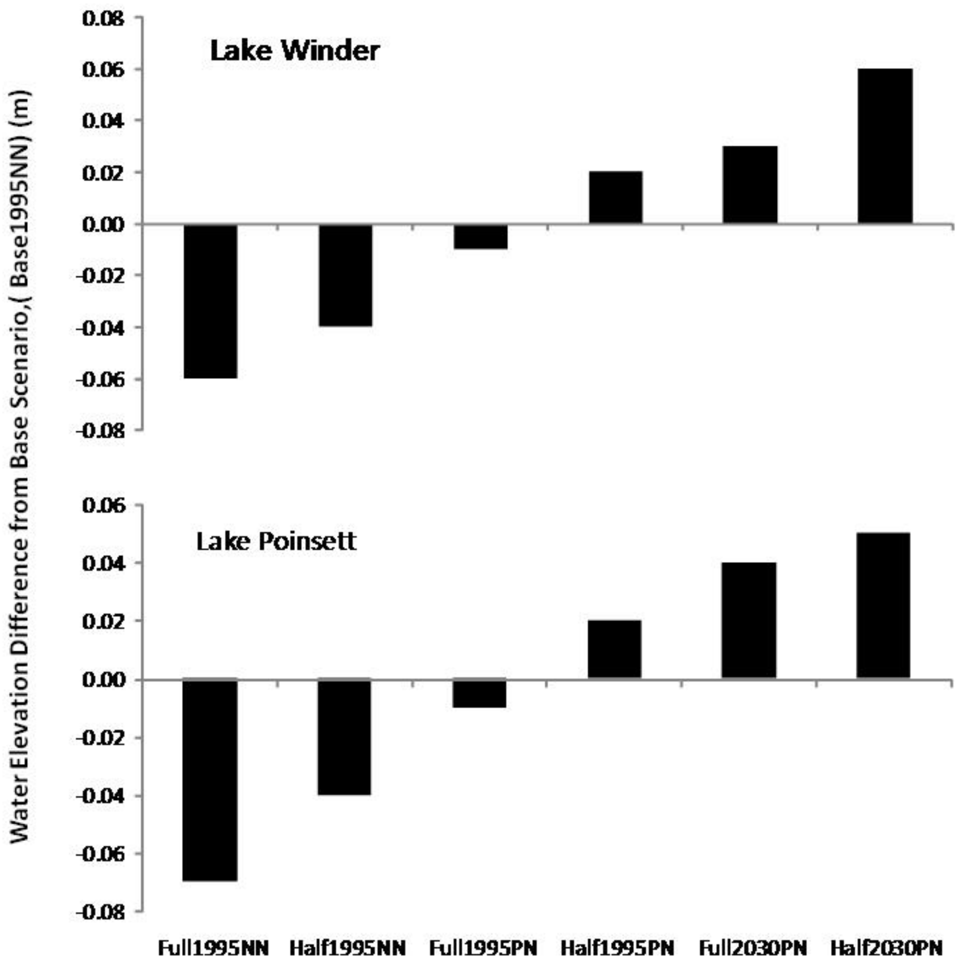

Figure 4–4. Average deviation from the base scenario (Base1995NN) in water elevation between the 20th and 80th percentile of water level exceedence frequency distribution for six withdrawal scenarios.

4.4 SYNOPTIC SURVEY OF WETLAND SOILS RESULTS

As described in methods (Section 2–4), wetland surface soil samples (0 to 10 cm depth; n = 80 and n = 86) were collected in various rivers segments throughout the St. Johns River. Summary statistics of biogeochemical soil characteristics are presented in Appendix 7A.

At Lake Poinsett, the I95-31 wetland site (i.e., the site closest to where most experimental data were collected) had little variation in soil organic matter and data distribution approximated normality (Figure 4–5). Data distributions for soil [TP], [OM], and [inorganic P fractions] also approximated normality. [Inorganic P] and [Mehlich III P], [TN], [TC], [DOC], and microbial biomass carbon (MBC) concentrations were in the midrange relative to all other wetland surface soils collected at Lake Poinsett during this study (Figure 4–6). Distribution maps, summary statistics, and raw data for a wide range of soil characteristics are provided in Appendix 7A.

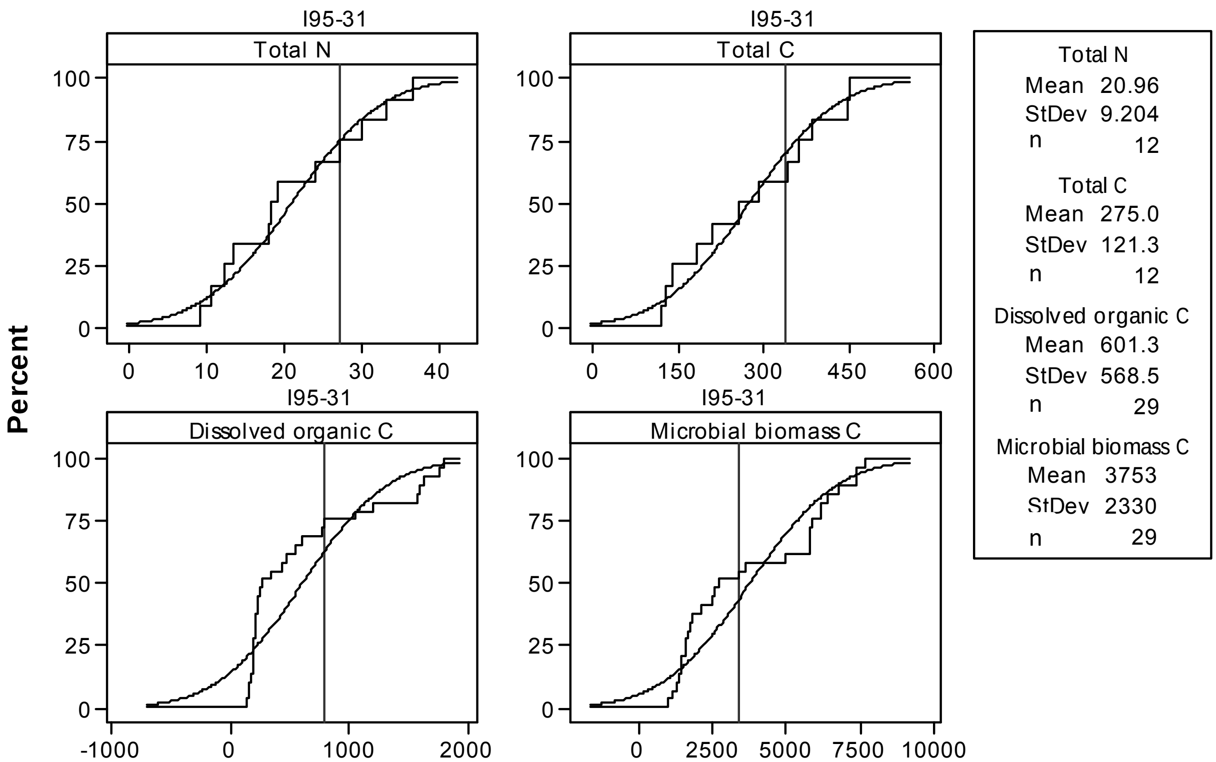

Figure 4–5. Cumulative distribution functions of wetland surface soil (0 to 10 cm) chemistry collected at Lake Poinsett. Percentiles are represented on the y-axis and nutrient concentrations (g kg^{-1} or mg kg^{-1}) are on the x-axis. Total nitrogen and total carbon are in g kg^{-1}; whereas, dissolved organic carbon (DOC) and microbial biomass carbon are in mg kg^{-1}. The smoothed hatched line represents a normal distribution curve for the data, while the solid stepped line represents actual collected data. The average nutrient concentration of wetland soils at the intensively sampled I95-31 site is represented in each panel as a vertical solid line. Some summary statistics for each respective parameter are presented on the right hand side of the figure. N = nitrogen, C = carbon, n = number of observations.

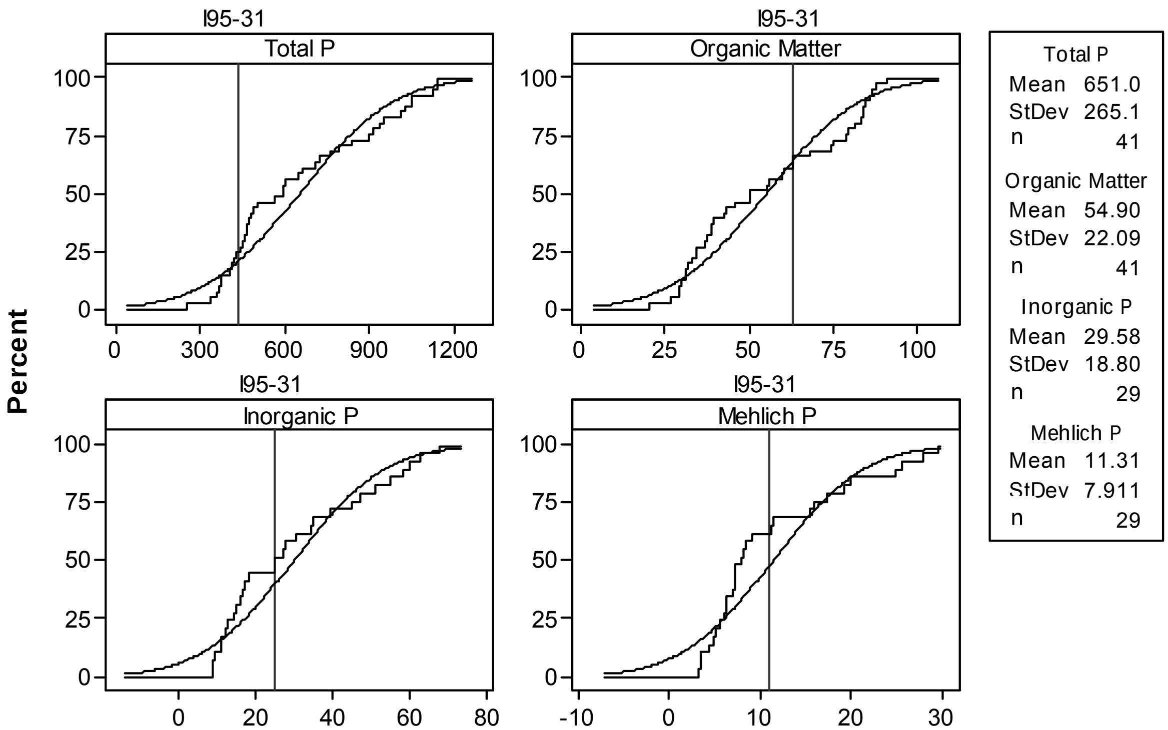

Figure 4–6. Cumulative distribution functions of wetland soil (0 to 10 cm) characteristics collected along entire length of the St. Johns River. Percentiles are represented on the y-axis and nutrient concentrations measured in mg kg^{-1} or percent are on the x-axis. Total phosphorus, inorganic phosphorus, and Mehlich III phosphorus are in mg kg^{-1}; whereas, soil organic matter is in percent (%). The smoothed hatched line represents a normal distribution curve fit, while the solid stepped line represents the observed data. The average phosphorus and organic matter concentrations of wetland soils adjacent to Lake Poinsett are represented by the vertical solid line in each panel. Some summary statistics are presented on the right hand side of the figure. P = phosphorus, n = number of observations.

The susceptibility of soil OM to decomposition was evaluated for surface soils (0 to 10 cm) based on the C to N ratio (C:N) of the soil. This ratio is related to the decomposition rate of the C components of a soil under drained conditions. According to Brady and Weil (2008), activity levels fall into three groups based on the C:N ratio:

- Active—C:N 15 to 30; decomposition over 1 to 2 yrs
- Slow—C:N 10 to 25; decomposition over 15 to 100 yrs
- Passive—C:N 7 to 10; decomposition over 500 to 5,000 yrs

Soils around Lake Poinsett and soils in downstream segments 7 and 6 have a low C:N ratio of approximately 13. These segments also have a low [OM], as determined by loss on ignition (LOI), relative to most histosol soils. Because of the low [OM] and the low C:N ratio, one can, a priori, predict very little oxidation and release from these soils due to exposure (see Table 3–1).

The primary releases of DOC and nutrients would be from detritus produced during the previous year, which is mostly labile. Only the core diameter study would have caught releases from decomposition of this recent production, because it was the only study that determined release from time zero at the onset of the dry season. Other studies either estimated release as the difference between two non-zero exposure periods, or were initiated after an unusual dry season flood. In both of the latter cases, the recent labile annual production would already have been oxidized. This conjecture is supported by our observation that the segment 8 C:N ratio was approximately 13 before the winter storm flooding and was approximately 10 afterward. This demonstrated rapid breakdown of the labile carbon fraction of the recent detritus and litter.

4.5 INLET/OUTLET LAKE MASS BALANCE BASED ON FULL1995NN HYDROLOGY RESULTS

To assess the significance of the added load, discharges into and out of Lakes Winder and Poinsett were modeled for years 1996 through 2008 for the test scenario (Full1995NN). Modeled discharges during the wet season (June through October) were similar each year (Table 4–2). Wet season discharges from Lake Poinsett (the upstream lake) were consistently higher than those from Lake Winder. Annual discharge from Lake Poinsett was highest in 2004 (104,830 ha-m) and lowest in 2000 (25,138 ha-m). The sum of lake discharge and modeled surface water withdrawals increased approximately linearly until 2005, but decreased in 2006 and 2007.

Table 4–2. Modeled discharges (Base1995NN) determined for inflow (in) and outflows (out) to and from Lakes Winder and Poinsett.

Year	Lake Winder			Lake Poinsett			
	In	Out		In	Out	Withdrawal	Discharge + Withdrawal
	Discharge (ha-m) per season (June-October)			Discharge (ha-m) per season (June-October)			
1996	23,291	23,556		28,757	29,263	3,616	29,430
1997	37,077	37,670		45,295	45,386	4,224	47,835
1998	36,818	37,376		43,702	43,709	2,877	45,022
1999	53,581	53,850		61,039	57,939	3,326	62,377
2000	21,268	21,628		26,680	25,138	2,617	26,893
2001	78,846	79,414		87,528	84,917	2,488	89,137
2002	74,048	73,912		78,792	76,104	3,498	81,526
2003	51,585	51,524		55,538	52,856	4,104	58,672
2004	100,792	101,096		108,232	104,830	2,948	109,679
2005	79,195	79,749		86,197	84,274	3,477	89,669
2006	22,565	22,502		27,426	25,409	3,410	28,469
2007	31,708	31,887		38,170	36,080	4,375	40,449
2008	81,400	82,454		92,790	92,165	3,227	95,211

We calculated mass loads using both modeled discharge and measured water quality data from the inlets and outlets of Lakes Winder and Poinsett. Both Lake Winder and Lake Poinsett accumulated DOC during the wet season periods between 2005 and 2008 (Table 4–3). For TKN, Lakes Winder and Poinsett were, on average, net sinks, whereas they were a net source of NH_4-N between wet season years 2001 and 2003 (see Table 4–1). However, in any given year, both lakes could be either a source or a sink for TKN and NH_4-N. Lake Winder typically released P during the wet season years at a rate of 4.5 mt yr^{-1}. However, Lake Poinsett typically retained P loads at an average rate of 17.4 mt yr^{-1}.

Table 4–3. Mass loads of C, P, and N inflows (In) and outflows (Out) for Lake Winder and Lake Poinsett during the rising limb of the hydrograph (June through October) for years 1996 through 2008. Years with partial or no data are represented with "—."

	Lake Winder		Lake Poinsett			
Year	**In**	**Out**	**In**	**Out**	**Withdrawal**	**Discharge + Withdrawal**
	DOC (mt)		**DOC (mt)**			
2004	—	—	—	—	—	—
2005	19,988	20,322	22,012	19,924	819	21,144
2006	6,421	6,070	7,366	6,320	829	7,012
2007	11,247	10,201	12,220	9,914	1,247	10,694
2008	27,269	13,028	28,216	26,505	916	27,481
	SRP (kg)		**SRP (kg)**			
2004	—	—	—	—	—	—
2005	—	—	8,281	5,337	143	5,888
2006	—	—	40,896	21,811	2,603	22,822
2007	—	—	43,052	33,269	4,412	33,703
2008	—	—	161,094	119,561	3,928	123,092
	TP (kg)		**TP (kg)**			
1996	16,145	20,098	24,887	10,896	1,048	11,014
1997	44,968	—	54,428	62,773	5,130	65,923
1998	10,360	64,399	82,209	46,537	2,608	44,157
1999	100,714	125,767	131,211	123,963	5,896	132,742
2000	59,240	40,587	53,232	28,217	2,631	29,669
2001	116,228	133,270	155,250	164,206	4,799	169,287
2002	116,723	142,073	153,321	176,520	7,669	201,052
2003	50,527	74,018	75,540	60,834	4,831	68,281
2004	114,856	119,109	145,938	106,547	2,369	108,885
2005	40,501	74,422	89,613	70,802	2,866	74,925
2006	52,913	52,887	74,190	40,971	4,968	43,240
2007	68,177	63,244	80,161	57,698	7,362	59,812
2008	184,163	124,204	244,297	188,922	6351	194,610

Year	Lake Winder In	Lake Winder Out	Lake Poinsett In	Lake Poinsett Out	Lake Poinsett Withdrawal	Lake Poinsett Discharge + Withdrawal
	TKN (kg)		TKN (kg)			
1996	236,994	352,539	455,719	201,314	19,671	205,223
1997	604,261	—	800,812	885,000	83,356	932,469
1998	163,338	708,488	836,975	745,878	46,955	760,859
1999	1,141,090	1,215,754	1,285,556	1,386,149	91,309	1,514,086
2000	462,613	461,835	526,072	581,836	60,509	626,568
2001	1,625,305	1,771,218	1,769,085	1,672,743	62,208	1,774,773
2002	1,484,168	1,425,854	1,556,135	1,550,805	70,888	1,687,247
2003	917,130	925,570	1,052,451	1,032,331	81,656	1,137,610
2004	1,466,261	1,505,582	1,633,382	1,643,809	49,539	1,734,184
2005	911,762	982,230	1,174,555	1,029,743	44,003	1,092,508
2006	452,178	455,478	630,694	475,250	60,088	512,297
2007	720,962	676,294	816,459	697,750	86,474	753,161
2008	1,465,734	824,014	1,731,370	1,630,123	56,305	1,698,886
	NH_4-N (kg)		NH_4-N (kg)			
1996	7,021	18,815	28,792	4,575	436	4,603
1997	22,335	—	50,927	44,082	3,104	46,486
1998	3,733	14,462	31,796	14,466	1,009	14,719
1999	31,985	67,773	76,699	93,619	2,864	93,398
2000	34,613	21,100	40,241	16,619	1,433	17,162
2001	99,057	85,750	137,740	221,240	5,697	224,791
2002	88,391	84,759	124,886	195,637	8,536	206,363
2003	21,272	44,464	59,682	75,442	5,658	84,400
2004	135,630	114,537	148,278	125,697	1,531	127,249
2005	16,078	19,852	41,730	42,517	1,869	43,705
2006	—	—	—	—	—	—
2007	—	—	—	—	—	—
2008	—	—	—	—	—	—

Note:
In = into Lakes Winder and Poinsett
Out = out of Lakes Winder and Poinsett
DOC = dissolved organic carbon
SRP = soluble reactive phosphorus
TP = total phosphorus
TKN = total Kjeldahl nitrogen
NH_4-N = ammonium nitrogen
— = years with partial or no data

Discharge-weighted C, N, and P concentrations were also calculated and, for most parameters, we report wet season values for both lakes for 1996 through 2008. Similar to the patterns we observed in mass loads, inflow-weighted concentrations of C and N were on average greater than outflow-weighted concentrations (Table 4–4). Outflow-weighted concentrations of NH_4-N were greater than inflow-weighted concentrations of NH_4-N. Between 1996 and 2008, Lake Winder had, on average, greater concentrations in outflows relative to inflows, whereas the opposite was true for Lake Poinsett. Discharge-weighted concentrations were important input parameters for assessing loading effects.

Table 4–4. Discharge weighted concentrations into (in) and out of (out) Lakes Winder and Poinsett during the rising limb of the hydrograph (June through October wet season) for a given year. Years with partial or no data are represented with "—."

Year	Lake Winder In	Lake Winder Out		Lake Poinsett In	Lake Poinsett Out	Lake Poinsett Withdrawal	Lake Poinsett Discharge + Withdrawal
	DOC (mg L^{-1})			DOC (mg L^{-1})			
2004	—	—		—	—	—	—
2005	25.2	25.5		25.5	23.6	23.6	23.6
2006	28.5	27.0		26.9	24.3	24.3	24.6
2007	35.5	32.0		32.0	30.7	28.5	26.4
2008	33.5	15.8		30.4	28.8	28.4	28.9
	SRP (mgL^{-1})			SRP (mg L^{-1})			
2004	—	—		—	—	—	—
2005	—	—		0.01	0.01	0.00	0.01
2006	—	—		0.15	0.08	0.08	0.08
2007	—	—		0.11	0.10	0.10	0.08
2008	—	—		0.17	0.13	0.12	0.13
	TP (mg L^{-1})			TP (mg L^{-1})			
1996	0.07	0.09		0.09	0.08	0.03	0.04
1997	0.12	—		0.12	0.14	0.12	0.14
1998	0.03	0.17		0.19	0.11	0.09	0.10
1999	0.19	0.23		0.21	0.21	0.18	0.21
2000	0.28	0.19		0.20	0.11	0.10	0.11
2001	0.15	0.17		0.18	0.20	0.19	0.19
2002	0.16	0.19		0.19	0.23	0.22	0.25
2003	0.10	0.14		0.14	0.12	0.12	0.12
2004	0.11	0.12		0.13	0.10	0.08	0.10
2005	0.05	0.09		0.10	0.08	0.08	0.08
2006	0.23	0.24		0.27	0.16	0.15	0.15
2007	0.22	0.20		0.21	0.18	0.17	0.15
2008	0.23	0.15		0.26	0.20	0.20	0.20
	TKN (mg L^{-1})			TKN (mg L^{-1})			
1996	1.0	1.5		1.58	1.47	0.54	0.70
1997	1.6	—		1.77	1.95	1.97	1.95

Year	Lake Winder		Lake Poinsett				
	In	Out		In	Out	Withdrawal	Discharge + Withdrawal
1998	0.4	1.9		1.92	1.71	1.63	1.69
1999	2.1	2.3		2.11	2.39	2.75	2.43
2000	2.2	2.1		1.97	2.31	2.31	2.33
2001	2.1	2.2		2.02	2.02	2.50	1.99
2002	2.0	1.9		1.98	2.04	2.03	2.07
2003	1.8	1.8		1.89	1.95	1.99	1.94
2004	1.5	1.5		1.51	1.57	1.68	1.58
2005	1.2	1.2		1.36	1.22	1.27	1.22
2006	2.0	2.0		2.30	1.83	1.76	1.80
2007	2.3	2.1		2.14	2.16	1.98	1.86
2008	1.8	1.0		1.87	1.77	1.74	1.78
	NH_4-N (mg L^{-1})			**NH_4-N (mg L^{-1})**			
1996	0.03	0.08		0.10	0.03	0.01	0.02
1997	0.06	—		0.11	0.10	0.07	0.10
1998	0.01	0.04		0.07	0.03	0.04	0.03
1999	0.06	0.13		0.13	0.16	0.09	0.15
2000	0.16	0.10		0.15	0.07	0.05	0.06
2001	0.13	0.11		0.16	0.27	0.23	0.25
2002	0.12	0.11		0.16	0.26	0.24	0.25
2003	0.04	0.09		0.11	0.14	0.14	0.14
2004	0.13	0.11		0.14	0.12	0.05	0.12
2005	0.02	0.02		0.05	0.05	0.05	0.05
2006	—	—		—	—	—	—
2007	—	—		—	—	—	—
2008	—	—		—	—	—	—

Note:
In = into Lakes Winder and Poinsett
Out = out of Lakes Winder and Poinsett
DOC = dissolved organic carbon
SRP = soluble reactive phosphorus
TP = total phosphorus
TKN = total Kjeldahl nitrogen
NH_4-N = ammonium nitrogen
— = years with partial or no data

4.6 Effect of Soil Water Loss (Air-drying) on Carbon and Nutrient Release Results

We found relationships between the moisture content of wetland soils collected from Lake Poinsett and the number of days those wetland soils were exposed to air on trays in the laboratory (Figure 4–7). After 8 days of exposure at room temperature, moisture content of wetland soils declined from approximately 68% to approximately 57%. At 16 days of exposure, soil moisture dropped significantly to about 18% (Figure 4–7).

In wetland soils exposed in the laboratory, the water extractable fractions of P and C typically increased as soil moisture content declined (Figure 4-8). This was especially true after soil moisture dropped below 50%. However, the NH_4-N concentrations decreased when soil moisture content declined. There were no obvious patterns between wetland soil total Kjeldahl nitrogen and moisture content.

Moisture content in wetland soils that underwent exposure in the field (i.e., nonflooded conditions) ranged between approximately 55% and 80%, and this moisture content range was independent of exposure time.

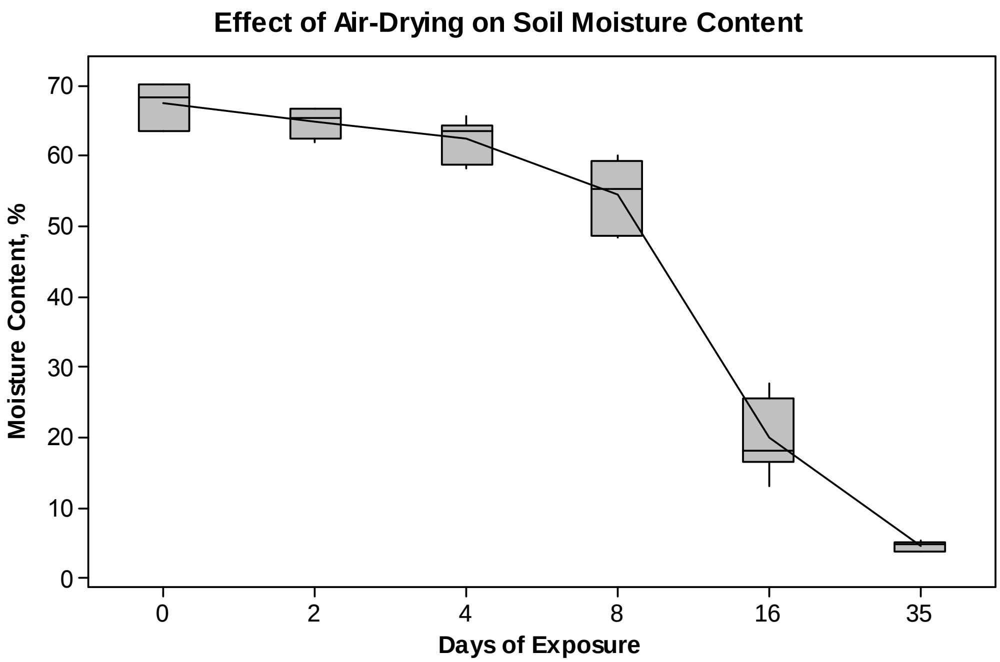

Figure 4–7. Soil moisture content plotted against days of exposure to air in the laboratory. Center lines are medians, lined circles are means. The box represents the 1st and 3rd quartiles (25th and 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box,

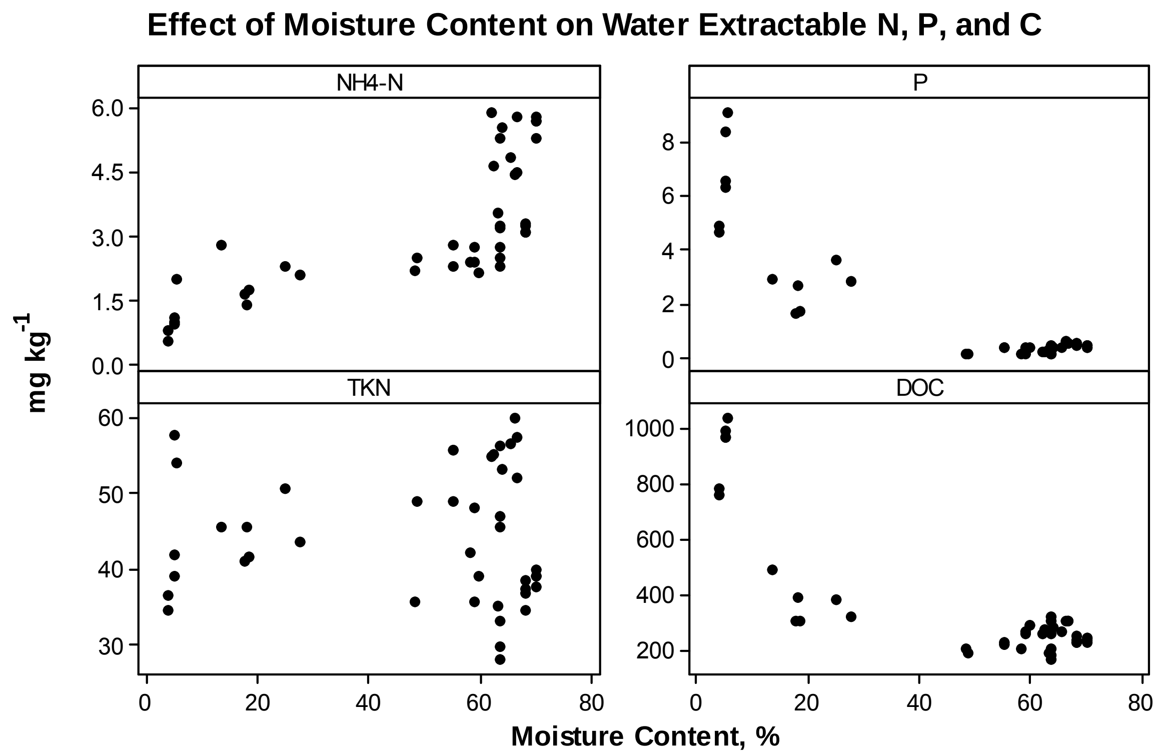

Figure 4-8. Nutrient concentrations (mg kg^{-1}) measured in re-flooded wetland bulk surface soil (0 to 10 cm depth) samples after various days of exposure in the laboratory, plotted against soil moisture content. N = nitrogen, P = phosphorus, C = carbon, NH_4-N = ammonium nitrogen, TKN = total Kjeldahl nitrogen, DOC = dissolved organic carbon.

4.7 Carbon, Nitrogen, and Phosphorus Releases Results

4.7.1 Influence of Core Diameter

All soil cores for the core diameter study (0 to 30 cm depth) were taken at the Little Taylor Creek site located northwest of Lake Poinsett near the county line transect. During the first flooding cycle of the cores, we found that soils collected with the smallest core diameter tubes (5 cm), when flooded, tended to release the least amounts of TKN, TP, and TOC (Figure 4–9). However, this difference was not significant ($p > 0.05$). Releases of SRP, DOC, and TKN tended to increase, as core diameter increased, but this increase also was not significant.

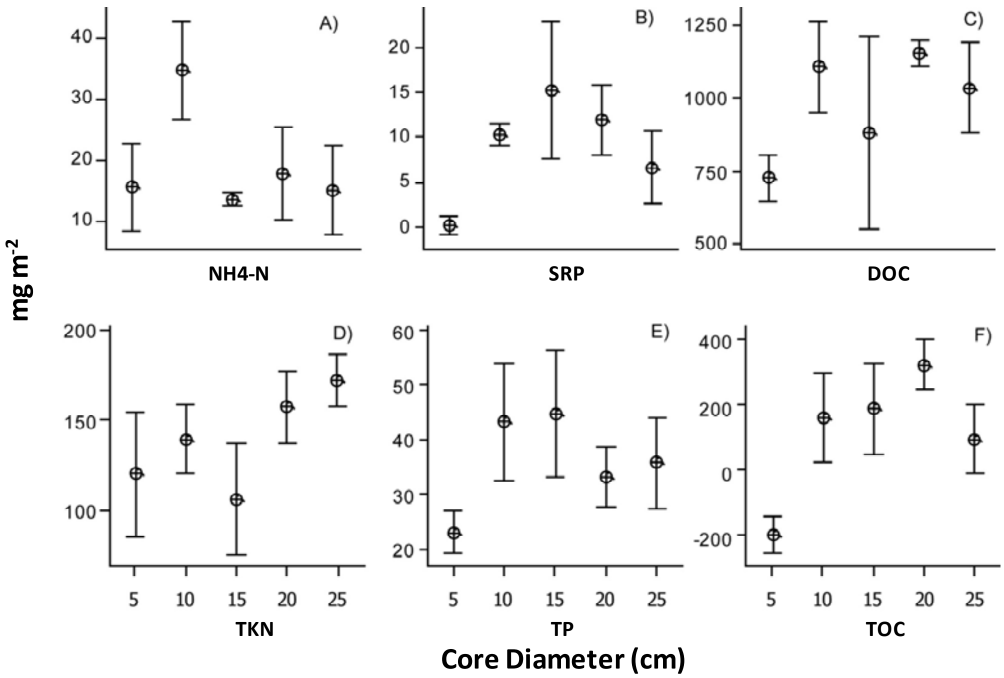

Figure 4–9. Average concentration results from diameter study showing standard errors (n = 3). (A) Ammonium nitrogen = NH_4-N, (B) soluble reactive phosphorus = SRP, (C) dissolved organic carbon = DOC, (D) total Kjeldahl nitrogen = TKN, (E) total phosphorus = TP, (F) total organic carbon = TOC.

After the end of two 30-day flooding cycles, soils from the core diameters were analyzed for water extractable nutrients (Table 4–5). The moisture content of all soils was approximately 70%. And there were no significant difference between the water extractable nutrient concentrations in soils based on different core diameter sizes.

Table 4–5. Mean Moisture content and soil nutrient concentrations for cores of different diameters. (n=3).

Core Diameter Category	Actual Core Diameter (cm)	Moisture Content (%)	NH_4-N	SRP	DOC	TKN	TP
			mg kg^{-1} dry weight				
5	5.1	77	9	3	392	108	6
10	10.2	71	5	2	268	71	5
15	15.2	70	4	2	326	61	4
20	20.3	73	5	2	373	81	5
25	24.8	71	5	2	378	107	6

Note:
NH_4-N = ammonium nitrogen
SRP = soluble reactive phosphorus
DOC = dissolved organic carbon
TKN = total Kjeldahl nitrogen

The results of the water analysis from the first 4-day flooding period in the diameter study were also used to calculate a combined release rate (Figure 4–10 and Table 4–6). These values followed an exposure period of 61 days. Combining all diameters in results give 15 observations per constituent (n = 15).

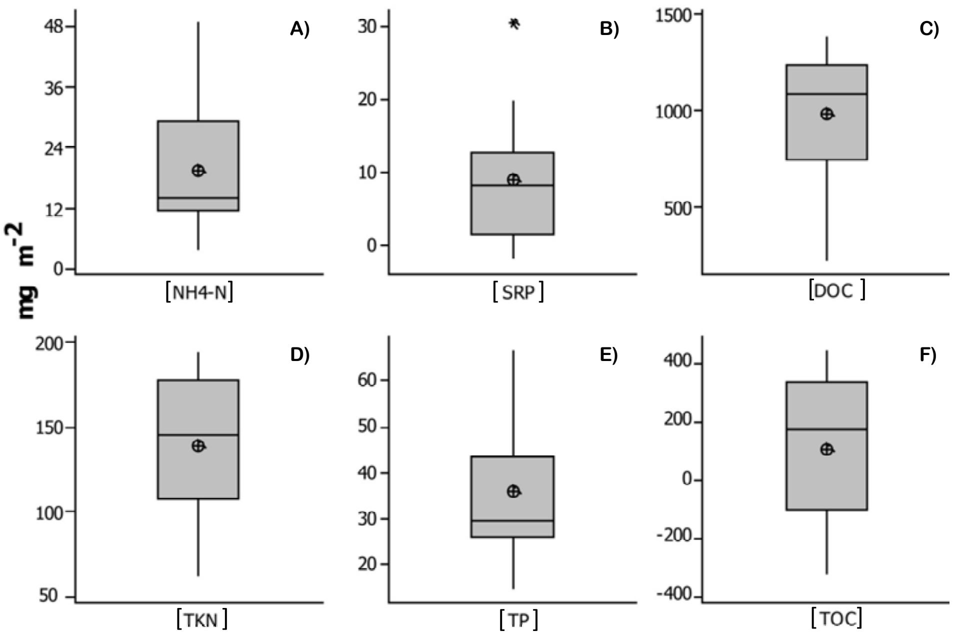

Figure 4–10. The release of nutrients (mg m^{-2}) to overlying water during 4 days of flooding of soil cores with diameters of 5 to 25 cm. Nutrient release from soil to water is the difference between areal storage (mg m^{-2}) in the water column on days 0 and 4 during the first flooding cycle. Center lines are medians, lined circles are means. The box represents the 1st and 3rd quartiles (25th and 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box, (n = 15). (A) Ammonium nitrogen = NH_4-N, (B) soluble reactive phosphorus = SRP, (C) dissolved organic carbon = DOC, (D) total Kjeldahl nitrogen = TKN, (E) total phosphorus = TP, (F) total organic carbon = TOC.

Table 4–6. Carbon, nitrogen, and phosphorus releases per day of exposure estimated from core diameter study soils. Estimates were determined by dividing the mass release after flooding for 4 days by the number of days of exposure in field.

Variable	Regression fit ($r^{2)}$	Days of Exposure	n Number of observations	Release rate ($mg\ m^{-2}\ d^{-1}$)
[DOC]	$r^2 = 0.841$	61	30	16.7
[NH_4-N]	$r^2 = 0.545$	61	30	0.32
[SRP]	$r^2 = 0.376$	61	30	0.14
[TKN]	$r^2 = 0.841$	61	30	2.28
[TP]	$r^2 = 0.755$	61	30	0.59

Note:
DOC = Dissolved organic carbon
NH_4-N = Ammonium nitrogen
SRP = Soluble reactive phosphorus
TKN = Total Kjeldahl nitrogen
TP = Total phosphorus
r^2 = Coefficient of determination

4.7.2 Field Core Soil Analysis to Determine Carbon and Nutrient Releases

An unseasonal extreme storm event flooded the wetlands from 14 March 2010 to 24 May 2010. This flooding, which occurred in the middle of the typical dry season, ended many of our studies early and cut short the number of sampling events for collecting exposure data. We are only presenting results that used the pre-storm data.

No significant differences were observed in the soil inventory of water extractable nutrients in the field cores with respect to duration of exposure. The average mass of NH_4-N remained around 1,000 mg N m^{-2} throughout the exposure period (95 to 125 days) with a p-value of 0.625 (Figure 4–11A). SRP averaged 10 mg P m^{-2} throughout the study with a p-value of 0.912 (Figure 4–11C). Slight increases were observed in TKN and TP. However these increases were not significant (p-values = 0.367 and 0.372, respectively; Figure 4–11B and D).

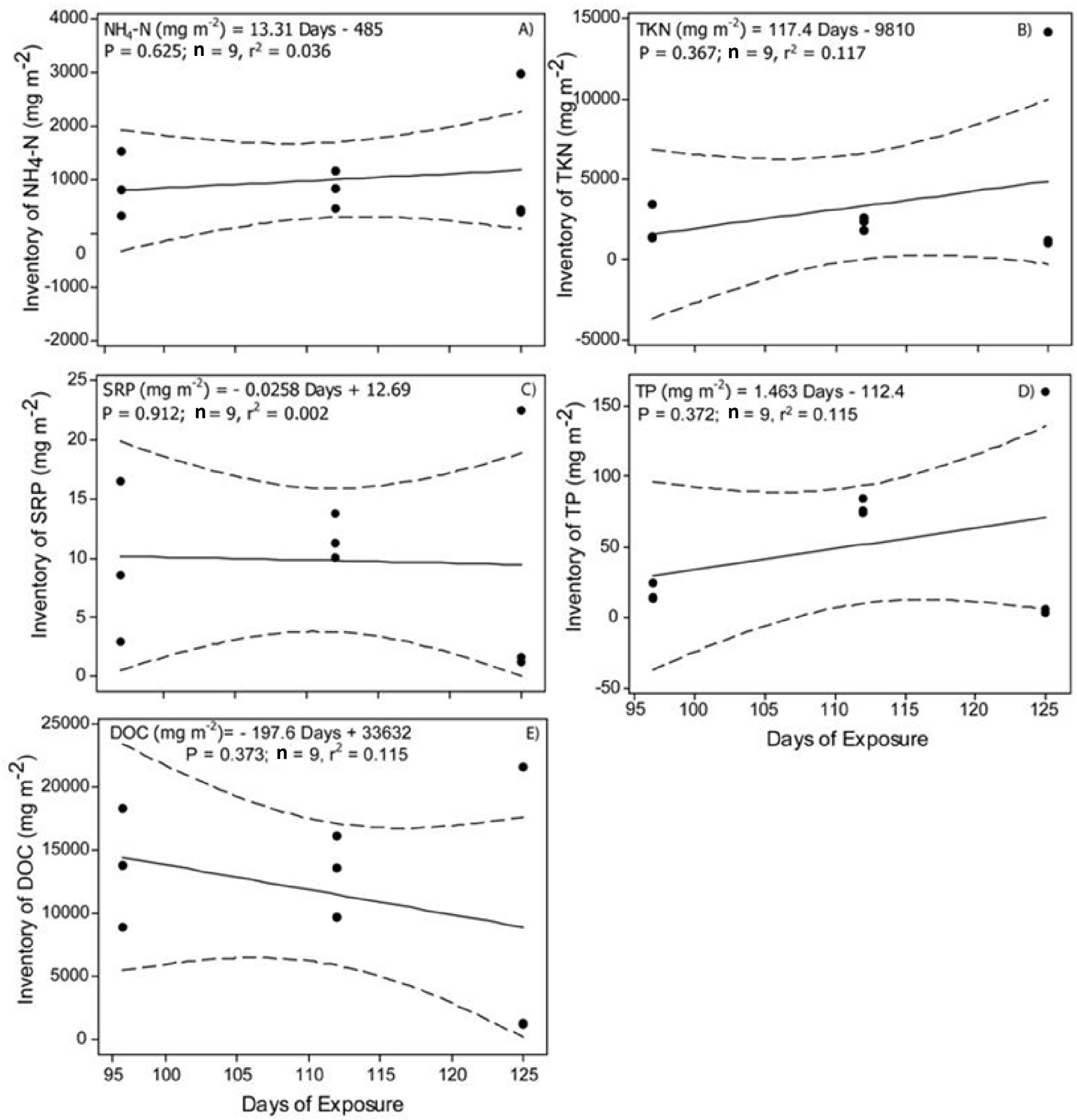

Figure 4–11. Extractable nutrients for soil cores collected after various days of exposure in the field and measured as the bulk mass of water extractable nutrients in the top 10 cm of soil. (A) ammonium nitrogen (NH_4-N), (B) total Kjeldahl nitrogen (TKN), (C) soluble reactive phosphorus (SRP), (D) total phosphorus (TP), (E) dissolved organic carbon (DOC). r^2 = coefficient of determination, p = probability, n = number of observations

4.7.3 Field Core Flooding to Determine Carbon and Nutrient Releases

This study suffered the same reduction of usable sample dates as the inventory study due to the extreme storm event that flooded the wetlands from 14 March 2010 to 24 May 2010, as described in Section 3.7.2. We are only presenting results that used pre-storm data.

The field core flooding study showed significant declines in the release rate of [NH_4-N] with duration of prior exposure; the average decline was from 134.6 to 13.8 mg N m^{-2} and the rate of change was -4.34 mg N m^{-2} day^{-1} (p-value < 0.001, r^2 = 0.81; Figure 4–12A). However, significant changes were not found to occur in the release rates of any other constituents. Soluble reactive P remained constant at -6.1 mg P m^{-2} (Figure 4–12C), TKN averaged 1,842 mg N m^{-2} through the study period (p-value = 0.718, Figure 4–12B), and TP averaged 307 mg P m^{-2} (p-value = 0.465, Figure 4–12D).

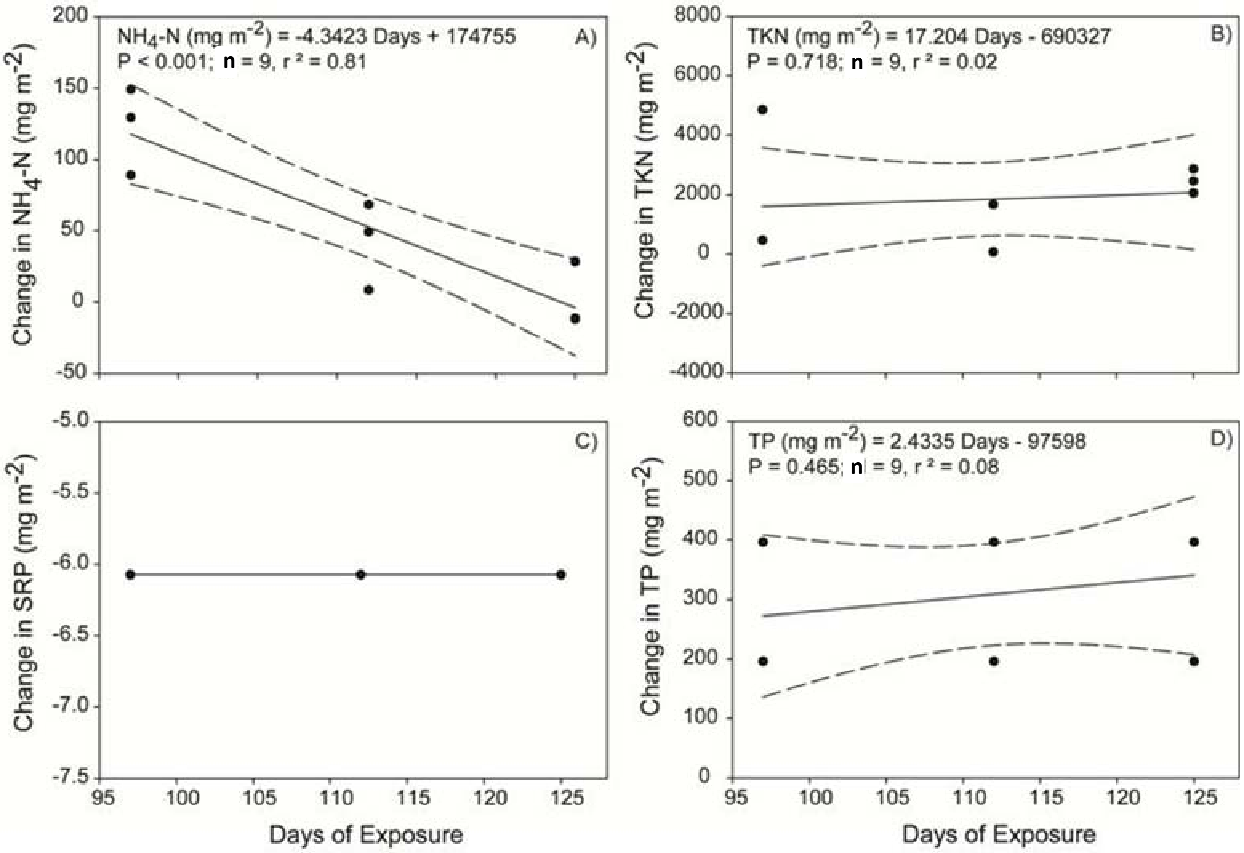

Figure 4–12. Nutrient release for field cores collected after various days of wetland exposure, which were immediately flooded and measured as the difference in the bulk mass of nutrients in the water column between day four and day zero. (A) NH_4-N = ammonium nitrogen (B) TKN = total Kjeldahl nitrogen, (C) SRP = soluble reactive phosphorus, (D) TP = total phosphorus. r^2 = coefficient of determination, p = probability, N = number of observation.

4.7.4 Release of Dissolved Organic Carbon from Flooded Cores

It was apparent that a laboratory error occurred in the analysis of samples collected on the 2nd and 3rd sample dates for the site water. All of the constituents recorded had exactly the same values for the first three sample periods Figure 4–13A) indicating that the [DOC] was not recorded correctly. The incorrect values were replaced with predicted values, which were used in all further analysis of [DOC] (p-value < 0.0001; r^2 = 0.9995; Figure 4–13B).

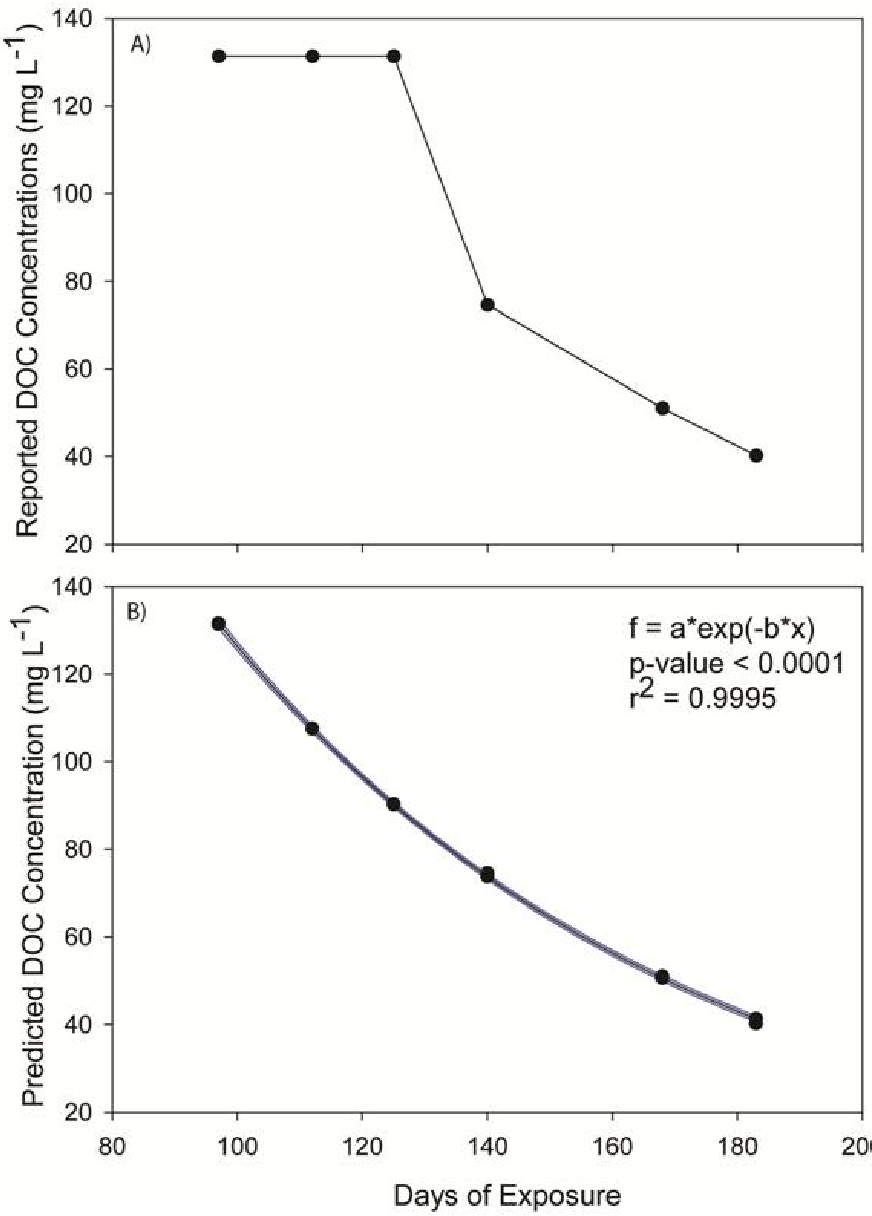

Figure 4–13. (A) Reported site water dissolved organic carbon (DOC) concentrations added to each intact core for release experiments and how it degraded with time. (B) Corrected site water (2nd and 3rd collection dates) DOC by use of a nonlinear regression, where a = 498.02 and b = 0.014. r^2 = coefficient of determination.

Over the term of the study, [DOC] decreased in the stored site water that was used for flooding laboratory cores, indicating that DOC consumption, sorption, or precipitation occurred even though the site water was stored in the dark. The first intact field cores collected were flooded with very high [DOC] site water, and these cores showed uptake, rather than release, of DOC (Figure 4–14). With each subsequent field core collected, [DOC] declined in the stored site water. This could be due to microbial breakdown, adsorption to soil or container walls, or precipitation reactions. Regardless of the reason, the field cores inundated with low [DOC] in the site water showed greater releases of [DOC].

The declining [DOC] in stored water probably accounts for the observation that the field cores collected for the first exposure period, when DOC was high, (97 to 125 days of exposure) resulted in uptake of DOC from the water column, whereas, the field cores collected after the second exposure period (30 days of exposure), when [DOC] in the stored water was low, resulted in release of DOC to the water column (Table 4–7). Others have noted similar behavior in different systems (Tao et al. 2000, Qualls and Richardson 2003, Reddy and DeLaune 2008, Kadlec and Wallace 2008, Vidon et al 2010).

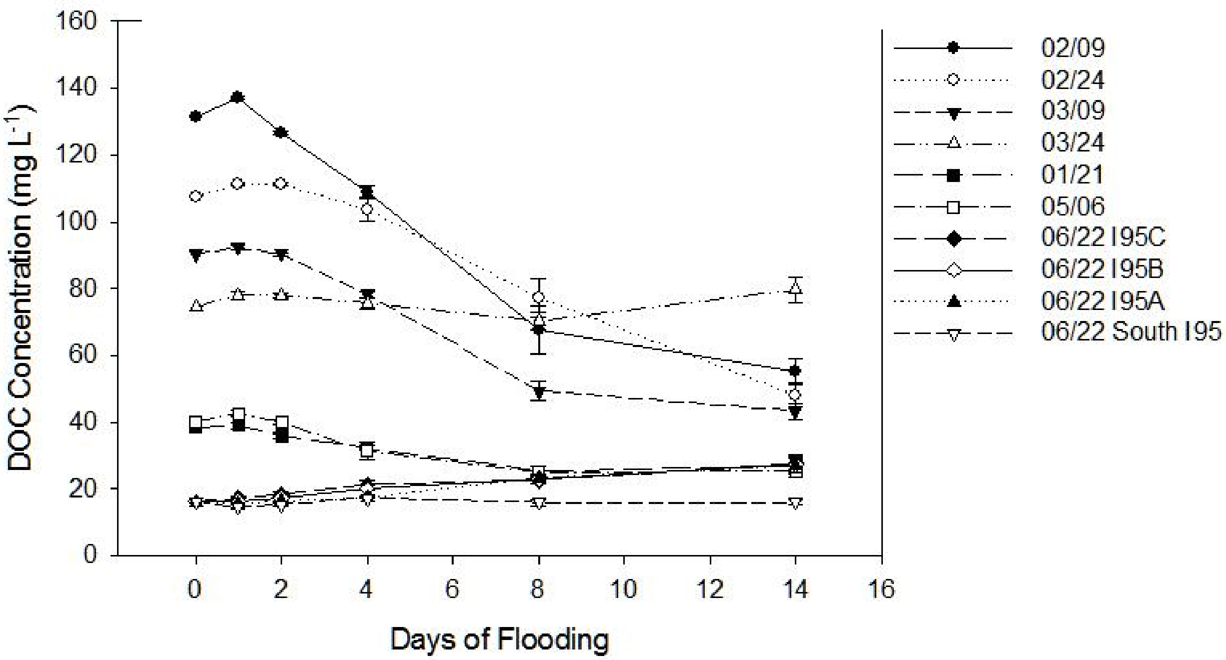

Figure 4–14. Effects of the decline in the initial dissolved organic carbon [DOC] of the site water collected for the incubation of the field cores. Each line represents the average change in [DOC] of one set of cores collected on a given date with standard errors (n=3) and incubated for 14 days. Cores from dates 9 February 2010 to 6 May 2010 were incubated with the same site water, which was collected on 9 February 2010. All cores collected 22 June 2010 were incubated with new site water collected on 22 June 2010.

Table 4–7. Dissolved organic carbon (DOC) release for each site with varying days of exposure. Release was determined as the difference in [DOC] between day 0 and day 4 during a bench-scale incubation study on intact cores.

Station	Date Collected	Days of Exposure*	Average Mass Release (mg m^{-2})	Standard Error SE	Additional Release† (mg m^{-2} d^{-1})	Site Water Condition‡
First exposure						
I95 main	09/02/10	97	-4,471	(±406)	-46.1	131.5
I95 main	24/02/10	112	-5,576	(±714)	-49.8	107.5
I95 main	09/03/10	125	-10,694	(±214)	-85.6	90.3
Second exposure						
I95C	22/06/10	30	1,032	(±236)	34.4	16.2
I95B	22/06/10	30	763	(±66)	25.4	16.2
I95A	22/06/10	30	227	(±39)	7.6	16.2
South of I95	22/06/10	30	225	(±40)	7.5	16.2
Marsh average	22/06/10	30	562	(±40)	18.7	16.2

* Number of days since standing water was measured on soil surface
† Additional release for marsh because the water levels were above the soil surface
‡ Indication of high dissolved organic carbon (DOC) concentration in site water added to cores

4.7.5 SUMMARY OF CARBON AND NUTRIENT RELEASE RATE ESTIMATES FOR MODELING

Areal release rates (mg m^{-2} d^{-1}) varied by study and by the number of days soils were exposed to air (Table 4–8). For example, DOC was typically retained by soil cores in the core diameter study, but was released from soil during portions of the field core and soil inventory studies.

Table 4–8. Carbon, nitrogen, and phosphorus release per day of exposure estimates from the soil core diameter study (diameter), field core study (field core), and soil inventory approach (inventory). Release rates shaded in grey were used as inputs to the Wetland Constituent Release Models.

Variable	Study	Days of Exposure	n	Release per Days of Exposure ($mg\ m^{-2}\ d^{-1}$)
[DOC]	Diameter	61†	30	16.1*
	Field Cores	30†,‡	12	18.7*
	Field Cores	97, 112, 125§	9	-218*
	Inventory	97, 112, 125§	9	-198
[NH_4-N]	Diameter	61†	30	0.32*
	Field Cores	30	12	1.4*
	Field Cores	97, 112, 125	9	-4.3*
	Inventory	97, 112, 125	9	13
[SRP]	Diameter	61†	30	0.14*
	Field Cores	30	12	0
	Field Cores	97, 112, 125	9	0
	Inventory	97, 112, 125	9	0.03
[TKN]	Diameter	61†	30	2.28*
	Field Cores	30**	12	125*
	Field Cores	97, 112, 125	9	17.2
	Inventory	97, 112, 125	9	117
[TP]	Diameter	61†	30	0.59*
	Field Cores	30	12	-0.3
	Field Cores	97, 112, 125	9	2.4
	Inventory	97, 112, 125	9	1.5

* Significance is denoted with an asterisk

† The release rates for diameter and field cores for 61 and 30 days of exposure, respectively were determined greater than zero at the $p < 0.05$ level.

‡ The exposure of field cores for 30 days occurred after an inundation period of about 60 days, which followed the previous 125, 112, and 97 days of exposure

§ Release per day of exposure for both field cores and inventory were determined as the linear slope between areal release from soils ($mg\ m^{-2}$) and exposure days.

** TKN for field cores had extremely high values. Values not possible given the [DOC], so these values were not used

Note:

n	=	sample size
[DOC]	=	concentration of dissolved organic carbon
[NH4-N]	=	concentration of ammonium nitrogen
[SRP]	=	soluble reactive phosphorus
[TKN]	=	total Kjeldahl nitrogen
[TP]	=	total phosphorus

Results for NH_4-N and SRP in the core diameter study were not used. As explained earlier, Day zero in the field was defined to have a release of zero. The result is that one half of the data going into the regression was defined. Therefore, we used a higher standard to help balance out the inclusion of the defined data: a regression r^2 value of 0.75 or greater. The r^2 value for NH_4-N and SRP both fell below this threshold. Measurements of [TKN] from the field core study were much higher than those for [DOC], indicating errors in the analyses. We cannot be certain of the sources of the error(s) but, given the consistency of the [DOC] measurements among cores and sites, the TKN values were likely erroneous and were not used in modeling. Therefore, the TKN release rate from the core diameter study was used in modeling. We did use release rates of DOC and TP per days of additional exposure. These rates were generated using the field core study and the core diameter study, respectively (See Table3-8).

4.7.6 SUMMARY OF CARBON AND NUTRIENT REDUCTION RATE COEFFICIENT ESTIMATES FOR MODELING

Because we had no measured reduction rate data from this study or other onsite studies, we used literature values for C and nutrient reduction rate estimates. The reduction rate coefficient estimates were taken from the EPA's North American Treatment Wetland Database (Knight et al. 1994). We compared the distribution of all the wetlands presented by Kadlec and Wallace (2008) against the natural wetlands in the database, then the subset of the marsh and pool natural systems, and finally only the Orlando Easterly Wetlands (OEW), formally known as Iron Bridge. The results are shown in Figure 4–15.

The reduction rate coefficients varied among all wetlands, natural wetlands, marsh and pool systems, and the OEW (Figure 4–15). We used statistics of reduction rate coefficient distribution for OEW because OEW is close (< 30 km) to Lake Poinsett and has similar habitats (i.e., herbaceous and pond habitats). The rates for OEW are also similar to those for natural systems and for natural freshwater surface flow marsh hybrid systems (NAT-FWS-MAR-HYB) subsets. Therefore, we would obtain similar results with whichever subset we used. Again, BOD was used as a surrogate for DOC because DOC is seldom measured in treatment wetlands and we assumed, with cause, that the DOC is labile (i.e., a BOD load). BOD is also different in that the removal rate coefficient was back calculated from a TIS model with a single tank in the Wetland Constituent Reduction Model

Also presented are the rate coefficients used in the Wetland Constituent Reduction Model (Table 4–9). Note that 1st, 2nd, and 3rd quartiles of OEW reduction rates were used in modeling to provide a magnitude of scale to the variability and uncertainty throughout the rest of the assessment process.

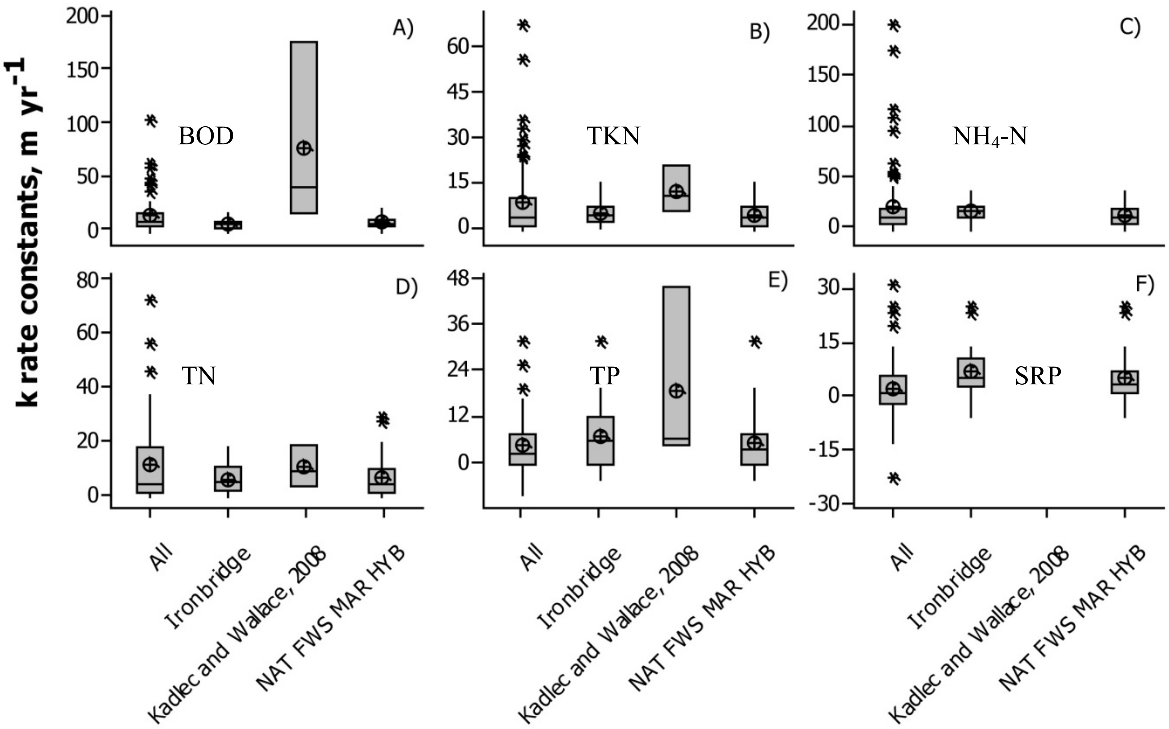

Figure 4–15. Reduction rate constants: ranges from Kadlec and Wallace (2008) natural and created treatment wetlands and EPA's North American Treatment Wetland Database (Knight et al. 1994). All = all native wetlands used for treatment; NAT-FWS-MAR-HYB= all the natural freshwater marsh and hybrid, marsh and pond, treatment wetlands; and Iron Bridge = Orlando Easterly Wetlands. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. (A) BOD = biochemical oxygen demand, (B) TKN = total Kjeldahl nitrogen, (C) NH4-N = ammonium nitrogen, (D) = total nitrogen, (E) TP = total phosphorus, (F) SRP = soluble reactive phosphorus.

Table 4–9. Quartile reduction rates from the Orlando Easterly Wetlands (OEW or Iron Bridge) used in the Wetland Constituent Reduction Model.

Constituent	1st Quartile	2nd Quartile	3rd Quartile
BOD/DOC	0.03	2.72	6.79
TKN	1.21	3.69	7.08
TP	-0.33	5.80	11.89

Note:
BOD/DOC = biochemical oxygen demand equated to dissolved organic carbon
TKN = total Kjeldahl nitrogen
TP = total phosphorus

4.8 MODELS RESULTS

4.8.1 WETLAND CONSTITUENT RELEASE MODEL

The Wetland Constituent Release Model was used first to determine the daily difference in exposure between the base (Base1995NN) and test (Full1995NN) scenarios. The total annual release is the release rate multiplied by the difference in annual exposure in temperature adjusted hectare days (Figure 4–16).

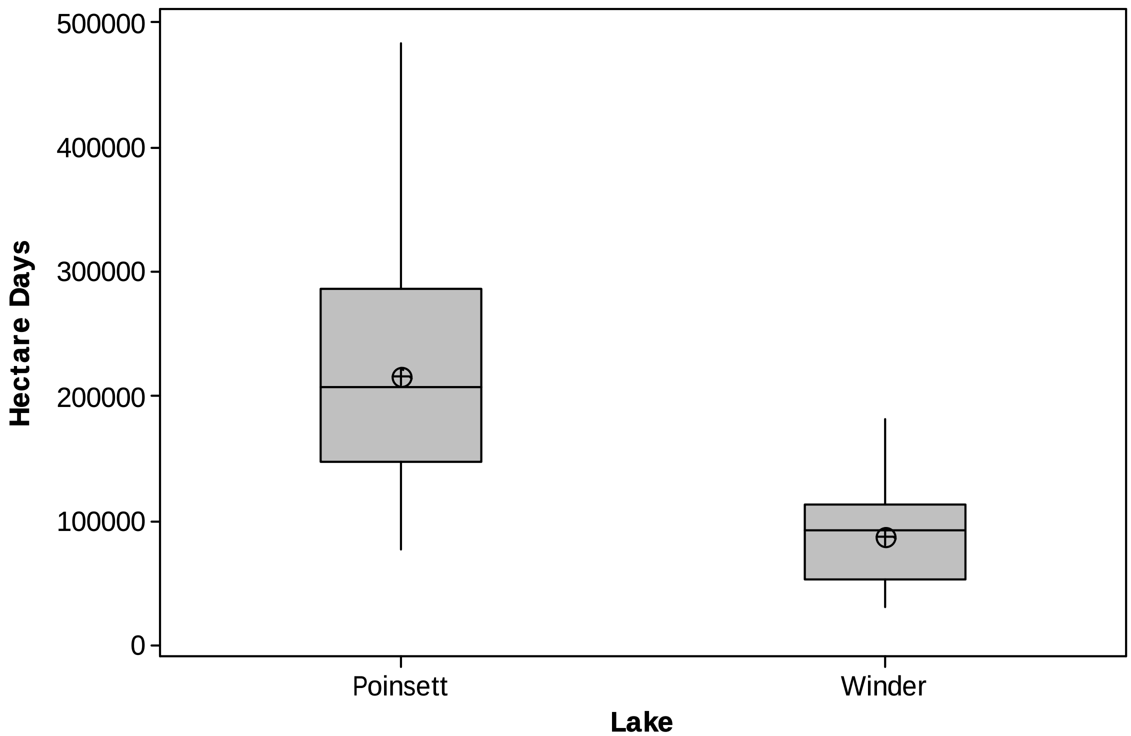

Figure 4–16. Predicted additional annual hectare days of exposure of wetland soils from Lakes Poinsett and Winder between the base (Base1995NN) and test (Full1995NN) scenarios. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box. Data were estimated for additional hectare days for a given year in a 33-year time series for water years 1976 through 2008.

We predicted what additional C, N, and P loads might be released from floodplain wetland soils associated with Lakes Winder and Poinsett under the test scenario (Full1995NN) relative to the base (Base1995NN) scenario (Figure 4–17). The potential for release was greatest for Lake Poinsett wetland soils, because the greatest hydrologic deviations from the base scenario occurred at elevations coincident with those of the floodplain (see Figure 3–1and Figure 4–2). Of the three constituents modeled, C had the highest releases in both areas, and these releases were about six and 20 times greater than the estimated releases of N and P, respectively.

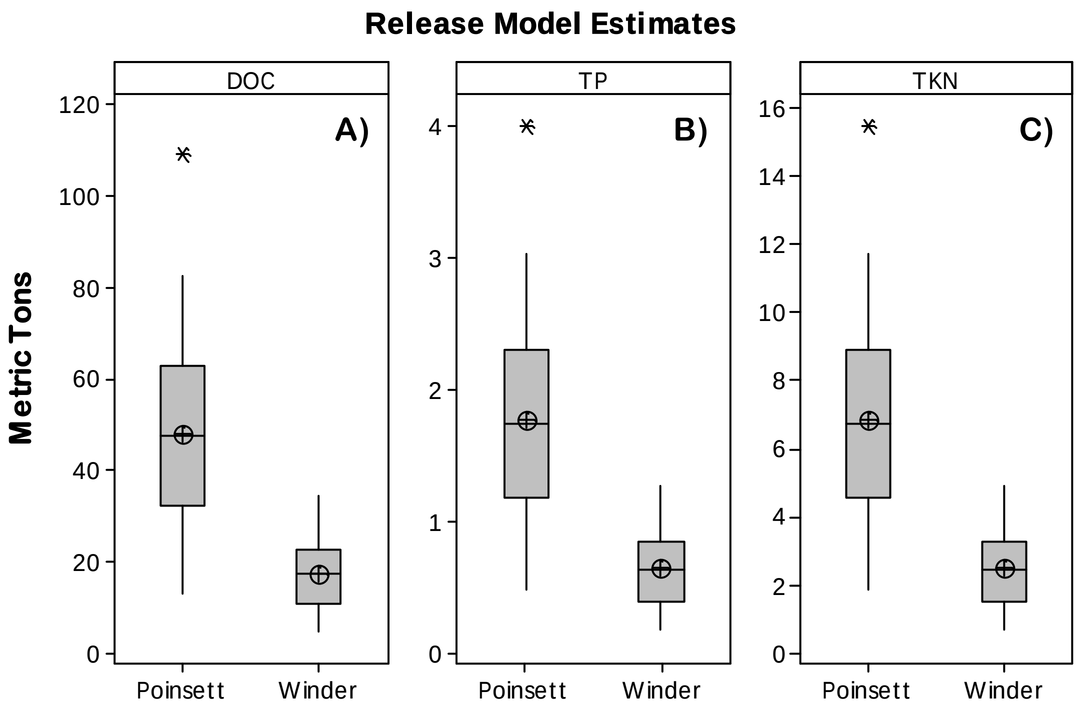

Figure 4–17. Modeled mean annual increased releases (metric tons) from Lake Poinsett and Lake Winder floodplain wetland soils of (A) dissolved organic carbon (DOC), (B) total phosphorus (TP), and (C) total Kjeldahl nitrogen (TKN) for scenario Full1995NN. Center lines are medians, lined circles are means. Lower and upper boundaries of the box represent the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data points that are within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were estimated loads for a 33-year time series of water years 1976 through 2008.

4.8.2 WETLAND CONSTITUENT REDUCTION MODEL

Following a pattern similar to that in the Wetland Constituent Release Model, Lake Poinsett wetland soils had the greatest potential increases in loads reaching the river, about twice that predicted for Lake Winder wetland soils (Figure 4–18). The 1st quartile (25th percentile) reduction rates (Figure 4–19) did little to reduce the additional mass release from the wetlands, so additional loads to the lake were nearly equivalent to the additional releases from wetlands. Indeed, the modeled additional load for TP was higher than the additional release, because the release rate constant at the 1st quartile is negative. In other words, TP was calculated to be added to water as it flowed through the wetlands to the lake.

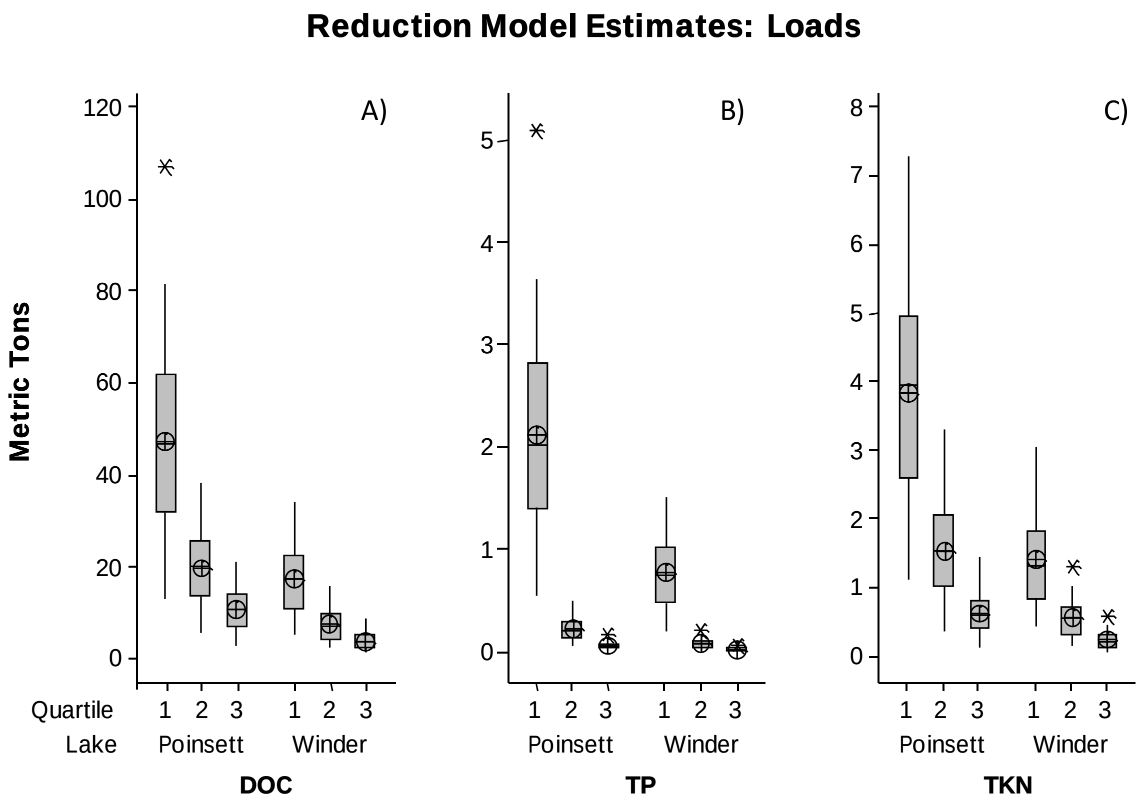

Figure 4–18. Predicted additional annual loads of (A) DOC = dissolved organic carbon, (B) TP = total phosphorus, and (C) TKN = total Kjeldahl nitrogen to Lakes Winder and Poinsett under the Full1995NN scenario compared to the Base1995NN scenario. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were estimated loads for a 33-year time series between water years 1976 and 2008.

We scaled the loadings in two ways. First, we divided the additional load by the additional release to yield the percent of release that reached the river (Figure 4–19). As explained above this scaling of TP exceeds 100% for the 1st quartile because the reduction rate for that quartile is a negative.

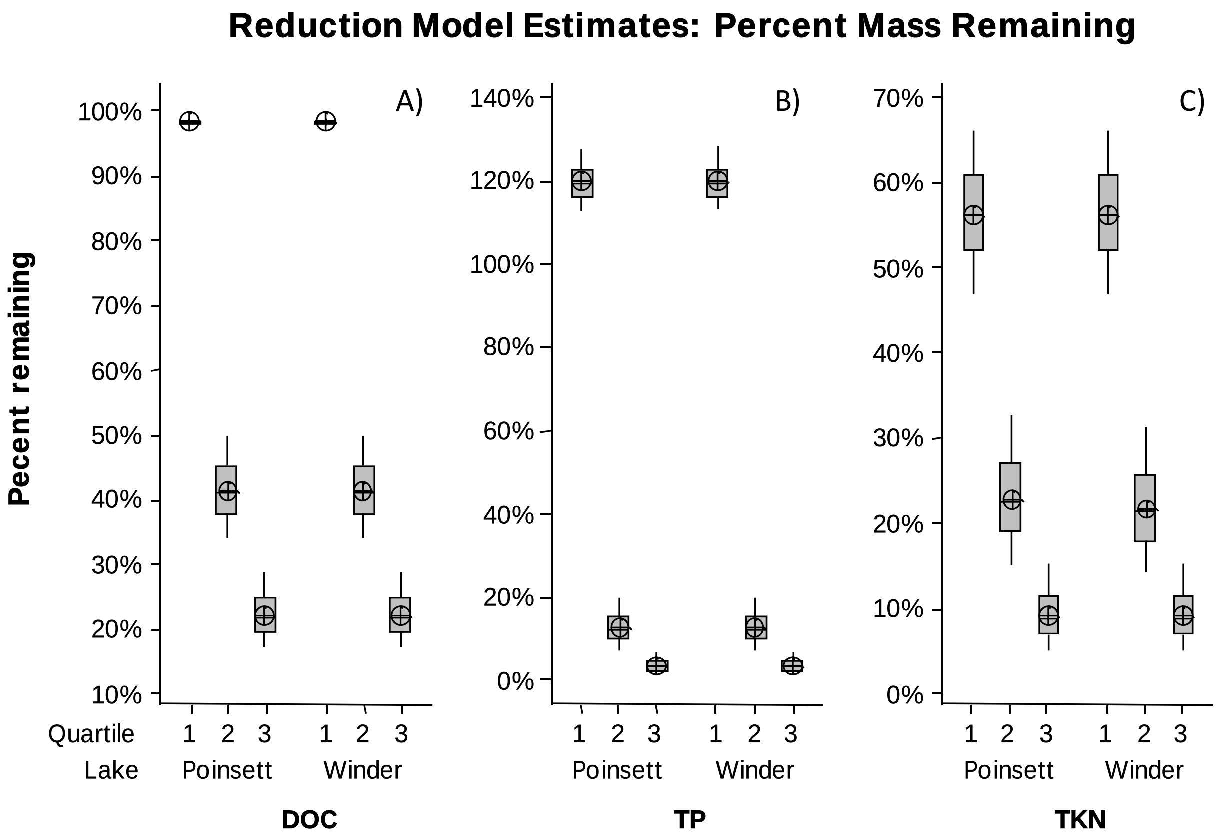

Figure 4–19. Predicted annual percent mass reaching the river from that released from the wetland for (A) dissolved organic carbon (DOC), (B) total phosphorus (TP), and (C) total Kjeldahl nitrogen (TKN) calculated for the Full1995NN scenario. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box. Data were estimated values for a 33-year time series between water years 1976 and 2008.

A second scaling was provided by calculating the increase in discharge-weighted concentration. This scaling entailed dividing C, N, and P additional loads by the sum of monthly discharges for the rising limb of the hydrograph (June through October wet season). As expected, the greatest concentration increases were associated with models using the 1st quartile reduction rate constants; whereas, the smallest concentration increases occurred with models using the 3rd quartile reduction rate constants. Typically, modeled increases in concentration were highest for Lake Poinsett (Figure 4–20).

Lakes Winder and Poinsett are in series along the St. Johns River. Therefore, we combined the estimated loads to provide an estimate of the total increase in metric tons of C, N, and P reaching the river in Lake Poinsett (Figure 4–21) and the associated potential increase in concentration (Figure 4–22). This data was used to predict changes in [DO] and was provided to the Plankton and Submersed Aquatic Vegetation Working Groups.

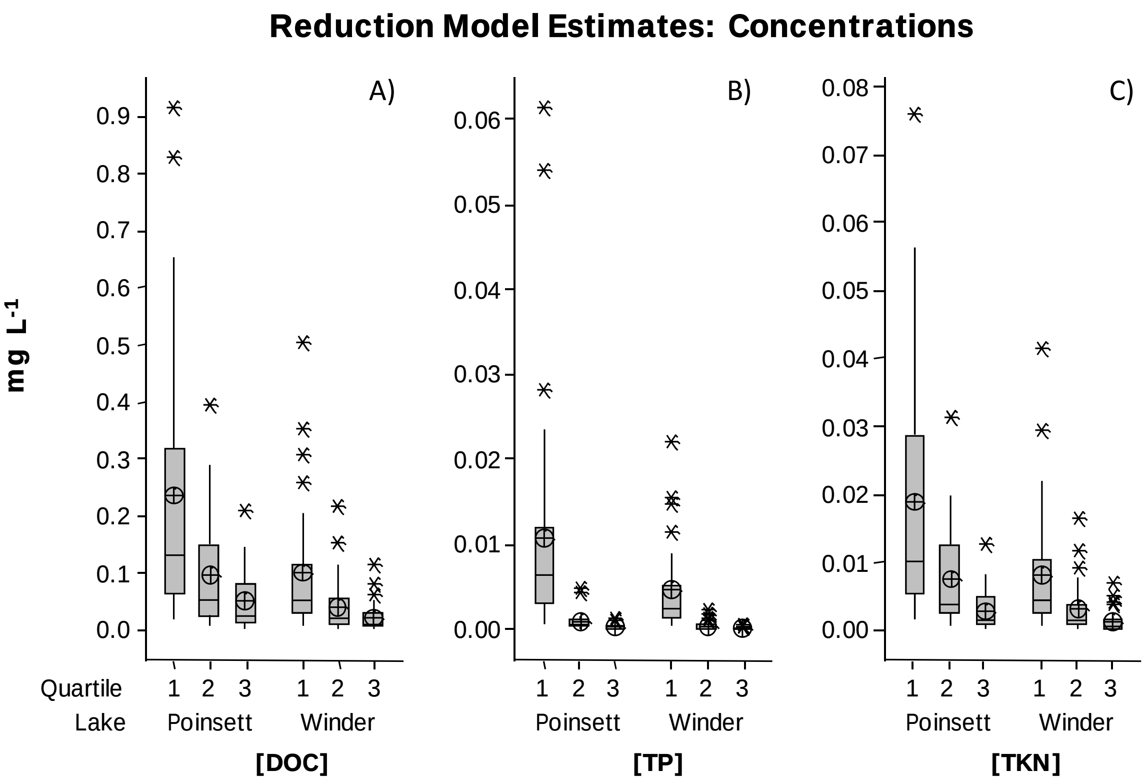

Figure 4–20. Predicted annual average concentrations of (A) dissolved organic carbon (DOC), (B) total phosphorus (TP), and (C) total Kjeldahl nitrogen (TKN) calculated from the additional load from exposed wetland soils that reaches the river under the Full1995NN scenario divided by the average seasonal discharge. Center lines are medians, lined circles are means. The box represents 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were estimated values for a 33-year time series between water years 1976 and 2008.

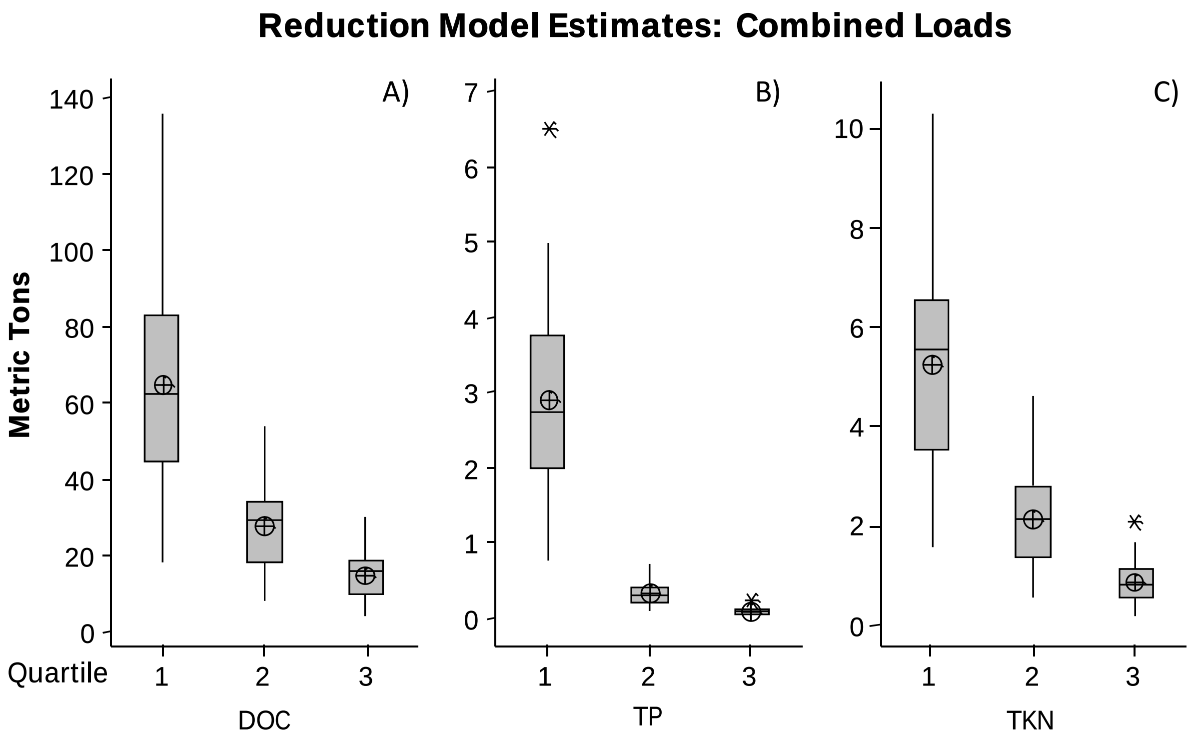

Figure 4–21. Combined predicted annual additional loads for (A) DOC = dissolved organic carbon, (B) TP = total phosphorus, and (C) TKN = total Kjeldahl nitrogen reaching the river from wetlands associated with both Lake Winder and Lake Poinsett under the Base1995NN scenario versus the Full1995NN scenario. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were estimated loads for a 33-year time series between water years 1976 through 2008.

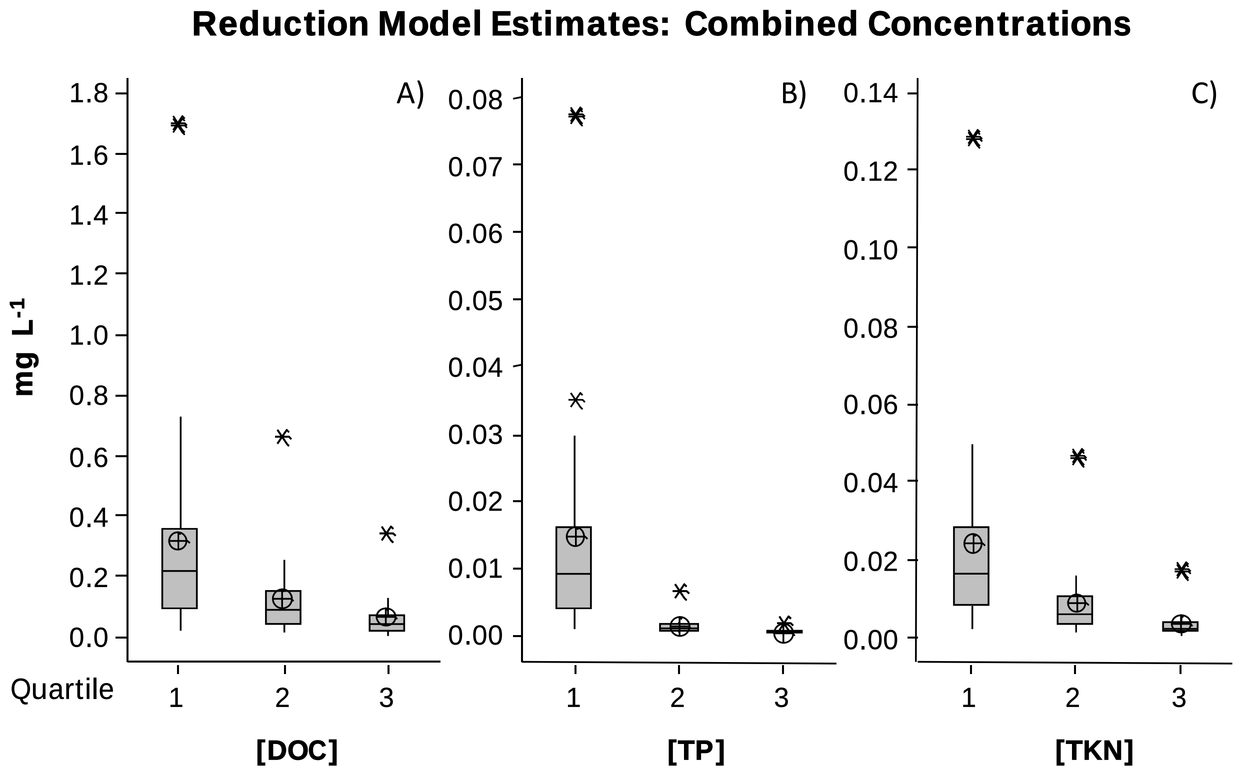

Figure 4–22. Predicted annual average concentrations of (A) DOC = dissolved organic carbon, (B) TP = total phosphorus, and (C) TKN = total Kjeldahl nitrogen calculated from the additional load from exposed wetland soils that reaches the river under the Full1995NN scenario divided by the summed seasonal discharge. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were estimated values for a 33-year time series between water years 1976 and 2008.

4.9 Relationships of Lake Water Quality Parameters and Dissolved Oxygen Results

We investigated the long-term water quality datasets for Lake Winder and Lake Poinsett between 1979 and 2010. Summary results for Lake Poinsett are in Table 4–10.

Table 4–10. Summary of water quality parameters in Lake Poinsett. The period of record is from 1979 through 2010.

Label	N	Mean	Std Dev	Median	Min	Max	Parameter (Unit)
[DO]	259	6.3	2.2	6.4	0.8	16.6	Dissolved oxygen (mg L^{-1})
[Chl-a]	243	10.8	13.5	6.8	0.0	136.2	Chlorophyll-a (µg L^{-1})
[TP]	254	0.10	0.07	0.09	0.01	0.45	Total phosphorus (mg L^{-1})
[PO_4]	187	0.05	0.04	0.04	0.00	0.33	Dissolved orthophosphorus (mg L^{-1})
[TN]	240	1.90	0.52	1.80	0.00	4.37	Total nitrogen (mg L^{-1})
[NO_x]	253	0.07	0.09	0.04	-0.01	0.55	Dissolved nitrate and nitrite (mg L^{-1})
[NH_4-N]	239	0.09	0.07	0.07	0.01	0.51	Dissolved ammonium nitrogen (mg L^{-1})
[TOC]	210	27	5	26	18	44	Total organic carbon (mg L^{-1})
[DOC]	67	28	6	27	18	44	Dissolved organic carbon (mg L^{-1})
Color	251	193	114	150	38	700	Color (PCU)
pH Field	253	7.4	0.5	7.4	5.9	9.9	pH units
Secchi	216	0.7	0.3	0.7	0.2	1.9	Secchi depth (m)
Water_Temp	261	24	5	25	8	34	Water temperature (^oC)
Wind_Speed	110	7.4	5.5	6.1	0.0	25.0	Wind velocity (m S^{-1})

Note:
n = number of observations
Std Dev = standard deviation
PCU = platinum cobalt units

Within a given hydrologic year, concentration increases for most water quality parameters but decreases for [DO] during the summer and fall periods. These are concomitant with increasing lake stage and discharge (Figure 4–23).

To get a seasonal insight into these various patterns, we grouped months into a dry season (November through May) and a wet season (June through October). Concentrations of water quality parameters tended to have stronger and more significant relationships with [DO] during the wet season than in the dry season.

During the wet season, [DO] increased with pH ($r = 0.73$; $p <0.001$) but declined with increases in color, [NH_4-N], discharge, [TP], [SRP], stage, and [TOC]. [DO] also declined with increases in [TKN], but not significantly.

In contrast, during the dry season, [DO] decreased with increasing wind velocity ($r = -0.58$; $p < 0.001$) and increased with rising water temperature ($r = 0.28$; $p = 0.024$). Much weaker, but still significant relationships existed between [DO] and increases in [NO_x], discharge, [TP], [SRP], and [NH_4-N] during the dry season.

Phytoplankton productivity, as indicated by [Chl-a], did not influence lake [DO] during the dry or the wet season.

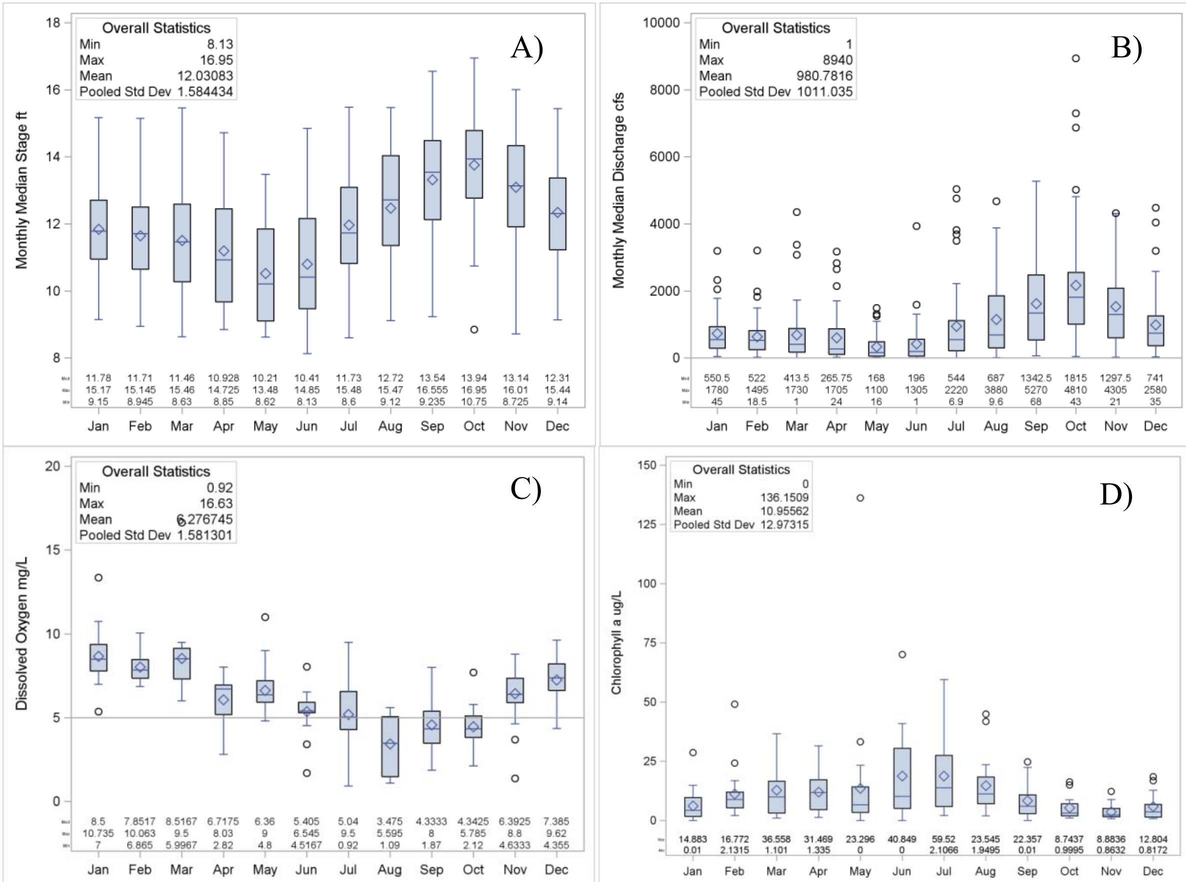

Figure 4–23. Monthly box plots representing statistical values for Lake Poinsett, (A) stage (ft), (B) discharge (cfs), (C) dissolved oxygen, (D) and chlorophyll-a between 1970 through 2009. The boundaries of the box indicate the 1st and 3rd quartiles (25th to 75th percentiles), a line within the box marks the median, and the diamond center is the mean. Whiskers above and below the box indicate the 90th and 10th percentiles, while circles are identified as data outliers.

We hypothesized that, during the June through October wet season, a major factor regulating lake [DO] is hydrology driven. The influx of floodplain water, which has higher [C], [N], and [P] and lower [DO] than lake water, is an important contributor to decreases in lake [DO] and associated biological impacts such as fish kills. During the summer and fall periods, rainfall at Lake Hell ‘n’ Blazes was contemporaneous with declines in [DO] (see Figure 2–4). Recent evidence from Lake Winder suggests that when there is an influx of floodplain water, biologically extreme hypoxia tends to be widespread throughout the upper basin lakes. For example, fish kills tend to occur in the upper basin lakes shortly after summer and fall rainfall events (see Figure 2–5). However, this widespread hypoxia only occurred when lake water elevations were increasing and floodplain soils were inundated (Keenan et al. 2008).

4.10 Projected Changes in Dissolved Oxygen Concentration Results

The best model ($p<0.0001$; adjusted $r^2 = 0.415$) for predicting changes in [DO] in lake water from available information was a multiple regression:

$$\Delta[DO] = (-0.1014 \text{ mg L}^{-1} \text{ m}^{-1} \times \Delta \text{water elevation}) + (-4.61097 \times \Delta[TP]) + (-0.07393 \times \Delta[TOC]) \quad \textbf{[Eq. 3–1]}$$

where water elevation is in meters above sea level NGVD29 and all concentrations are in mg L^{-1}. All parameters were significant at the $p < 0.05$ level. The change in DO is the difference between the predicted DO of the base scenario (Base1995NN) and the test scenario (e.g. Full1995NN).

The empirical data were derived from analysis of monthly water samples. Therefore, we used estimates of monthly average water elevations and concentrations of TP and TOC to as inputs into Equation 3.–1 to model potential effects on [DO]. We estimated monthly average concentrations by determining, for each year, the proportion of total wet season (June through October) discharge that occurs in each month. The annual additional load was multiplied by these monthly proportions to yield monthly additional loads. We then divided these monthly loads by the discharge for that month to produce an average monthly increase in concentration. Loads resulting from all three quartiles of reduction rates were used. The change in water elevations was derived from output from the HSPF hydrologic model for each scenario.

We used these estimated monthly average concentrations and change in water level to model monthly predicted changes in [DO] for each month of the wet season over the 33 years of simulation for a scenario using each quartile for reduction rate coefficients (Figure 4–24). When using the median reduction rate coefficient for OEW, is a reduction in [DO] of 0.005 mg L^{-1}. The range of results for modeled depletion of [DO]—using high (3rd quartile), median, and low (1st quartile) OEW reduction rate coefficients—is not large. Even the largest water level reduction resulted in a small decrease in [DO] for the Full1995NN, the test scenario, which would produce the largest [DO] change of all the scenarios.

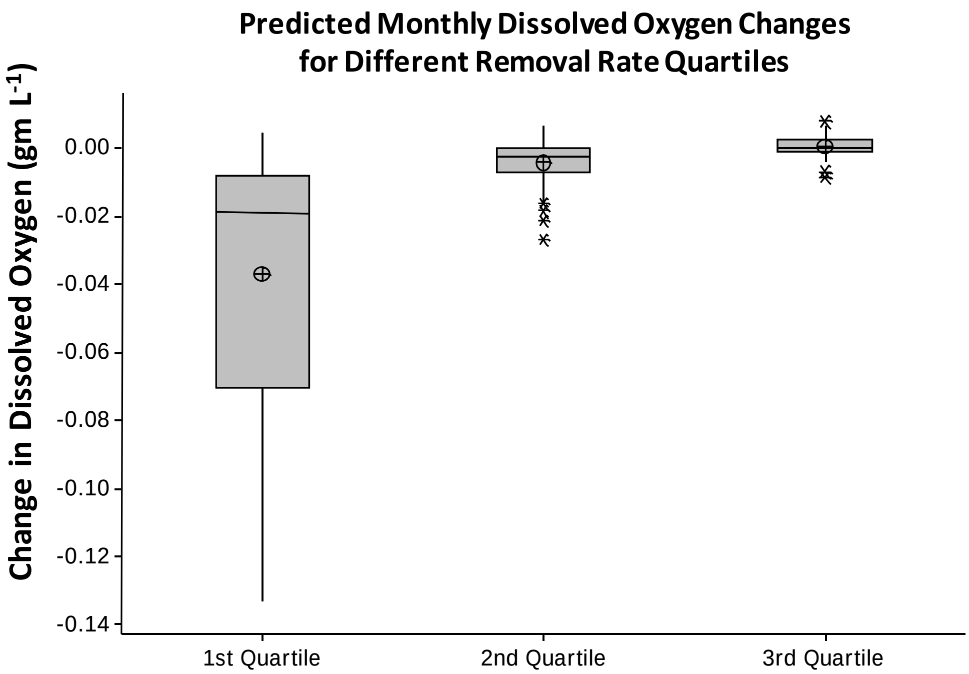

Figure 4–24. Modeled depletion of monthly dissolved oxygen concentration [DO] based on modeled monthly additional total organic carbon and total phosphorus loads for the Full1995NN scenario. Center lines are medians, lined circles are means. The box represents the 1st to 3rd quartiles (25th to 75th percentiles). The whiskers extend to the highest and lowest data point that is within 1.5 times the interquartile width past the ends of the box; outliers are represented by asterisks. Data were predicted monthly changes in [DO] for a 33-year time series for water years 1976 through 2008. Negative values indicate a decrease in [DO]

5 DISCUSSION

Among scenarios, the Full1995NN test scenario showed the largest hydrologic deviations from the base scenario (Base1995NN), and, among areas surveyed, Lakes Poinsett and Winder showed the largest hydrologic deviations. The synoptic soil survey indicated that soils at the main study site on Lake Poinsett fell into the mid-range of values relative to those sampled at other sites around Lake Poinsett and at Lake Winder and were reasonably similar to soils downstream.

All of the release rate studies indicated that the wetland organic soils around Lake Poinsett are exceptionally resistant to oxidation. This finding is congruent with expectations derived from the soil analyses. The Lake Poinsett wetland soils had low C:N ratios (approximate range of 10 to 13). Wetland soils with these low C:N ratios would be expected to have oxidation rates in the order of decades to centuries (Brady and Weil, 2008). Thus, the weight of evidence from soil analysis, and from the various studies to quantify release rates, indicates that soils of the floodplain wetlands of Lake Poinsett are mostly non-labile and have refractory or recalcitrant, types of OM. However, soils with high percentages of organics and higher C:N ratios could have strikingly different results as discussed in Section 5.3.

The Wetland Constituent Release Models indicated that even with low oxidation rates there is the potential for a large mass release of DOC and nutrients from the wetlands (see Figure 4–17). This is principally due to the large wetland area affected. However, only a proportion of that release would actually reach the lakes as indicated by the Wetland Constituent Reduction Model (see Figure 4–19).

Because these lakes are in a series on the river, we evaluated the effect of the total additional DOC and nutrient loads (Lake Winder wetland loads plus Lake Poinsett wetland loads) on Lake Poinsett for the test scenario with the strongest potential effects (Full1995NN) and found that the load increases were small (see Figure 4–21). Because DOC and nutrient loading are important drivers for [DO], the potential effects on DO would also be small. Indeed, the multiple regression model yielded an estimated median monthly change in [DO] of only -0.005 mg L^{-1} (see Figure 4–24). The modeled effects on [DO] were quite small (approximately -0.04 mg L^{-1}), even when we used a low reduction rate coefficient (OEW 1st quartile).

All evidence indicates that the refractory floodplain soils, even in areas that should show the strongest response to water withdrawals, yield weak responses to water withdrawals in terms of additional loadimgs of C and nutrients and their subsequent effects on [DO]. This conclusion holds for the most extreme (unrealistic) test scenario (Full1995NN) in the location with the strongest hydrologic response. Logically, we can conclude that the potential effects will be even smaller for all lesser ranked scenarios and for all less responsive areas. This is especially true for all scenarios using 2030 land use, given the forecast of higher discharges than in the base scenario (Base1995NN). This conclusion follows from our ranking of scenarios (see Figure 4–4). Consequently, we conclude that potential effects stemming from any increase in organic soil exposure will be small for all segments and all modeled scenarios.

5.1 Discussion of General Effects

We assessed the significance of the potential environmental effects of modeled water withdrawals by considering three factors: the strength of the effect, its persistence, and the diversity of species and functions affected (see Chapter 2. Comprehensive Integrated Assessment).

Strength

As defined for this study, the strength of an effect considers both its intensity and the area affected. Thus, an effect of high intensity that is widespread has very high strength, whereas an effect of low intensity over a small area would be very weak. In between these extremes are effects that are more moderate. Both components of strength can be evaluated based on the degree of deviation from the base condition. The deviation in the level of an attribute, such as [DO], can serve as a measure of intensity and the fraction of the area affected can serve as a measure of extensiveness.

Due to mixing, effects of [DO] would be very widespread in the affected lakes, so the significance of [DO] effects will vary only according to their intensity. The intensity of a particular [DO] reduction depends on the current [DO], or percent saturation, which is variable from month to month and year to year. Consequently, in our assessment of intensity of [DO] effects we developed a scale based on the knowledge that during the July through October wet season, when the [DO] reductions will be expressed, [DO] is usually below 5 mg L^{-1} and often below 3 mg L^{-1} (see Figure 2–4). With advice from the Fish and Benthic Macroinvertebrates Working Groups, we employed the following ordinal scale of potential changes in [DO] at low ambient concentrations (<5 mg L^{-1}), such as occurs in the low [DO] period of late spring and early fall:

- Low (1) = Δ [DO] < 0.2 mg L^{-1}
- Medium (2) = 0.2 $\leq \Delta$ [DO] < 1 mg L^{-1}
- High (3) = Δ [DO] $\geq$ 1 mg L^{-1}

Even at initial concentrations < 5 mg L^{-1}, a decrease of 0.2 mg L^{-1} or less is unlikely to have more than a marginal effect on growth and reproduction of benthic macroinvertebrates and fish (Benthic Macroinvertebrates and Fish Working Group leaders, Robert Mattson and Steven J. Miller, SJRWMD, pers. comm., 2010). However, decreases of 1 mg L^{-1} or more would be expected to increase stress and affect growth and reproduction, perhaps even causing mortality.

Persistence

Persistence relates the recovery time (RT) of an effect, or perturbation, to its return interval (RI). This could be reduced productivity or diversity due to low oxygen stress and how long it would take the system to recover to its normal state after the reduction. We also developed a three-level, ordinal scale for persistence. We judged that a perturbation that recurs before the affected components have recovered to their previous state (i.e., High (3): RT/RI ≥1) has a high level of persistence. At this level of persistence, some effect from the perturbation is always present and there is the potential for accumulation of effects over time. We defined a moderate level of persistence (Medium (2)) to be cases where 0.1 < RT/RI < 1.0. At this level of persistence, recovery will occur before the next perturbation but there can be extended periods when residual effects are present. When RT/RI < 0.1, the effect is ephemeral and at most times will there will be no appreciable difference from the antecedent condition (Low (1)). The ordinal scale is as follows:

- Low (1) = RT/RI < 0.1
- Medium (2) = 0.1 < RT/RI < 1.0
- High (3) = RT/RI ≥ 1.0

Diversity

Briefly, diversity encompasses both the breadth of species and the suite of ecosystem functions and characteristics affected. Here, the number or fraction of species affected or the number of functions affected can be used as measures of the diversity of the effect. Because a general reduction in DO affects the entire ecosystem, DO effects will always be broadly diverse. However, if the effect was so weak as to be immaterial for most species, it merits a low ranking for the diversity. The ordinal scale is described below

- Low (1) = effects minimal to most faunal species and lake and river functions and on the order of background noise
- Medium (2) = several species or functions affected to a measurable degree but fewer than half also no mortality or long term (> 1 yr) density changes
- High (3) = many species or functions affected and mortality or density changes in some species or fundamental changes in direction of degree of river and lake functions

Strength, Persistence, and Diversity

We used the three ordinal scales to develop a combined overall scale according to the following scheme (Figure 5–1).

Diversity

Strength, Persistence	High (3)	Medium (2)	Low (1)
3,3	3,3,3	3,3,2	3,3,1
3,2	3,2,3	3,2,2	3,2,1
2,3	2,3,3	2,3,2	2,3,1
2,2	2,2,3	2,2,2	2,2,1
3,1	3,1,3	3,1,2	3,1,1
2,1	2,1,3	2,1,2	2,1,1
1,3	1,3,3	1,3,2	1,3,1
1,2	1,2,3	1,2,2	1,2,1
1,1	1,1,3	1,1,2	1,1,1

Extreme
Major
Moderate
Minor
Negligible

Figure 5–1. Matrix for determining degree of effect based on the three factors. Strength and persistence are in the leftmost column and diversity is across the top. Once strength, persistence, and diversity have been assessed as high (3), medium (2), or low (1), reading the matrix across and down will indicate the appropriate level of overall effects to assign to that environmental variable in that segment for that scenario.

5.2 Effects by River Segment Using Scenario Full1995NN

River Segment 8

River segment 8 had the largest modeled deviations from the base condition (Base1995NN). Consequently, it should be the segment with the strongest response to modeled water withdrawals. In our assessment of the levels of effects to segment 8, we received and considered evaluations of the Plankton and Submersed Aquatic Vegetation Working Groups with respect to the potential effects of modeled deviations in DOC and nutrient loadings. Both working groups indicated that our modeled potential releases of DOC and nutrients to the river would have negligible effects on the attributes they considered.

We also modeled the potential [DO] reduction associated with [DOC] and [TP] releases at different water elevations during the wet season using a multiple regression analysis. For river segment 8, an array of potential changes to [DO] were predicted from multivariate regression using different quartiles of DOC and TP loadings, which resulted in a range of potential reductions of 0.000 mg L^{-1} to -0.049 mg L^{-1}. We used median values in the final assessments. The results from the 1st and 3rd quartiles (25th and 75th percentiles) served as indicators of the uncertainty resulting from both release and uptake rates and models. The medians (50th

percentiles) for all variables produced a projected annual median change in [DO] of -0.005 mg L^{-1}, which would be low (1) in the strength category.

While occurring throughout the lakes in river segment 8, these changes in [DO] are not expected to cause a significant enough perturbation that full recovery would not readily occur within the year. Given that the effects on [DO] would only occur at the onset of the wet season (i.e. June through October), when water levels rise onto the floodplain, the return interval would be approximately one year. An average annual decrease of 0.005 mg L^{-1} or less is unlikely to have a sufficiently significant effect to take more than a minimal time for recovery. Moreover, it is likely that any effects present would be immeasurable against the natural background variability to which the system should be well adapted for rapid recovery. Therefore, we predicted a Persistence value of 1 for the most extreme scenario (Full1994NN).

Finally, we gave a low (1) ranking for the diversity of the [DO] effect, because the effect was judged too small to materially affect species or functions. The combination of strength, persistence, and diversity rankings results in an expected effect of negligible (1, 1, 1) (see Figure 5–1).

Judging from their effects on [DO], and considering the judgments of the Plankton and Submersed Aquatic Vegetation Working Groups on their importance to the plankton and submersed aquatic vegetation, none of the effects we evaluated rose above a negligible level for segment 8 (Figure 5–1 and Figure 5–2). Because the Full1995NN scenario had stronger effects on the hydrologic driver than did any other scenario, we can conclude that effects for all other scenarios also would be negligible.

Scenario – Full1995NN

River Segment	Δ DOC loading	Δ TP loading	Δ TKN loading	Δ Dissolved Oxygen	Overall effect
1	*	*	*	*	*
2	*	*	*	*	*
3	*	*	*	*	*
4	*	*	*	*	*
5	**	**	**	**	**
6	**	**	**	**	**
7	****	****	****	****	****
8	***	***	***	***	***

Level of Effect

- Negligible
- Minor
- Moderate
- Major
- Extreme

Cross-hatching indicates abbreviated analysis

Uncertainty

*	Very Low
**	Low
***	Medium
****	High
*****	Very high

Figure 5–2. Summary and characterization of potential effects for test scenario Full1995NN. Uncertainty levels are discussed in Section 4.4. DOC = dissolved organic carbon, TP = total phosphorus, and TKN = total Kjeldahl nitrogen.

Other Segments

Given the potential effects for segment 8 were negligible, we concluded that the results of a full analysis for segments 7 through 5 would also be negligible, because the associated deviations in water levels were even smaller. We tested this conclusion using an abbreviated analysis to assess the potential for effects in segments 7 through 5 in which we considered three factors: (1) the deviation in water elevation, (2) the ratio of wetland area to river discharge, and (3) the carbon to nitrogen ratio of organic soils.

As stated above, segment 8 had the largest deviations from the base scenario (Base1995NN) in water elevation, which was the hydrologic driver for all the effects we evaluated. However, we felt it was important to bolster this assumption by considering other relevant factors. Perhaps, foremost among these was the possibility that the ratio of wetland area to discharge in segments 7 through 5 might be higher than in segment 8. If this were so, then a smaller deviation in water level could still yield a stronger effect.

In terms of the modeled deviation in water level, segment 8 in the area of Lakes Winder and Poinsett was the most strongly affected of all segments on the river (see Figure 4–4). River segment 7 had the next greatest modeled deviations, followed by segments 6 and 5.

In our estimates of the wetland area to discharge, we subtracted the area of forested wetlands from the total wetland area because the Wetland Vegetation Working Group determined that the forested wetlands were nearly always associated with tributaries or seeps and would not be directly affected by variations in river level. Although the Soil Conservation Service soil maps (Soil Survey Staff 2010) defined most of river segment 7 as having mineral soils, MFLs transect data reported at least a few centimeters of organic epipedon on the surface of nearly all soils sampled in segments 7 and 8 (Mace 2006, 2007a, 2007c). Consequently, we treated all of the soils of herbaceous wetlands in segment 7 as organic soils.

Areas of organic soils in segments 8 through 5 are shown in Table 4–1, along with average annual river discharge for each segment. We divided wetland area by discharge to get a wetland area-weighted discharge for segments 8 through 5 and standardized these ratios to segment 8. We compared only segments 8 through 5. Segments 8 and 7 had the greatest wetland areas. Segments 6 and 5 had the greatest discharges. Segment 8 had the largest wetland area discharge-weighted ratio, and that ratio decreased with each downstream river segment. Because segment 8 has the highest wetland area to discharge ratio, it would show the strongest effects from deviations in water levels, with the other segments showing lesser and decreasing effects.

In addition to further decreasing wetland area-weighted discharges, river segments 1 through 4 had very small modeled deviations in water elevation, with water elevations affected more by sea level than water withdrawals. In these segments, hydrodynamic modeling indicated that decreases in stage would be less than a centimeter; therefore, these segments were easily determined to have negligible risk for effects caused by deviations in water elevations without further analytical effort.

As a final test, we used the biogeochemical data that was collected during the synoptic soil sampling to compare the nutrient ratios for soil samples collected in segment 7 to those of soils in segment 8. The C:N ratio of segment 7, an indicator of soil lability, was similar to the ratio for segment 8. Therefore, we were comfortable that N and P release rates should also be similar. Because the same condition of low soil C:N ratio (approximately 13) was also true for segment 6, we were comfortable that N and P release rates for this segment also would be similar to those for segment 8. Segment 5 had a higher C:N ratio (16), which indicates that oxidation and release per square meter could be higher than in segment 8. However, its organic wetland area to discharge ratio is one eighth that of Lake Poinsett. In addition, for the Full1995NN test scenario the average decrease in water elevation in segment 5 is 1.5 cm compared to the 5 cm average decrease in segment 8. The combination of a lower wetland area to discharge ratio and a lesser water elevation change should more than offset the higher C:N ratio in segment 5 and result in lesser effects than predicted for segment 8. Based on these tests, we conclude that the effects of withdrawal will be less in segment 5 than in segment 8, resulting in a negligible ranking.

5.3 EXAMPLE OF POTENTIAL EFFECTS USING UPSTREAM SOILS

The potential effects of withdrawals on wetland organic soils are to a great extent governed by the lability of OM. To illustrate this point, we substituted release values for medium to highly labile wetland organic soils collected from Blue Cypress MCA (the location is in segment 9 to the south, an area that will not be affected by the modeled withdrawals) in place of the measured release values of Lakes Winder and Poinsett and reran the full analysis. As shown in .Table 5–1, C:N ratios and organic content (loss on ignition) is high in the southern reaches of the river (i.e., the four marsh conservation areas) upstream of Lake Poinsett (Lake Poinsett wetlands). These are strong indicators of a higher potential for oxidative release of C and nutrients.

Table 5–1. A comparison between Lake Poinsett wetlands and the upper basin marsh conservation areas (MCAs).

Marsh Conservation Area (MCA)	n	Histosol Suborder	Bulk Density (g cm^{-3})	Loss on Ignition (%)	C:N Ratio (mass basis)	Soil Organic Matter (SOM) Activity*
Fort Drum MCA	12	Fibrists	0.06	95	17	Active
St. Johns MCA	36	Hemists	0.13	91	14	Slow
Blue Cypress MCA	6	Fibrists	0.08	95	17	Active
Three Forks MCA	6	Hemists	0.08	90	14	Slow
Lake Poinsett Wetlands	$6^{\dagger}$	Saprists	0.20	58	10	Passive

Note:
*Soil organic matter activity is a measure of how quickly the organic matter fraction of soil will decompose, and is categorized as active, slow, or passive based on the C:N ratio of the soil. Sample size for Lake Poinsett for Loss on Ignition only is 77.

n	=	Number of observations
C:N	=	Carbon to nitrogen ratio
Active SOM	=	C:N of 15 to 30, decomposition in 1 to 2 yrs
Slow SOM	=	C:N of 10 to 20, decomposition in 15 to 100 yrs
Passive SOM	=	C:N of 7 to 10, decomposition in 500 to 5,000 yrs

For the Blue Cypress MCA, we can calculate C and nutrient release rates from previously completed soil subsidence studies (Reddy et al. 2006). Details of these calculations and the release and removal model results can be found in Appendix 7E. If we apply these values to the Wetland Constituent Release Model, followed by the Wetland Constituent Reduction Model, much larger loads reach the lakes than in the modeling for segment 8. Using the multiple regression equation, the added increase in concentrations corresponded to a change of -2.47 mg L^{-1}in lake [DO]. This exercise also predicted an increase of 0.076 mg L^{-1} in [TP] for the combined loads, which is almost 85% of the existing current [TP] goal for Lake Poinsett of 0.090 mg L^{-1}. This analysis indicates that were the withdrawal in more labile soils, it would result in ecological effects falling into the 'extreme' category in our analysis. However, the upstream areas will not be affected by the proposed withdrawal scenarios. This exemplifies the danger of extrapolating the results of our analyses on Lakes Winder and Poinsett to other areas. Site specific hydrology and soil characteristics must be carefully considered when addressing these types of analyses.

5.4 Uncertainty

5.4.1 General Uncertainty

Our evaluation of the level of uncertainty considered three components of the strength of scientific evidence: (1) the weight of supporting evidence (SE), (2) the degree of understanding of the causal mechanisms (UM) and (3) the strength of the predictive models (PM) (Chapter 2. Comprehensive Integrated Assessment). Weakness in of any of these three facets would increase the uncertainty of our analyses.

The synoptic soil survey provided good supporting evidence for the results of our release rate studies in segment 8. Although many other wetland organic soils have much higher potentials for oxidative release of C and nutrients, the low C:N ratios of soils in segment 8 supports the findings of the release rate studies that the soils are recalcitrant and subject only to slow oxidation rates (Aikenhead and McDowel 2000, Brady and Weil 2008). However, given the course spatial coverage of this data, we could only rate the strength of the SE as Medium.

There was agreement within the Biogeochemistry Working Group that the environmental mechanisms underlying the examined effects are well understood. While there were no concerns about our understanding of the oxidative processes and drivers, the degree of oxidation observed during our studies was expected to be larger. We rate UM as high.

We could not quantitatively evaluate the goodness of fit of our quantitative models. However, by using the highest release rates measured and by using upper and lower statistics for the reduction rate coefficients, the modeling was not likely to underestimate the potential effects. Even though the consistent finding of extremely low release rates in all the field and laboratory studies indicates that release rates were unlikely to have been significantly underestimated, the quantitative strength of our predictive models could not be rated high. As previously mentioned, the spatial coverage of our release rate studies was relatively small. Further, a highly predictive model would require a calibrated two-dimensional hydrologic model to provide better estimates of wetland hydrology and discharges, which are both important to assessment of potential release rates. Because we necessarily used a flat-surface water assumption for the wetland hydrology and had weak quantitative data in the Wetland Constituent Release Model, the certainty of the PM was negatively affected in a biased manner toward over-prediction of oxidation. Finally, the uptake coefficients we used for the Wetland Constituent Reduction Model had to be extracted from literature rather than from onsite measurements. Even though the literature estimate is from a nearby site, it is a municipal treatment wetland with high nutrient loading rates rather than a natural wetland with much lower loadings. In consideration of these limitations of the predictive modeling in segment eight, we would not have the strength in the PM component of uncertainty be more than medium except that all the effects were deemed to be negligible.

We used conservative assumptions and conservative values in ranges at nearly every juncture in our analyses; the flat water elevation assumption from the river through the wetlands being the most conservative. In the special case of determination of a negligible levels of effects, the strong bias caused by using a flat water elevation surface assumption justifies a PM of high certainty. In other words, exposure was over-estimated to such a degree that there was little chance of underestimating the level of effects. Therefore, a determination of negligible level of effects is unlikely to be in error.

5.4.2 Uncertainty Level Determination by River Segment for Full1995NN Test Scenario

Our uncertainty results incorporated the uncertainty as provided by the Hydrology and Hydrodynamics modeling Working Group (see Chapter 3. Watershed Hydrology and Chapter 5. River Hydrodynamic Calibration). We restricted our descriptions to our own assessments.

Segments 1 through 4

Many lines of evidence show that water levels in these segments are largely controlled by ocean levels. Consequently, the hydrodynamic model and empirical data indicate that effects on water levels in segments 1 through 4 from surface water withdrawals in all scenarios would be very small. Moreover, existing constituent loads in these areas are already high compared to the potential releases from floodplain wetland soils. The current information on water level change, wetland area, and river discharge combine to make any material effect on [DOC] or nutrients in the river due to biogeochemical changes in the wetland soils highly unlikely. Because we can have high confidence in the hydrologic predictions for these segments and in consideration of the pre-existing high loadings as compared to potential floodplain releases, these segments require no further analysis to determine that our uncertainty for these segments is very low, as depicted in Figure 5–2.

Segments 5 through 6

These segments had very small, but not negligible, modeled deviations in water levels. They also had low wetland to river load ratios such that environmental effects are highly unlikely (Table 4–1). However, because we lack the release data to assess the lability of the soils and because the modeled deviations in water levels were not negligible, we judge that there is a slightly higher level of uncertainty (i.e., low) associated with a conclusion of negligible effects in these segments.

Segments 7 and 8

Segment 7 soils are largely mineral, so most of the soil column would be insensitive to oxidation. Moreover, although much of the area has an organic epipedon that could be oxidized, its chemistry indicates that it has already undergone extensive decay and would be quite refractory to further oxidation. We did not measure release rates in this area, however, so there is no site-specific predictive model. The lack of a predictive model, coupled with the large wetland area to discharge ratio, indicates that the uncertainty associated with a conclusion of negligible effects is higher here than in the downstream segments. We determined that segment 7 uncertainty is high because it is similar to segment 8 in other factors (e.g., hydrologic deviation; soil C:N ratio, organic soil area to discharge ratio) but lacks any site specific release rate data. If segment 7 release rates were higher than expected, effects could be stronger than in segment 8 (Figure 5–2).

Segment 8 had the largest wetland area to discharge ratio, and its soils were classified as predominately organic. Additionally, segment 8 was predicted to have the largest decrease in water elevations with withdrawals, so it warranted the most scrutiny. We had site-specific data on release rates and conducted the full analysis, so the uncertainty is lower than for segment 7. However, the spatial distribution of the release rate studies was narrow, so we judged our assessment of uncertainty as medium.

<u>Other scenarios</u>

Because the hydrologic driver (i.e., decrease in water elevation) decreases in strength, the uncertainty rankings also decrease as we move down through the ranked scenarios. The patterns of uncertainty described above could still apply to Half1995NN, the next scenario on the ranking, although the uncertainties should be lower by one level to reflect the weaker hydrologic driver (Figure 5–3). The next scenario, Full1995PN, has very little decrease in water elevation; the Biogeochemistry Working Group agreed it was appropriate to reduce the levels of uncertainty again to reflect the increasing certainty that effects would be negligible (Figure 5–4). The remaining scenarios all have no modeled decrease in water elevation so uncertainty would be very low in all cells for all remaining scenarios.

Scenario – Half1995NN

River Segment	Δ DOC loading	Δ TP loading	Δ TKN loading	Δ Dissolved Oxygen	Overall effect
1	*	*	*	*	*
2	*	*	*	*	*
3	*	*	*	*	*
4	*	*	*	*	*
5	*	*	*	*	*
6	*	*	*	*	*
7	***	***	***	***	***
8	**	**	**	**	**

Level of Effect

- Negligible
- Minor
- Moderate
- Major
- Extreme

Cross-hatching indicates abbreviated analysis

Uncertainty

*	Very Low
**	Low
***	Medium
****	High
*****	Very high

Figure 5–3. Summary and characterization of potential effects for scenario Half1995NN including indication of uncertainty level. DOC = dissolved organic carbon, TP = total phosphorus, and TKN = total Kjeldahl nitrogen.

Scenario – Full1995PN

River Segment	Δ DOC loading	Δ TP loading	Δ TKN loading	Δ Dissolved Oxygen	Overall effect
1	*	*	*	*	*
2	*	*	*	*	*
3	*	*	*	*	*
4	*	*	*	*	*
5	*	*	*	*	*
6	*	*	*	*	*
7	**	**	**	**	**
8	*	*	*	*	*

Level of Effect

- Negligible
- Minor
- Moderate
- Major
- Extreme

Uncertainty

*	Very Low
**	Low
***	Medium
****	High
*****	Very high

Cross-hatching indicates abbreviated analysis

Figure 5–4. Summary and characterization of potential effects for scenario Full1995PN including indication of uncertainty level. DOC = dissolved organic carbon, TP = total phosphorus, and TKN = total Kjeldahl nitrogen.

5.5 RECOMMENDATIONS

The conclusions reached in the biogeochemistry analysis depend on the accuracy of several hydrologic and ecohydrologic models. Moreover, forecast test scenarios have conditions that have not yet occurred or are unlikely to occur. Given the uncertainties associated with modeling and forecasts of future conditions, adaptive management is warranted. Monitoring is a critical component of adaptive management and it would be prudent to monitor the hydrologic and hydrodynamic drivers and ecological attributes. With respect to water quality attributes, large interannual variation will make change difficult to detect. Current monthly sampling of lake inflows and outflows would not be sufficient to confirm or reject the presence of any but extreme effects within any reasonable time period (years) in order to be used in an adaptive management strategy. Water quality sampling should also take place in wetlands and with rainfall events. Perhaps remote sensing could effectively and affordably allow tracking of the colored DOC (Brezonik et al. 2005; Sawaya et al. 2003; Kloiber et al. 2002). This would have the potential advantage of showing how much DOC derives from exchanges with adjacent wetlands as opposed to further upstream. Additionally, because DOC is highly correlated with many water quality variables in storm pulses in the upper basin, it may be a useful surrogate for several water quality constituents of interest.

6 ACKNOWLEDGEMENTS

We wish to acknowledge and thank the members of the National Research Council of the National Academy of Sciences:

Patrick L. Brezonik, Chair, University of Minnesota;
M. Siobhan Fennessy, Kenyon College;
Ben R. Hodges, University of Texas;
James R. Karr, University of Washington;
Mark S. Peterson, University of Southern Mississippi;
James L. Pinckney, University of South Carolina;
Jorge I. Restrepo, Florida Atlantic University;
Roland C. Steiner, Washington Suburban Sanitary Commission;
J. Court Stevenson, University of Maryland;
Laura Elders, Study Director for the National Research Council;

for their invaluable feedback and guidance which substantially improved the science presented in this chapter. We also gratefully acknowledge the William Beardall Tosohatchee State Reserve and Personnel for providing access to reserve properties; to staff from both the University of Florida and SJRWMD Palm Bay Service Center for dirty work under diverse conditions ranging from surprisingly numbing cold to brutal relentless heat. We also express our appreciation to the many GIS analysts who do magic with multiple and diverse sources of data and metadata, especially Sandra Fox, who made our LiDAR usable and accurate in herbaceous wetlands.

7 REFERENCES

Aitkenhead, J.A. and W.H. McDowell. .2000. Soil C:N Ratio as a Predictor of Annual Riverine DOC Flux at Local and Global Scales. GLOBAL Biogeochemical Cycles. 14:127-138

Alderidge, F. J. and C. L. Schelske. 2000. Study of Light and Nutrient Limitation on Phytoplankton in Lake Washington. Report to St. Johns River Water Management District

Aldous, A., P. Mccormick, C. Ferguson, S. Graham, and C. Craft. 2005. Hydrologic regime controls soil phosphorus fluxes in restoration and undisturbed wetlands. Restoration Ecology **13:** 341-347.

Baldwin, D. S., and A. M. Mitchell. 2000. The effects of drying and re-flooding on the sediment and soil nutrient dynamics of lowland river-floodplain systems: a synthesis. Regulated Rivers Research and Management **16:** 457-467.

Bass, D. G., and D. T. Cox. 1988. River habitat and fishery resources of Florida, p. 121-187. *In* W. Seaman, Jr. [ed.], Florida aquatic habitat and fishery resources Florida Chapter of the American Fisheries Society.

Belanger, T. V., F. E. Dierberg, and J. Roberts. 1985. Dissolved Oxygen Concentrations in Florida's Humic-Colored Waters and Water Quality Standard Implications. Fla. Sci. **48:** 107-119.

Benke, A. C., I. Chaubey, G. M. Ward, and E. L. Dunn. 2000. Flood Pulse Synamics of an Unregulated River Floodplain in the Southeastern U.S. Coastal Plain. Ecology **81:** 2730-2741.

Berg, B. 2000. Litter Decomposition and Organic Matter Turnover in Northern Forest Soils. Forest Ecology and Management. 133:13-22

Brady, N.C. and R.R. Weil. 2008. The Nature and Properties of Soils. Prentice Hall. The University of Virginia. 965 pages

Brezonik, P.L., K. Menken, and M. Bauer. 2005. Landsat-based Remote Sensing of Lake Water Quality Characteristics, Including Chlorophyll and Colored Dissolved Organic Matter (CDOM). Lake and Reservoir Management. 21:373-382

Burt, T. P., and G. Pinay. 2005. Linking hydrology and biogeochemistry in complex landscapes. Progress in Physical Geography 29: 297-316.

Carpenter, S. R., and M. L. Pace. 1997. Dystrophy and Eutrophy in Lake Ecosystems: Implications of Fluctuating Inputs. Oikos **78:** 3-14.

Cox, D. T., and E. D. Vosatka. 1976. Stream Investigations: Upper St. Johns River study 1971-1976. Completion Report for Investigations Project as required by Federal Aid in Fish Restoration, Dingell-Johnson Project F-25. Florida Game and Fresh Water Fish Commission.

Craft, C. B., and C. J. Richardson. 1993. Peat accretion and N,P, and organic C accumulation in nutrient-enriched and unenriched Everglades peatlands. Ecol. Appl. **3:** 446-458.

DB Environmental, I. 2007. Sediment Oxygen Demand in Six Lakes of the Upper St. Johns River Basin. St. Johns River Water Management District.

DeBusk, W. F. and K. R. Reddy. 2003. Nutrient and hydrology effects on soil respiration in a northern everglades marsh. J. Environ. Qual. 32(2):702–710.

Demort, C. L. 1990. The St. Johns River System, p. 97-120. In R. J. Livingston [ed.], The Rivers of Florida. Springer-Verlag. New York, Inc.

Dunne, E. J., J. Smith, D. B. Perkins, M. W. Clark, J. W. Jawitz, and K. R. Reddy. 2007. Phosphorus storages in historically isolated wetland ecosystems and surrounding pasture uplands. Ecol. Eng. **31:** 16-28.

Ensign, S. H., and M. W. Doyle. 2005. In-channel transient storage and associated nutrient retention: Evidence from experimental manipulations. Limnol. Oceanogr. **50:** 1740-1751.

Fisher, M. M., S. J. Miller, A. D. Chapman, and L. W. Keenan. 2009. Phytoplankton dynamics in a chain of subtropical blackwater lakes: the Upper St. Johns River, Florida, USA. Lake Reserv. Manage. **25:** 73-86.

Gao, X. 2005. Nutrient and DO TMDLs for the St. Johns River above Lake Poinsett (WBID 2893L), Lake Hell n' Blazes (WBID 2893Q), and St. Johns River above Sawgrass Lake (WBID 2893X), p. 44 pp. Florida Department of Environmental Protection.

Goolsby, D. A., and B. F. Mcpherson. 1970. Preliminary evaluation of chemical and biological characteristics of the upper St. Johns River Basin, Florida, p. 74. *In* U. S. G. Survey [ed.].

Graham, S. A., C. B. Craft, P. V. Mccormick, and A. Aldous. 2005. Forms and accumulation of soil P in natural and recently restored peatlands - Upper Klamath Lake, Oregon, USA. Wetlands **25:** 594-606.

Hamilton, S. K., S. J. Sippel, D. F. Calheiros, and J. M. Melack. 1997. An anoxic event and other biogeochemical effects of the Pantanal wetland on the Paraguay River. Limnol. Oceanogr. **42:** 257-272.

Hanson, Kaj. 1962. The Dystrophic Lake Type. Hydrobiologia, 19:183-191.

Holden, J. 2005. Peatland hydrology and carbon release: why small-scale process matters. Philosophical Transactions of the Royal Society A **363:** 2891-2913.

Holden, J., P. J. Chapman, and J. C. Labadz. 2004. Artificial drainage of peatlands: hydrological and hydrochemical process and wetland restoration. Progress in Physical Geography **28:** 95-123.

Hutchinson, G.E. 1975. A Treatise on Limnology, Vol. III, Limnological Biology. Wiley, New York. pp 660

Jensson B.H. 1996. Nitrogen mineralization in relation to C:N ratio and decomposability of organic materials. Plant and Soil. 181:39-45, DOI: 10.1007/BF00011290

Junk, Wolfgang J. 1997. The central Amazon floodplain: ecology of a pulsing system. Ecological Studies, 126. Springer-Verlag. Berlin Heidelburg

Junk, W. J., P. B. Bayley, and R. E. Sparks. 1989. The Flood Pulse Concept in River-Floodplain Systems, p. 110-127. *In* D. P. Dodge [ed.], Proceedings of the International Large River Symposium. Canadian Special Publications of Fisheries and Aquatic Sciences.

Kadlec, R. H., and S. D. Wallace. 2008. Treatment Wetlands. CRC Press.

Keenan, L., E. Lowe, and D. Dobberfuhl. 2002. Pollutant load reduction goals for the Upper St. Johns River Basin, p. 33. St. Johns River Water Management District.

Keenan, L. W., M. M. Fisher, and S. J. Miller. 2008. Chronic and Acute Hypoxia in the Upper St. Johns River. Proceedings of the Florida Lake Management Society Annual Meeting June 2-5, 2008.

Kloibea, S.M., P.L. Brezonik, and M.E. Bauer. 2002. Application of Landsat imagery to egional-scale assessments of lake clarity. Water Research 36:4330-4340.

Knight, R. L., R. W. Ruble, R. H. Kadlec, S. C. Reed, J. D. Waterman*, and D. S. Brown*. 1994. Treatment Wetland Habitat And Wildlife Use Assessment And North American Treatment Wetland Database V 2.0. U.S. Environmental Protection Agency, Washington, DC, EPA/600/C-94/002 (NTIS PB2007500012), 1994.

Lowe, E. F., J. E. Brooks, C. J. Fall, L. R. Gerry, and G. B. Hall. 1984. U.S. EPA clean lakes program, phase I diagnostic-feasibility study of the upper St. Johns River chain of lakes - volume 1- diagnostic study. Technical Publication SJ 84-15. St. Johns River Water Management District.

Mace, J. W. 2006. Minimum flows and levels determination: St. Johns River at State Road 44 near Deland, Volusia County. St. Johns River Water Management District Technical Publication SJ2006-5

---. 2007a. Minimum flows and levels determination: St. Johns River at State Road 50, Orange and Brevard Counties, Florida. St. Johns River Water Management District Technical Publication SJ2007-1.

---. 2007c. Minimum levels determination: Lake Poinsett in Brevard, Orange, and Osceola counties. Unpublished manuscript St. Johns River Water Management District.

Martin, H. W., D. B. Ivanoff, D. A. Graetz, and K. R. Reddy. 1997. Water table effects on Histosol drainage water carbon, nitrogen, and phosphorus. Journal of Environmental Quality 26: 1062-1071.

McGrail, L., Berk, K., Brandes, D., Munch, D., Neubauer, C., Osburn, W., Rao, D., Thomson, J. & Toth, D 1998. St. Johns River Water Management District, p. 214-237. In E. A. Fernald and E. D. Purdum [eds.], Water Resources Atlas of Florida. Florida State University Institute of Science and Public Affairs.

Miles, C.J. and P.L. Brezonik 1981. Oxygen consumption in humic-colored waters by a photochemical ferrous-ferric catalytic cycle. Environmental Science and Technology, 15, 1089-1095.

Mladenov, N., and D. M. Mcknight. 2003. The Influence of Hydrologic Processes on the Transport and Biochemical Transformation of Aquatic Organic Matter in the Okavango Delta, Botswana. *In* A. G. Union [ed.]. The Smithsonian/NASA Astrophysics Data System.

National Academy of Sciences. 2002. Riparian Areas: Functions and Strategies for Management.

Qualls, R. G., and C. J. Richardson. 2003. Factors controlling concentration, export, and decomposition of dissolved organic nutrients in the Everglades of Florida. Biogeochemistry **62:** 197-229.

Reddy, K. R., and E. M. Dangelo. 1997. Biogeochemical indicators to evaluate pollutant removal efficiency in constructed wetlands. Water Sci. Technol. **35:** 1-10.

Reddy, K. R., R. D. DeLaune, W. F. DeBusk, and M. Koch. 1993. Long-term nutrient accumulation rates in the Everglades wetlands. Soil Sci. Soc. Am. J. 57:1145–1155.

Reddy, K. R., and R. D. Delaune. 2008. Biogeochemistry of Wetlands, Science and Applications. CRC Press.

Reddy, K. R., T. Z. Osborne, K. S. Inglett, and R. Corstanje. 2007. Influence of water levels on subsidence of organic soils in the Upper St. Johns River basin. Special Publication SJ2007-SP5. St. Johns River Water Management District

Restrepo, J. I., A. M. Montoya, and J. T. Obeysekera. 1998. A wetland simulation module for the MODFLOW ground water model. Ground Water 36: 764-770.

Sawaya, K.E., L.G. Olmanson, N.J. Heinert, P.L. Brezonik, and M.E. Bauer. 2003. Extending satellite remote sensing to local scales: land and water resource monitoring using high-resolution imagery. Remote Sensing of Environment. 88:144-156

Schiff, S., Aravena, R., Mewhinney, E., Elgood, R., Warner, B., Dillon, P. & Trumbore, S. E. 1998. Precambrian shield wetlands: hydrologic control of the sources and export of dissolved organic matter. Climate Change 40: 168-188.

Shannon, E. E., and P. L. Brezonik. 1972. Limnological Characteristics of North and Central Florida Lakes. Limnol. Oceanogr. **17:** 97-110.

Snyder, G. H. 2005. Everglades Agricultural Area soil subsidence and land use projections. Soil and Crop Science Society of Florida Proceedings **64:** 44-51.

St. Johns River Water Management District. 2004. 2004. Digital Orthophotography. Palatka, Florida.

St. Johns River Water Management District. 2007. SSURGO Soils for SJRWMD and Surrounding Regions. Spatial data layer from original Natural Resources Conservation Service (NRCS) data. Palatka, Florida

St. Johns River Water Management District, 2011. Land Cover and Land Use 2004. Second Edition. Spatial data layer. Imagery from December 2003 to March 2004. Palatka, Florida.

Soil Survey Staff, Natural Resources Conservation Service, United States Department of Agriculture. Official Soil Series Descriptions. Available online at http://soils.usda.gov/technical/classification/osd/index.html. Accessed 2010.

Stephens, J. C. 1956. Subsidence of organic soils in the Florida Everglades. Journal Series No. 352, Institute of Food and Agricultural Sciences. University of Florida.

Tank, J. L., E. J. Rosi-Marshall, N. A. Griffiths, S. A. Entrekin, and M. L. Stephen. 2010. A review of allochthonous organic matter dynamics and metabolism in streams. Journal of North American Benthological Society **29:** 118-146.

Tao, Shu, Bin Lin, Xiaohang Liu, and Jun Cao. 2000. Release Kinetics of water soluble organic carbon (WSOC) from river sediment and wetland soil. Water, Air, and Soil Pollution **118**:407-418

Thorp, J. H. 1994. The riverine productivity model: an heuristic view of carbon sources and organic processing in large river ecosystems. Oikos **70:** 305-308.

Tockner, K., F. Malard, and J. V. Ward. 2000. An extension of the pulse concept. Hydrol. Process. **14:** 2861-2883.

US Depatment of Agriculture. Natural Resources Conservation Service. Plants Database, online http://plants.usda.gov/java/

US Depatment of Agriculture. 1938-1944. Agricultural Adjustment Administration. 1:20,000 scale Black and white aerial photographs, Currently available from National Archives and Records Administratin, college Park, MD

US Depatment of Agriculture. 1987 – 2004. National Aerial Photography Program Color infrared aerial photographs. 1:40,000 scale. Currently available from USDA Aerial Photography Field Office, Salt Lake City, UT

Vannote, R. L., G. W. Minshall, K. W. Cummins, J. R. Sedell, and C. E. Cushing. 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. **37:** 130-137.

Vidon, P., C. Allan, D. Burns, T.P. Duval, N. Gurwick, S. Inamdar, R. Lowrance. J. Okay, D. Scott, and S. Sebestyen. 2010. Hot spots and hot moments in riparian zones: potential for improved water quality management. J. Am. Water Resourc. As. **46:** 278-298.

Webster, J. R., and B. C. Patten. 1979. Effects of watershed perturbation on stream potassium and calcium dynamics. Ecol. Monogr. **49:** 51-72.

Wetzel, R. G. 1984. Detrital dissolved and particulate organic carbon functions in aquatic ecosystems. Bull. Mar. Sci. **35:** 503-509.

---. 1992. Gradient-dominated ecosystems: sources and regulatory functions of dissolved organic matter in freshwater ecosystems Hydrobiologia **229:** 181-198.

Wilson, J. S., D. S. Baldwin, G. N. Rees, and B. P. Wilson. 2011. The effects of short-term inundation on carbon dynamics, microbial community structure and microbial activity in floodplain soil. River Research and Applications **27:** 213-225.

Yurova, A., A. Sirin, I. Buffam, K. Bishop, and H. Laudon. 2008. Modeling the dissolved organic output from a boreal mire using the convection-dispersion equation: Importance of representing sorption. Water Resour. Res. **44:** 1-15.

8 Appendices for Chapter 7. Biogeochemistry

Appendix 7A. Synoptic Soil Survey Maps, Data, and Summary
Appendix 7B. Identification of Methods for Sample Analysis
Appendix 7C. Wetland Constituent Release Model
Appendix 7D. Wetland Constituent Reduction Model
Appendix 7E. Blue Cypress Soil Release Calculations and Model Simulations