

ORGANIC SOIL SUBSIDENCE AND NUTRIENT LEACHING IN AN IMPACTED
FLORIDA WETLAND

By

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LIST OF ABBREVIATIONS

C	Carbon
CO ₂	Carbon Dioxide
DOC	Dissolved Organic Carbon
FID	Flame Ionization Detector
GC	Gas Chromatograph
ICP	Inductively Coupled Plasma
N	Nitrogen
OM	Organic Matter
P	Phosphorus
PVC	Polyvinyl Chloride
SJMCA	St Johns Marsh Conservation Area
TC	Total Carbon
TDKN	Total Dissolved Kjeldahl Nitrogen
TDP	Total Dissolved Phosphorus
TEA	Terminal Electron Acceptor
TN	Total Nitrogen
TP	Total Phosphorus
TS	Total Sulphur
USJRB	Upper St Johns River Basin

Abstract of Thesis Presented to the Graduate School
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Current water management practices in the Upper St. Johns River Basin have resulted in significant losses of wetland soils in the St. Johns Marsh Conservation Area (SJMCA) to subsidence. Because organic soils accrete at a very slow rate and oxidize rapidly under drained conditions, this soil subsidence is of major concern to water managers. With this permanent loss of soil material to oxidation comes the mineralization of other nutrients that contribute to downstream water quality degradation. Managers have relied on historic soil subsidence rates established as early as 1956 in the Everglades Agriculture Area which have been used to model soil loss in the SJMCA. While these well documented subsidence rates are certainly valid for the environments in which they were determined, their applicability to other systems is questionable. Therefore, we seek to model the effect of hydrologic status on soil subsidence for soils in the SJMCA. Field and laboratory studies were conducted in which CO₂ and CH₄ fluxes were monitored over changing hydrologic conditions. A separate lab study determined a subsidence rate that can be modeled along with gaseous carbon losses.

CHAPTER 1 INTRODUCTION

Wetlands, being ecosystems of significant biological productivity, are major contributors to the global carbon cycle. In response to changes in hydrologic, biologic, climatologic, or chemical conditions, wetlands can serve as either a source or a sink for carbon (Schedlbauer 2010). It is well established that when wetlands are flooded, the ability of microorganisms to use oxygen in their metabolic processes is greatly reduced (Schedlbauer et al. 2010; Freeman et al. 1996; Lafleur et al. 2005). While some microbes can use alternate oxidants like nitrate, Fe^{3+} or even sulfate, the rate of decomposition using these oxidants is much slower than for these microbes that use O_2 . Thus, when a wetland is flooded, soil organic matter (SOM) can build up in the soil (due to slower oxidation rates) causing an overall increase in soil volume (accretion). Conversely, when floodwater is removed, organic matter can be rapidly oxidized and released to the atmosphere as CO_2 . This loss of carbon to the atmosphere is one of the major causes of soil subsidence in wetlands, which can be defined as the lowering of surface elevation after drainage due to causes other than erosion (Parent et al. 1977).

Total subsidence rates in excess of $2 \text{ cm}^{-1} \text{ yr}$ have been shown to occur in drained Everglades soils (Reddy and Delaune 2008). In a previous study in the Everglades, Stephens (1956) found a slightly higher rate of approximately 1.25 in yr^{-1} (3.175 cm yr^{-1}). Subsidence rates from 3.4 (Schipper et al. 2002) to $0.6\text{-}1 \text{ cm yr}^{-1}$ (Schothorst 1977) have been shown to exist as well in New Zealand pasture and a Netherland moor respectively.

According to Vepraskas and Ewing (2006), total subsidence is a function of two processes, termed primary and secondary subsidence. Primary subsidence results from

the loss of the buoyant force provided by water; once pore water leaves the soil system, the support it provides to the overlying particles is lost. When air fills the pore spaces once occupied by water, the soil may compact under its own weight. The process of primary subsidence was estimated by Vepraskas and Ewing (2006) to account for 4 cm yr⁻¹ or 69% of total subsidence in a North Carolina agricultural field. Secondary subsidence is caused by oxidation of the organic carbon to inorganic carbon that can be lost to the atmosphere as CO₂. This process can occur through biotic means like microbial decomposition or through abiotic processes like fire. Vepraskas and Ewing (2006) estimate the soil lost to secondary subsidence to account for the remaining 1.9 cm yr⁻¹ or 31% of total subsidence for the same North Carolina histosol mentioned earlier. It appears that complete consensus has not been reached on the magnitude of post-drainage soil loss that can be expected. Much variability exists between subsidence rate estimates based on experimental methods, geography, pedology, and many other environmental conditions. Therefore, any models attempting to incorporate soil subsidence for environmental management purposes must take into account all of the previously mentioned factors, as small differences in subsidence rates could result in misleading results.

Once an organic soil is drained, both primary and secondary subsidence may act to lower the soil surface elevation. Literature rates for subsidence are listed in Table 1-1. As time progresses after drainage, the relative role of each fraction of subsidence will change. According to Wösten et al. (1997) and Vepraskas and Ewing (2006), the process of soil subsidence can be broken down into two main factors: compaction and shrinkage (primary subsidence), and oxidation (secondary subsidence). During the first

5-10 years of water table drawdown, compaction and shrinkage dominate relative to oxidation. In a study by Schothorst (1977) primary subsidence accounted for 1-4 cm of the 4-10 cm total subsidence over 6 years for peat in the Netherlands. One of the primary factors in physical subsidence is the destruction of macropore structure. Kechavarzi et al. (2010) demonstrated that peat soils that have retained natural hydrology contain more macropores than peat that has been drained. As water in flooded soil exerts a buoyant force on soil particles, upon draining, this force is removed, allowing the soil particles to settle and consolidate (Drexler et al. 2009). This physical compaction of drained portions in peat not only results in a lowering of the surface, but also causes an increase in bulk density (Drexler et al. 2009) and subsequent decrease in saturated hydraulic conductivity. As both density and conductivity influence plant and microbial activity in the rhizosphere, and these factors also influence subsidence, many feedback systems may exist, making the prediction of subsidence rates very difficult.

Table 1-1. Selected literature subsidence rates

Source	Region	Soil Type	Subsidence rate (cm yr ⁻¹)
Stephens 1956	Everglades	Peat	3.18
Ewing and Vepraskas 2006	North Carolina	Organic agricultural	4
Schipper et al. 2002	New Zealand	Peat bog	3.4
Rojstaczer and Deverel 1996	Northern California	Alluvial sediment	2.3 - 1.5
Wosten et al. 1997	Malaysia	Peat	2
Parent et al. 1977	Quebec	Muck	0.99-4.53
Reddy et al. 2006	Florida (Blue Cypress Marsh)	Peat	0.25 – 0.5 (only secondary subsidence)
Drexler et al. 2009	Northern California	Peat	3.5 - 8.2
Average literature subsidence			2.9
This Study	Florida	Peat	4.7

Oxidation, or mineralization of organic carbon to CO₂ is the primary cause of secondary subsidence. Under aerobic conditions (as when the water table is below the soil surface), this process can be accomplished by microorganisms using O₂ as an electron acceptor. The production of CO₂ during OM oxidation results in the permanent loss of carbon from the soil system, and contributes to overall soil subsidence (Reddy et al. 2006; Berglund and Berglund 2011; Vepraskas and Ewing 2006; Morris et al. 2004). While flooded, carbon mineralization is primarily a function of the oxygen diffusion rate. Once drained, the rate limiting factor shifts from terminal electron acceptor availability to temperature (Kechavarzi et al. 2010). At temperatures over 20°C, the rate of OM decomposition roughly doubles for every 10°C increase in temperature (Q₁₀~2). As temperature drops below 20°C, Q₁₀ increases exponentially, with a theoretical limit of Q₁₀ approaching ∞ at 0°C (Kirschbaum 1995). While the actual rate of decomposition decreases with decreasing temperature, the temperature sensitivity of decomposition increases with decreasing temperature. For example, if soil temperatures increase from 10 °C to 20 °C, the rate of SOM oxidation will increase by a larger factor than if the temperature increases from 20 °C to 30 °C (Inglett et al. 2011).

In addition to loss of soil carbon, aerobic decomposition also can lead to a release of inorganic nutrients that might otherwise remain in the soil under flooded conditions (Reddy and DeLaune 2008; Osborne et al. 2011). Wetlands play a large role in this nitrogen cycle; anaerobic conditions allow denitrifying microbes to compete for organic matter which facilitates the reduction of NO₃⁻ to N₂ gas (process of denitrification) which can be lost to the atmosphere (Bowden 1987; Knowles 1982). This is the primary means of N loss from the soil system, so without anaerobic soils,

inorganic N could freely move from upland soils into aquatic systems. D'Angelo and Reddy (1994) found the rates of denitrification in Lake Apopka Marsh (a hypereutrophic constructed wetland) to be between 20 and 50 mg N kg⁻¹ day⁻¹. According to Reddy and DeLaune (2008) the rate of organic nitrogen (ON) mineralization when O₂ is present is roughly 2.6 times greater than when the wetland is anaerobic. A previous study showed that for Everglades surface soil (0-10 cm), ON mineralization when O₂ was present was 8.4 times as high as when NO₃⁻ was used as a terminal electron acceptor (TEA) (White and Reddy 1997). The inorganic nitrogen (NH₄⁺) released by the mineralization of organic nitrogen can be converted to NO₃⁻ by various aerobic soil microbes (White and Reddy 1997). This nitrate released can be lost from the wetland and transferred to N-limited ecosystems like most lakes, rivers, and estuaries, ultimately leading to eutrophication and fish kills. Fish kills and harmful algal blooms on the St Johns River over the summer season have become a common yet environmentally and economically destructive force (Chapman et al. 2001; Paerl 2007; Steidinger et al. 1998; Maher 1997).

Phosphorus (P) is another element that plays a large role in shaping the chemical environment of wetlands. P is often a limiting nutrient in freshwater wetlands, meaning that increases in its concentration will lead to proportionate increases in biologic activity. As many Florida wetlands are already low P systems, any additional P may result in a change in trophic status (Reddy et al. 1999). This P-induced eutrophication of previously oligotrophic systems is epitomized in the recent history of the Florida Everglades. Here, P-loading has caused a significant change in microbial

ecology (White and Reddy 2001), resulting in altered sediment dynamics (Reddy and De'Laune 1993) and macrophyte community structure (Lorenzen et al. 2001).

Unlike nitrogen, there is no significant gaseous form of P, so any P added to a wetland will remain in the system unless it is removed through some exogenous process. Its availability in wetlands is governed mainly by adsorption-desorption and redox reactions occurring on soil particles (Twinch 1986; Cadenhead 1997; White and Reddy 2001). The degree to which P will adsorb or desorb in the soil is primarily a function of the difference between P concentrations in solution and in the soil, although other factors like Fe and Ca concentration also play a large role. If the total dissolved phosphorus (TDP) concentration is greater than the equilibrium P concentration ($EPC=[P]$ at which net adsorption equals 0), then P will be taken up by the soil. On the other hand, when TDP is less than EPC, the soil will release P to the water column. Other mechanisms influencing the mobility of phosphorus in soil include pH and Eh; both of these factors demonstrate a strong negative relationship with P availability (Reddy and DeLaune 2008). During periods of water table drawdown, aerobic conditions can extend into wetland soils, causing microbially associated P to be rapidly mineralized and lost to the water column. A study by Olila et al. (1997) showed that a wetland drained for 6 weeks released roughly 10 times as much dissolved reactive phosphate (DRP, also known as soluble reactive phosphorus (SRP)) as did a wetland that was drained for just 3 weeks. This study also showed that the availability of phosphorus within the soil intensified with increased drainage time. The phosphorus released during the drained period was highly labile and could possibly contribute to downstream eutrophication. A 1986 study by Twinch showed a slightly more complex

effect of drying on P exchange characteristics. In this study, sediment samples from a hypereutrophic lake were taken, with half being allowed to air dry, and the other half were stored wet. Although drying decreased the amount of bioavailable P in the sediment, the most notable effect of drying was to: increase labile P by 127%, increase EPC by 199%, and decrease the P buffer capacity by 20%.

Overall, the biogeochemistry of nutrients (primarily N and P) is strongly linked with water table dynamics. When flooded, wetlands can effectively remove nitrogen from the water column, and have the ability to sequester phosphorus (depending on various soil physical and chemical characteristics). Once drained, the process of denitrification halts, and bound phosphorus can be released (Olila et al. 1997; Twinch 1987; Reddy and DeLaune 2008). The biogeochemistry of carbon is also controlled strongly by hydrologic factors; when flooded, carbon generally remains in the soil system, but can be lost through oxidation when water is removed.

Environmental/Social Significance

Throughout the history of Florida's development, a major theme has been the draining and reclamation of wetland ecosystems. The Upper St Johns River Basin (USJRB) is no exception; altered hydrology has caused severe environmental degradation in the years since its reclamation in the early 20th century. Historically, wetlands covered almost 1800 km² of the USJRB. As a result of decades of draining and reclamation, much of the original USJRB wetland area was lost. After reaching a low of < 300 km² in 1985, the SJRWMD now manages roughly 590 km² of wetlands that are in various states of degradation (Brenner et al. 2001). The primary services of these wetlands are to ensure: flood protection, environmental enhancement, water supply, and recreation. The effect of draining status on soil physical properties is well

documented, with the overall end result being a “degradation of the soil structure and a reduction in the soil capacity to store, retain, and transmit water” (Kechavarzi et al. 2010). Drexler et al. (2009) noted that when wetlands in the San Joaquin Delta were drained for farming, bulk density increased significantly in the surface layer.

Often, changes in the physical properties (often bulk density and hydraulic conductivity) of peat soils after draining are permanent, and cannot be significantly restored even if the historic hydrologic conditions are reestablished (Holden et al. 2006). While Price (2003) noted a slight increase in hydraulic conductivity after peat soil was rewetted, this followed a decrease in conductivity of up to two orders of magnitude after draining. In wetlands like the SJMCA that are managed for flood control, the ability for peat to quickly assimilate and store water is vital. Holden et al. (2006) show that hydrographs for drained wetlands demonstrate quicker runoff responses to rainfall. While this response may have been beneficial when the wetlands were first drained for agricultural purposes, it now means that the wetland may be less effective at preventing flooding downstream.

Over the last few decades, fish kills and harmful algal blooms have become a common occurrence on the St Johns River (Steidinger et al. 1998; Chapman et al. 2001). Both of these events have a similar cause: excess nutrients. According to Chapman et al. (2001) cyanobacteria are the primary culprits of harmful algal blooms (HABs) in Floridian freshwater systems, and their occurrence is often predicated by changes in N or P limitations within the river. These Cyanobacterial blooms are not only unsightly, but they also may produce toxins that can attack the human liver, skin, and nervous system. Prolonged exposure to these HABs can lead to conditions ranging

from “swimmer’s itch” to paralysis and liver failure. The harmful effect of HABs are not limited to humans; frequently the occurrence of a HAB is followed by hypoxia and associated fish/invertebrate kills. In August of 1999, an *Anabaena* bloom was implicated in causing a fish kill where over 10,000 menhaden, *Brevoortia Tyrannus*, and *B. Smithi* died (Chapman et al. 2001). As St Johns River HABs are often linked with nutrient leaching from upstream wetlands, it is important to determine how hydrologic management effects nutrient movement.

St Johns Marsh Conservation Area:

The SJMCA is a region that contains primarily freshwater marshes, hardwood swamps, and small lakes. Hydrologic inputs to the SJMCA include precipitation, surface flow from the west, and channelized flow from the east and south. As seen in Figure 1-1, pasture land constitutes the western boundary of the wetland, while the eastern boundary consists of a water management and a water conservation area.

St Johns Marsh Conservation Area



SJMCA Field Sites



Figure 1-1. Location of study site

The C-40 canal was dug during a period in history when many Florida wetlands were being reclaimed for agriculture or development. The result of this canal has been a severe over-draining of the SJMCA. To combat the environmental degradation that has resulted from this over-draining, a series of 8 earthen dams were placed in the C-40 canal, although some have since been removed. The objective of this construction was to facilitate the re-flooding of the SJMCA. The average maximum and minimum annual air temperatures in the SJMCA are 27.5 and 17.3 °C respectively. The average annual precipitation 50.06 inches (Overpeck et al. 2010).

The St Johns Marsh Conservation Area (SJMCA) covers roughly 13,880 ha of marshes and lakes in Brevard County, Southeast Florida. This area has historically been subject to periods of severe draining and hydrologic isolation/alteration. Some of the primary results of this draining have been wetland system degradation, diminished water quality, and vegetation community shifts (commonly an invasion of high nutrient species like *Typha sp*) (Miller et al. 2009). Previous studies have shown that soil subsidence is occurring in the upper St Johns River Basin, although a rate for subsidence has not yet been established for this region (Reddy et al. 2006).

Objectives

The primary research objective of this study is to determine: 1) what are the major causes of soil subsidence, and 2) how can the current hydrologic management scheme best be modified to reduce or eliminate organic soil subsidence and allow the SJMCA to meet its requirements of: flood protection, environmental enhancement, water supply, and recreation. In this study, both field and laboratory experiments will be undertaken in order to build a subsidence model for soils in the SJMCA. As subsidence is related to carbon cycling, selected biogeochemical fluxes of carbon from the wetland to the atmosphere and hydrosphere will be determined.

CHAPTER 2 SOIL SUBSIDENCE IN THE SJMCA

Background Statement

When flooded, carbon fixed by wetland vegetation can be stored as organic matter in the soil. This process is termed accretion, and describes the buildup of organic matter that makes organic soils (muck, peat, and histosols). If flooding conditions are removed (either seasonally or anthropogenically), primary and secondary soil subsidence may occur, leading to a lowering of the soil surface elevation. According to Vepraskas and Ewing (2006), the significant factors influencing primary and secondary soil subsidence are: original soil thickness, depth to water table, mineral content, temperature, precipitation, and management practices. Other factors like wild fire can also cause rapid loss of soil carbon (Stephens 1956).

As secondary soil subsidence (loss of soil organic carbon) through aerobic decomposition) results in the permanent depletion of soil material, this process is of great concern for wetlands that rely on peat soils to aid in flood control. Holden et al. (2006) found that when large wetlands are drained, rainfall runoff became much more flashy and was transferred downstream with great efficiency.

The SJMCA is located in a subtropical region where intense rainfall is not uncommon. One of the primary purposes of the SJMCA is to catch runoff from high precipitation events like hurricanes, and prevent it from flooding nearby urban areas. Because previous management practices have altered the effectiveness of the wetland in flood prevention, it is vital to understand how these practices influence soil subsidence. When the factors leading to soil subsidence are better understood, the SJMCA can be managed to more effectively prevent floods and maintain water quality.

The overall objective of this research is to quantify the effect of current hydrologic management schemes on both primary and secondary subsidence. As many researchers claim that microbial oxidation of SOM is a major cause of subsidence (Morris et al. 2004; Reddy et al. 2006), a primary goal of this study will be to understand the relationship between hydrology and CO₂ flux (a measure of aerobic respiration). To accomplish this task, both field and laboratory studies were conducted in order to determine how carbon is cycled through the SJMCA, and what effect hydrology has on this cycling.

Methods

Field Methods

All of the field sites for this study are located within the SJMCA (Figure 1-1). The location of these sites were chosen to maximize elevation and vegetation gradients. Two transects of approximately 0.5 km each have been established along the main C-40 canal. These sites can be seen in Figure 2-1. Each Transect consists of three sites: one close to the canal, one at a moderate distance, and one far away from the canal, with the objective of creating a profile of wet to dry across the wetland. Four more sites were set up on the north side of the marsh where multiple vegetative communities existed in close proximity. The 4 vegetation communities chosen were: Willow (*Salix caroliniana*), Sawgrass (*Cladium jamaicense*), Cattail (*Typha spp.*), and open water. One last site (SJM3-CTR) was created on the western side of the SJMCA where the soil was characteristically dry. Thus, a total of 11 sites were established in the SJMCA. At each site, 20cm diameter PVC collars and 10cm diameter PVC pipes (4' tall) were installed in triplicate, as well as a single shallow (~1.5m) monitoring well. The 20cm diameter collars were 30cm in height, with the bottom half inserted into the soil; the top

half was left protruding from the soil to allow coupling of the Li-8100. These collars were left in place throughout the study so as to not disturb the soil when repeated measurements were taken. Each well was made by drilling a series of holes into a 5cm (2in) diameter, 1.2-1.5m long PVC pipe.

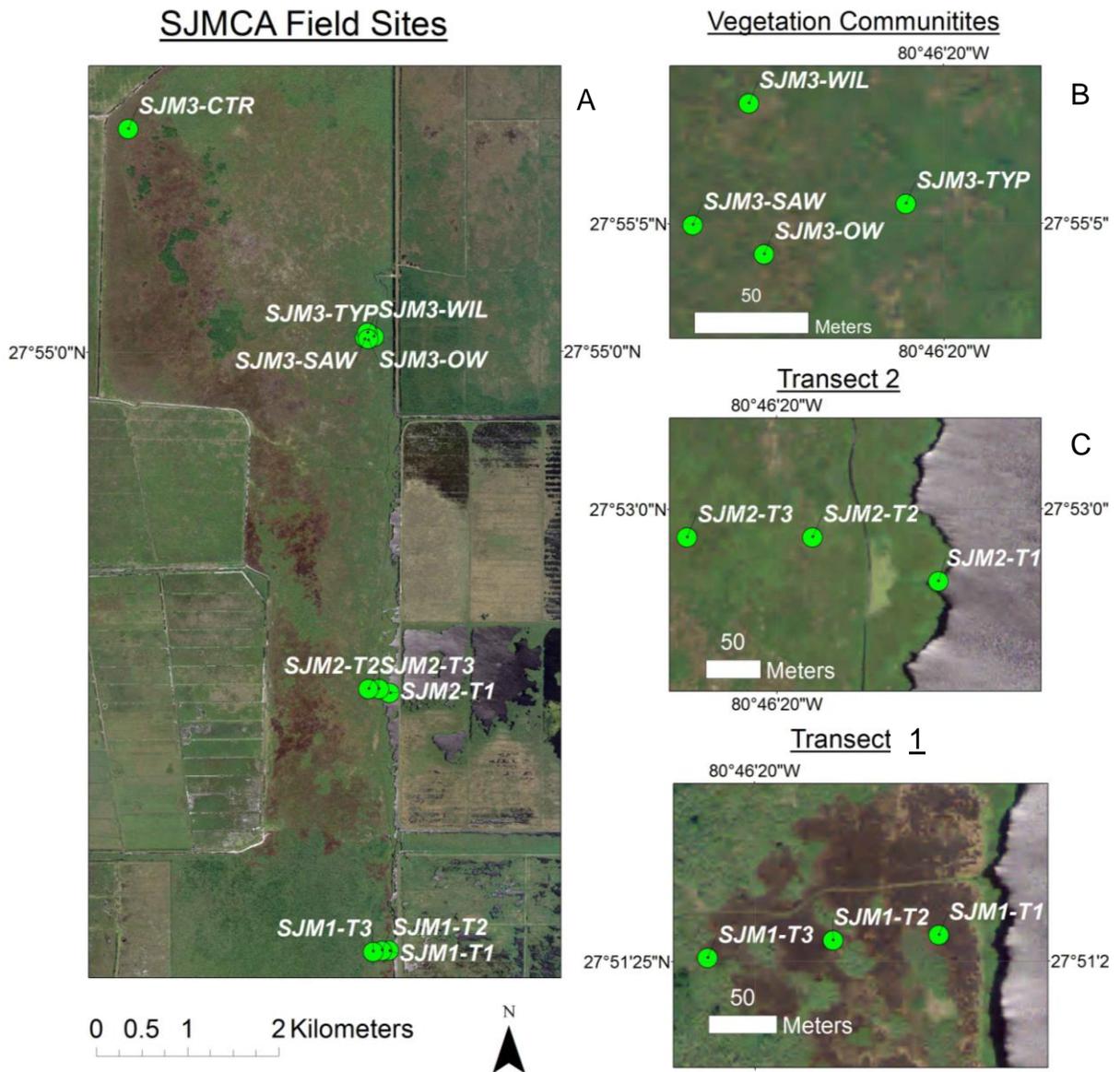


Figure 2-1. Map of field locations. A) Overall field sites in the SJMCA. B) Vegetation communities. C) Transect 2. D) Transect 1.

At the center of each field site, an auger just under 5cm (2in) in diameter was used to excavate soil to allow installation of the wells. Depth to water in the wells was measured using a tape measurer equipped with a water depth sounder. An image and diagram of an individual site can be seen in Figure 2-2.

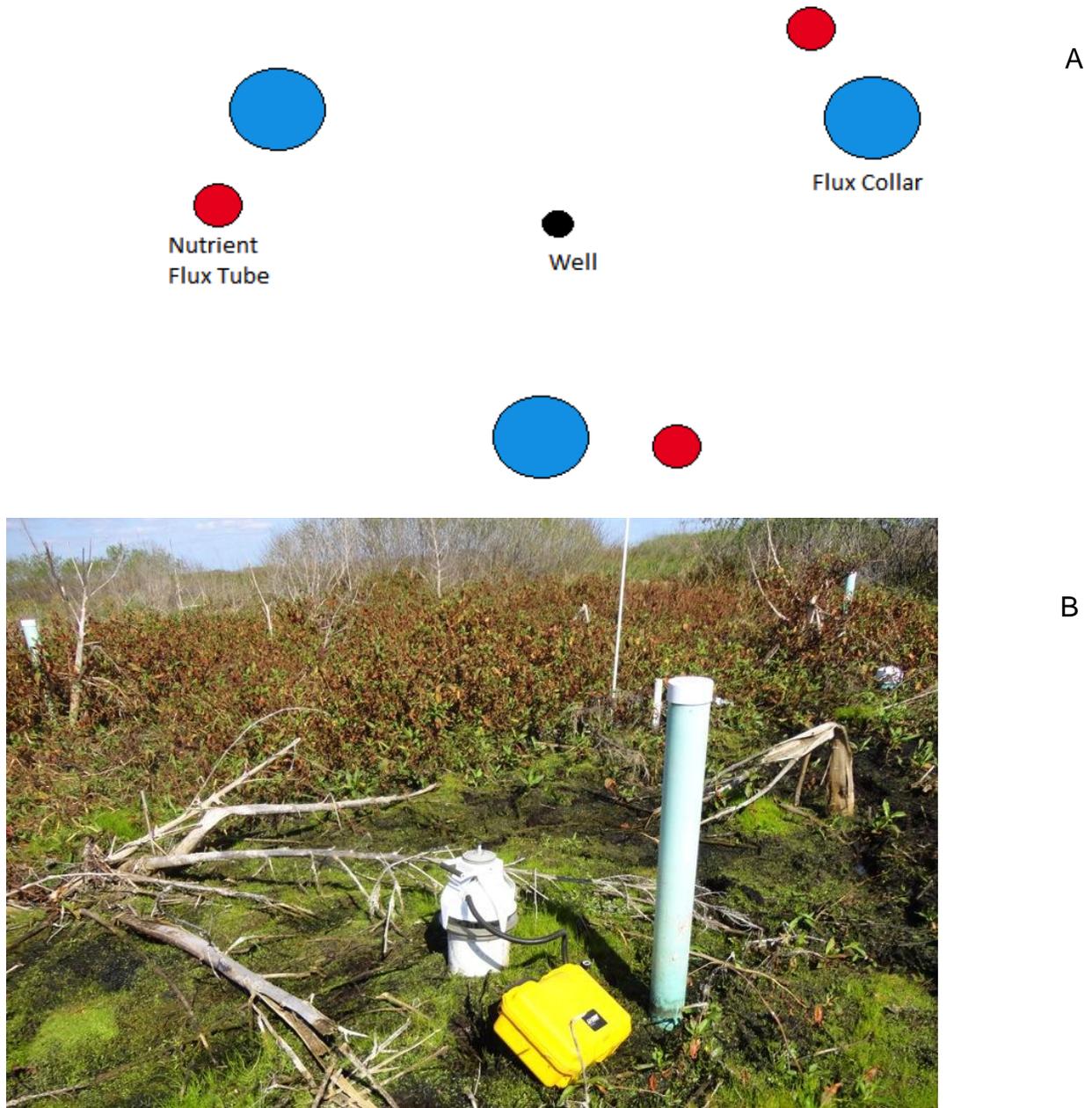


Figure 2-2. Diagram and picture of study site layout. A) Individual site diagram. B) Photograph of one site (SJM2-T3). Photo courtesy of Bryce Van Dam.

Monthly field measurements were conducted in SJMCA from March 2011 to September 2011. A Li-8100 soil respiration monitor equipped with a 20cm chamber was used to determine CO₂ flux. Table 2-1 shows the parameters that were used to run the Li-8100. CO₂ flux was measured at each site during each visit, along with water depth and soil temperature (11.7 cm Taylor digital temperature probe). For the first three months, three replicate CO₂ measurements were taken at each site, but in subsequent months, only 2 replicates were used because of time constraints. Every final CO₂ flux value is composed of two separate measurements, which were averaged together.

Table 2-1. Li-8100 parameters used for field and lab

	Obs Length (min)	Pre-Purge (min)	Post-Purge (min)	IRGA Averaging (sec)	Dead Band (min)	Chamber Diameter (cm)	Obs #
Field	1.5 – 1.75	0.5	0.5	4	0.5	10	2
Lab	1.5 - 2.0	0.5	0.5	4	0.5	20	2

As CH₄ sampling is a much more time consuming process, monitoring frequency was constrained to field access availability. To determine methane flux, a gas septum was installed on the Li-8100 (in line with tubing connecting the chamber and the analyzer) that allowed periodic gas sampling. A 20 cm Li-cor chamber was placed over the same PVC collars used for CO₂ measurement, and allowed to equilibrate for 30 minutes to 1 hr. During sampling in January, methane samples were collected using a chamber made from 20cm diameter PVC collars and end caps. Septa were installed to allow sampling, and small CPU cooling fans (the type commonly found in desktop computers) were used for air circulation. These fans were turned on for 5-10 seconds before each sample was taken to ensure complete mixing of the gas. Gas samples

were taken at pre-determined time intervals and transferred into evacuated gas vials. Analysis was completed on a GC equipped with an FID detector, and methane flux was calculated using a linear regression of concentration against time.

For this and all other studies, statistics were analyzed using JMP 8.0.2. All comparisons were made using one-way ANOVA or studentized t-tests at $\alpha=0.05$.

Lab Methods

To relate soil subsidence with corresponding rates of OM oxidation, our field work was supplemented with laboratory studies. The goal of these lab studies was to model how soils react to changing hydrology on shorter temporal scales. The data collected in these lab experiments were used along with the results from the field in order to create a more accurate model of soil subsidence. There were three laboratory studies conducted, and are termed “subsidence study”, “drain study”, and “water table study” respectively. The subsidence study involved a complete draining of soils in order to measure the potential rate of soil subsidence. In the drain experiment, soil cores were drained completely as in the subsidence study, but were monitored for CO₂ flux and soil moisture instead of subsidence. In the water table experiment, the goal was to determine the influence of water depth on carbon loss.

Subsidence Study

To determine total subsidence rates, 37 soil cores (10cm diameter) were taken from the SJMCA, and brought back to the laboratory. These cores were taken on March 16, 2011 when the SJMCA was dry at all field sites. 9 cores were taken from each of the 4 vegetation communities, and one from SJM3-CTR. These cores were all cut down to a common length of roughly 15 cm. A cap with three small holes was placed on the bottom of each core to facilitate wetting and draining. Flooding from bottom to top was

desired because we wanted to avoid moving nutrients down the soil profile, as well as prevent gas bubbles from being trapped in the soil matrix. The cores were then flooded with DI water for 9 days as part of a concurrent experiment, and subsequently were allowed to drain under gravity. Following draining, the depth of soil was measured 15 times over 148 days total. The ambient air temperature was regulated by an air conditioning unit, and was generally between 21-24°C. The location of these soil cores can be seen in Figure 3-1. A simple linear regression of soil depth and time was used to determine the total subsidence rate.

Water Table Study

To understand the link between hydrology and gaseous C losses, laboratory experiments were conducted in which the water table was manipulated to varying levels, and CO₂/CH₄ fluxes were measured. Additionally, an experiment was conducted wherein flooded cores were allowed to drain under gravity, and CO₂ flux was measured at pre-determined time intervals. For both experiments, 32 cores (10cm wide x 30cm deep) were taken from the SJMCA near site SJM1-T2. Caps with one small hole were placed on the bottom of each core, and the cores were put in 5 gallon buckets and moved to a South facing bench in a greenhouse. The cores were always covered with a black plastic tarp when not being sampled. 18 of the original 32 cores were randomly selected using a random number generator found at <<http://www.random.org>>. for each of 6 water level treatments (+1, -2.5, -7.5, -10, -15, and -20 cm). Each treatment contained three replicates, and the cores were all flooded with DI water as seen in Figure 2-3. Water was kept at the appropriate level by topping off the buckets containing the cores with DI water. The cores remained flooded for 22 days before measurement began to ensure that equilibrium was reached. To measure CO₂ flux, a Li-8100 with

10cm collar was used. As with the field measurements, CH₄ flux was determined by placing a septum in between the chamber and the analyzer to allow gas extraction. Gas samples were taken every 5-10 minutes for a total of 30 minutes and transferred into evacuated glass vials. Methane analysis was completed on a Shimadzu GC-FID, and flux was calculated using a linear regression. Soil temperature was also monitored in each core during flux measurement using a temperature probe.



Figure 2-3. Picture of water table experiment. Cores were randomly assigned to each of 6 water table treatments. Photo courtesy of Bryce Van Dam.

Drain Study

The other 18 cores were flooded to +2 cm with DI for 28 days (also placed in 5 gallon buckets on a South facing bench in a greenhouse), and were subsequently drained. CO₂ flux was measured 12 times over the following 90 days. CO₂ flux was measured using a Li-8100 equipped with a 10 cm collar. Measurements took place at varying times throughout the day in order to account for different temperatures that exist in the environment. Average daily air temperature was 31.5⁰C and ranged from 27.4 to 35.65 ⁰C, and the temperature during individual measurements ranged from 24.66 to

38.19 °C. Cores were periodically removed in triplicate, sectioned into 2cm samples, and analyzed for water content by drying at 105 °C for 24 hrs.

Results

In situ Study

Over the course of the study, CO₂ flux ranged from 106 to 7602 gC m⁻² yr⁻¹, and was 3036 ± 1712 gC m⁻² yr⁻¹ on average for all field sites. The water level was quite variable, ranging from a low of 116 cm below the soil surface during the summer to a high of 29 cm over the surface in the winter. Average surface soil temperature ranged from 14.2 to 27.7 °C (using an 11.7 cm probe), mirroring average air temperature for the same area. While CO₂ flux did not exhibit a strong linear relationship with water level (R²=0.28), flux measurements when flooded were significantly lower than when drained (Student's t-test, Alpha=0.05), as seen in Figure 2-4. When drained, CO₂ flux was 2595 ± 387 gC m⁻² yr⁻¹ (n=57), and decreased to 417 ± 78 gC m⁻² yr⁻¹ (n=12) when the water level rose above the soil surface. These standard errors for drained and flooded CO₂ flux of 387 and 78 gC m⁻² yr⁻¹ respectively indicate that aerobic decomposition is quite spatially and temporally variable.

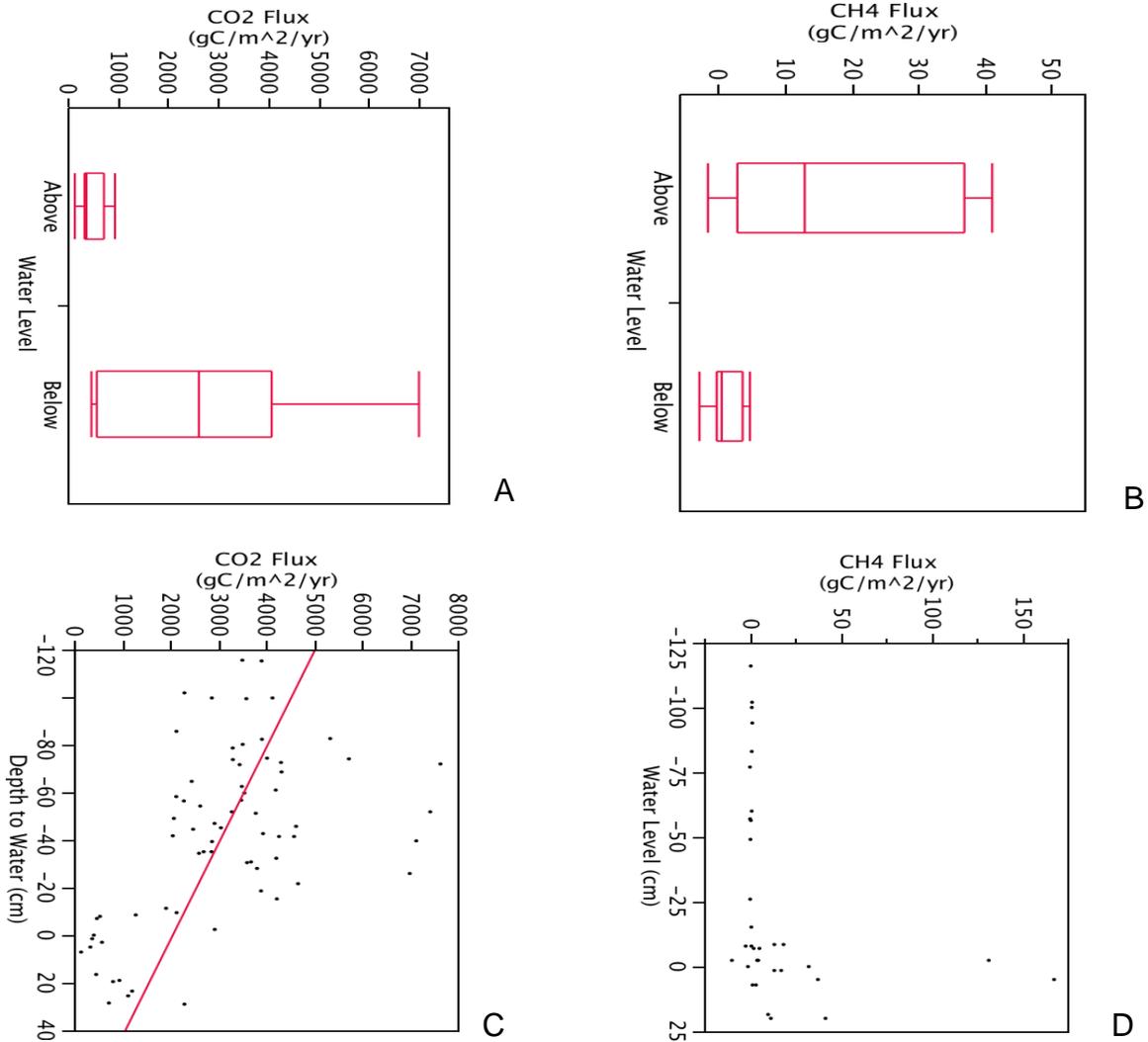


Figure 2-4. CO₂ and CH₄ fluxes with water depth. A) CO₂ flux (gC m⁻² yr⁻¹). B) CH₄ flux (gC m⁻² yr⁻¹), and flooding status. C) CO₂ Flux (gC m⁻² yr⁻¹) and depth to water (R² Adj = 0.27, n=69). D) CH₄ Flux (gC m⁻² yr⁻¹) and depth to water (n=33).

Methane flux contrasted with CO₂ flux in its relationship with water level. When drained, average CH₄ flux was 7.2 ± 6 gC m⁻² yr⁻¹ (n=22), and increased to 30 ± 14 gC m⁻² yr⁻¹ (n=11) when the water level rose above the soil surface. While flooding did result in a higher (and much more variable) CH₄ flux, a student's t-test ($\alpha=0.05$) showed that any difference was not significant.

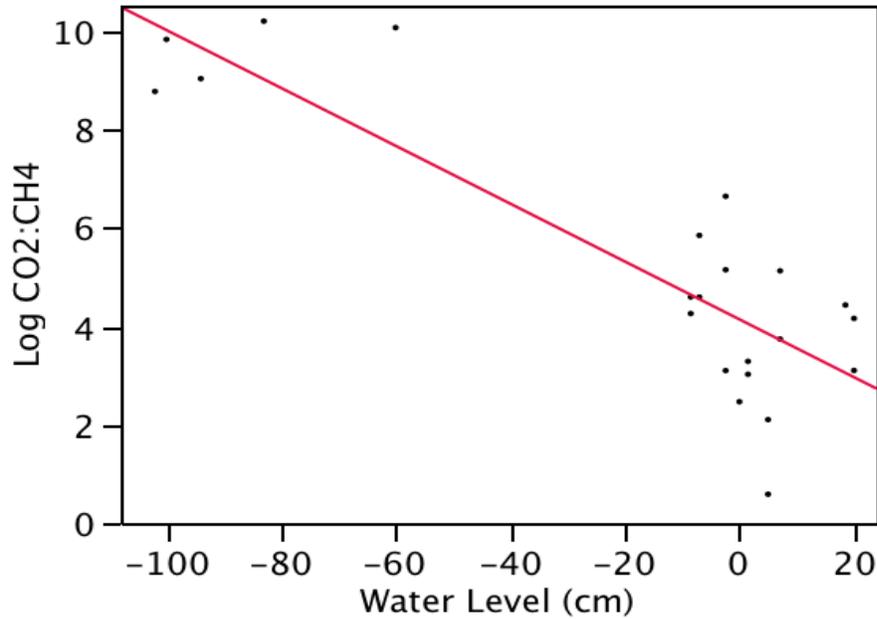


Figure 2-5. Log CO₂:CH₄ flux and Water Level (cm). n=22, R² Adj=0.72

Figure 2-5 shows that as the water level dropped, the ratio of CO₂ to CH₄ production rates increased logarithmically. While this relationship appears to be linear, extrapolation beyond water levels of -100 to +20 cm would likely not be accurate. As the water table rises over +20 cm, methane oxidation within the water column would cause CO₂:CH₄ to increase.

Total carbon respiration (CO₂ + CH₄) was $447 \pm 79 \text{ g C m}^{-2} \text{ yr}^{-1}$ when flooded, and increased to $2603 \pm 388 \text{ g C m}^{-2} \text{ yr}^{-1}$ when drained (Figure 2-6). According to a student's t-test ($\alpha=0.05$), these differences in total respiration are significant.

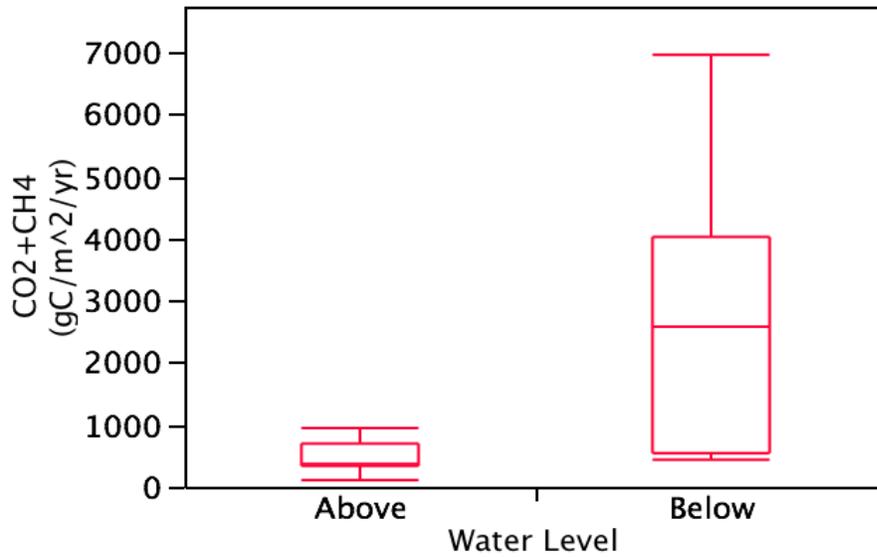


Figure 2-6. Total respiration ($\text{g C m}^{-2} \text{ yr}^{-1}$) and water level (cm)

Laboratory

Subsidence

Using a linear regression, total soil subsidence was found to occur at a rate of 4.7 cm yr^{-1} for the first 148 days after drainage, as shown in Figure 2-7. This rate accounts for a decrease in soil volume of at least 31% (each core was 15 cm long). Both primary and secondary subsidence influenced this rate.

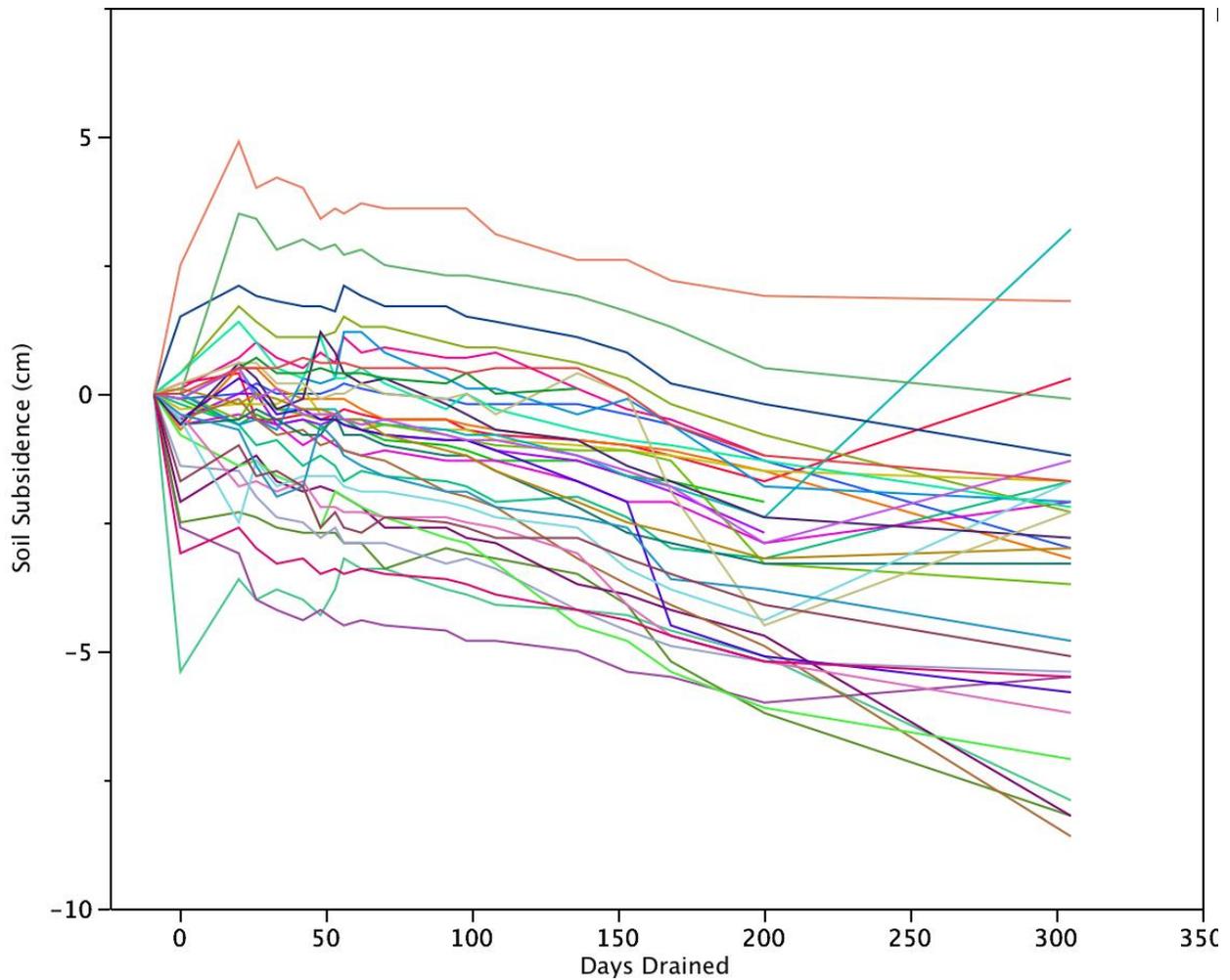


Figure 2-7. Average soil elevation (cm) and days drained (n=37)

Given field CO₂ fluxes of between 2066.2 and 3493.6 gC m⁻² yr⁻¹ (25 and 75% quartiles respectively), average bulk density of 0.173 g cm⁻³, and assuming the peat soil is 58% carbon, the relative contributions of primary and secondary subsidence can be calculated. Table 2-2 shows these findings. Secondary subsidence (oxidative losses) was found to account for between 44 and 83% of total subsidence.

Table 2-2. Estimates of subsidence fractions (percent of total in parenthesis). n=37

Factor	Low Estimate (cm yr-1)	High Estimate (cm yr-1)	Average (cm yr-1)
Primary Subsidence	0.78 (17%)	2.65 (56%)	1.72 (36.5%)
Secondary Subsidence	2.05 (44%)	3.92 (83%)	2.99 (63.5%)

It was noted that the peat pulled back from the sides of the cores over the study period, mirroring the peat cracking during the dry season in the field. In most cores, peat was observed floating upon flooding (for a concurrent study). Additionally, peat was seen floating in the SJMCA when it became flooding, indicating that similar processes are happening in the field as in the laboratory.

Water table study

In this study, CO₂ flux increased linearly from 441 to 923 g C m⁻² yr⁻¹ as the water table decreased from +1 to -20 cm (Figure 2-8). All CH₄ fluxes recorded when the water table was beneath the surface were essentially negligible, but increased to an average of 30 g C m⁻² yr⁻¹ when flooded.

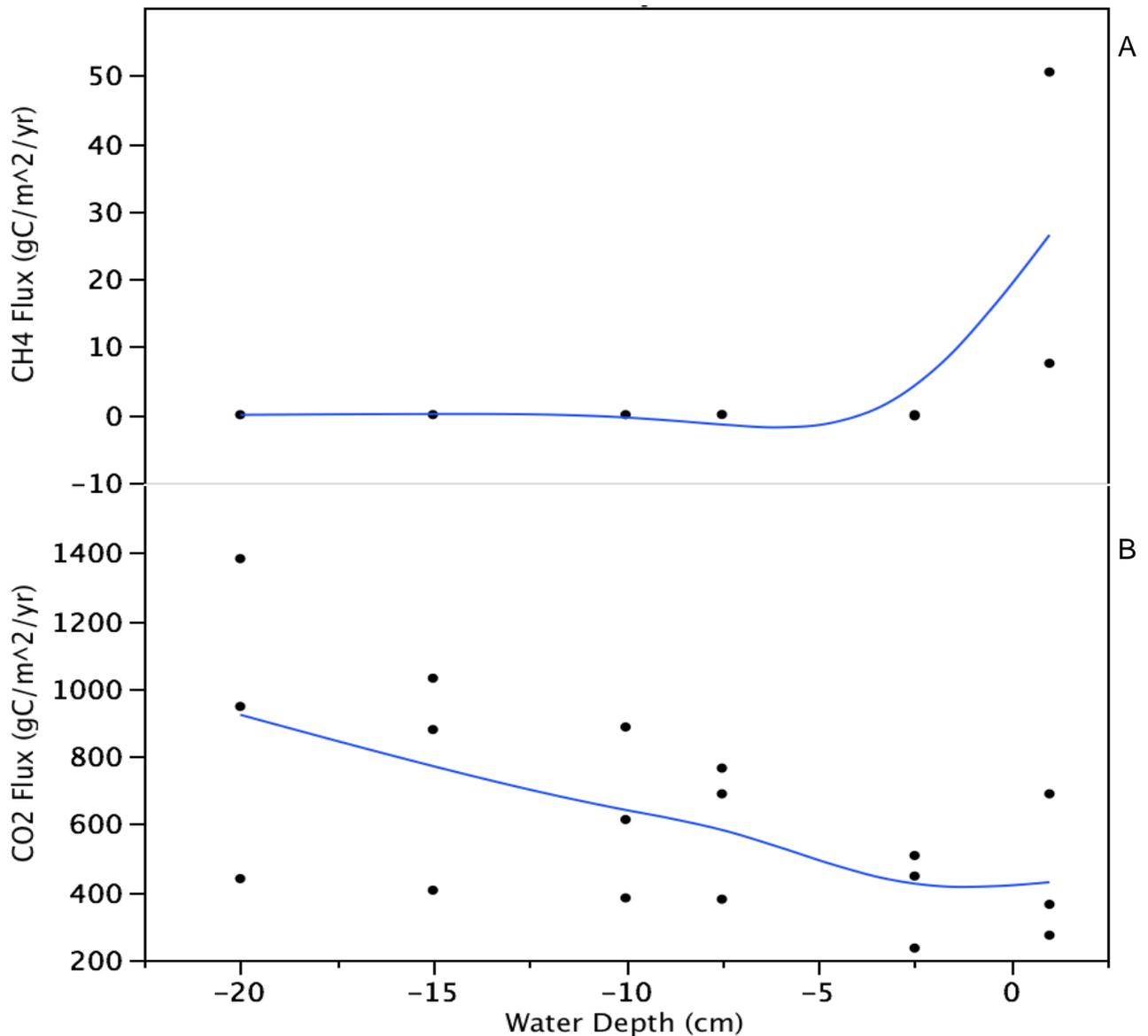


Figure 2-8. Laboratory CO₂ and CH₄ fluxes with water depth. A) CO₂ (g C m⁻² yr⁻¹) flux. B) CH₄ (g C m⁻² yr⁻¹) Fluxes at different water table treatments (cm).

A least squares method was used to fit models for the cumulative effect of soil temperature and water depth on both CO₂ and CH₄ fluxes. These models are shown in Table 2-3. While all soil cores showed an increase in CO₂ flux with increasing temperatures, treatments with lower water levels (drier surface soil) were more sensitive to temperature changes. Using the definition of Q₁₀ from Inglett et al. (2011)

$$Q_{10} = (k_2/k_1)^{10/(T_2-T_1)}$$

(where k₁ and k₂ are the CO₂ fluxes at lower and higher

temperatures respectively, and T_1 and T_2 are the low and high temperatures respectively), it was found that Q_{10} showed a negative relationship with water level (Figure 2-9). Q_{10} increased from 1.48 when the soil was flooded to 4.37 when drained to -20cm.

Table 2-3. Least squares models for CO_2 ($g\ C\ m^{-2}\ yr^{-1}$) and CH_4 ($g\ C\ m^{-2}\ yr^{-1}$) Fluxes. Adj R^2 of 0.77 and 0.13 respectively.

CO2	Parameter Estimate	Std Error	P > t	CH4	Parameter Estimate	Std Error	P > t
Intercept	-649	194	0.0043	Intercept	28.3	19.6	0.1834
Soil Temp (0C)	53.2	9.34	< 0.0001	Soil Temp (0C)	-0.806	1.01	0.4452
Water Depth (cm)	-22.8	4.92	0.0003	Water Depth (cm)	0.926	0.553	0.1823

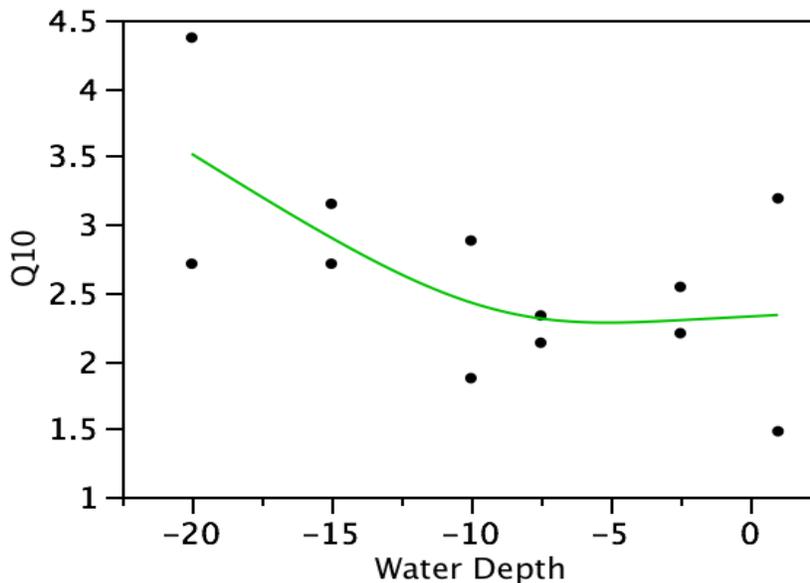


Figure 2-9. Average Q_{10} for CO_2 flux with varying water depths (cm). Smoothing Spline Fit, $\lambda=100$, $R^2= 0.42$

Drain study

After an unexpected initial CO_2 Flux as high as $50757\ g\ C\ m^{-2}\ yr^{-1}$ during the first day after draining, a steady rate of between 600 and $1200\ g\ C\ m^{-2}\ yr^{-1}$ was observed

for the remainder of the study. Results of this experiment are shown graphically in Figure 2-10.

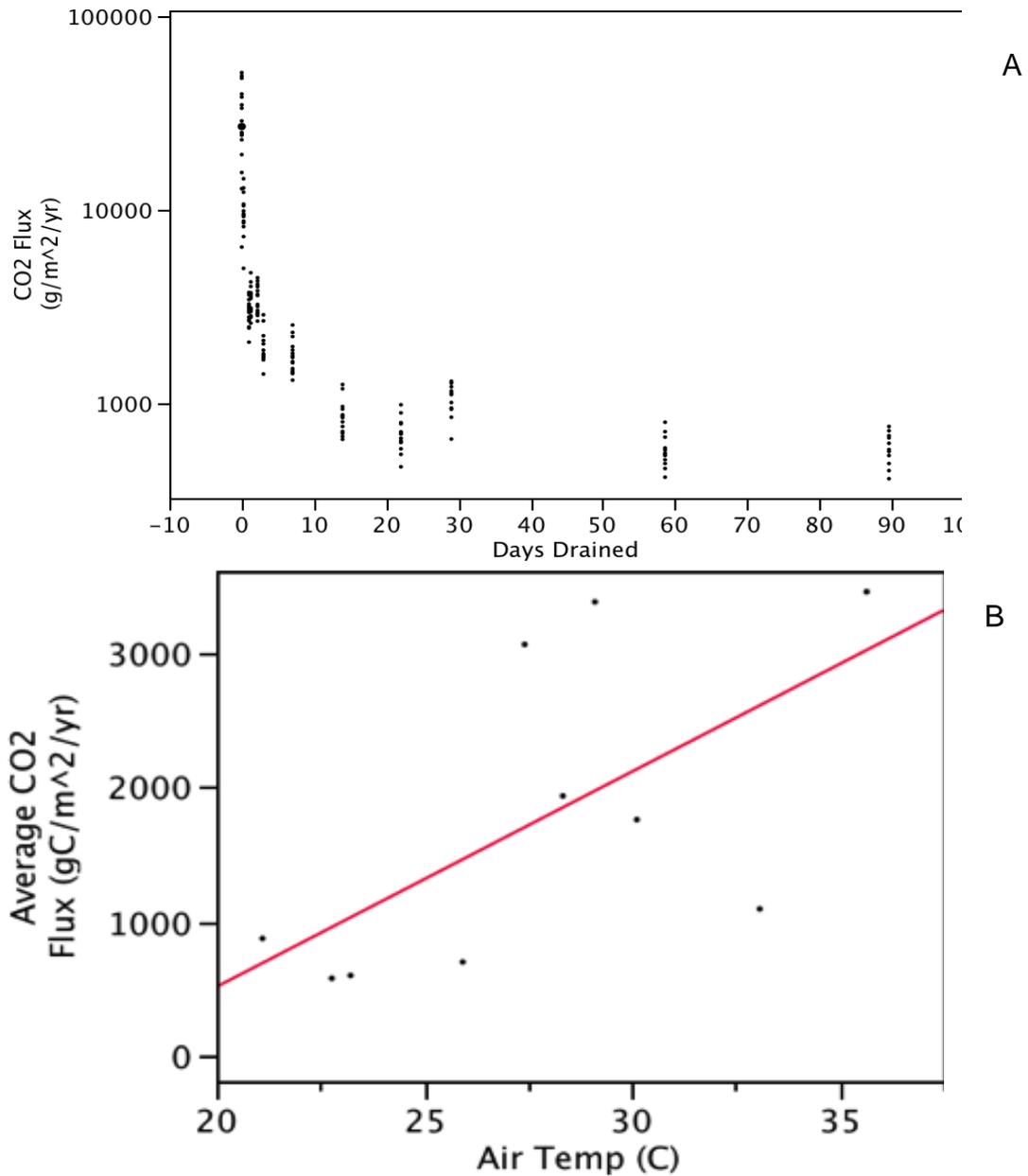


Figure 2-10. CO₂ flux after draining, and with air temperature. A) CO₂ flux (g C m⁻² yr⁻¹) after draining. B) Air temperature (°C) and CO₂ Flux (g C m⁻² yr⁻¹), Adj R²=0.32, n=18

Because these cores were incubated in a greenhouse, ambient air temperature was quite variable (ranging from 21.11 to 35.65 °C). When potential outlier CO₂ fluxes

were removed, air temperature accounted for 32% of the variability between measurements (using a linear regression).

While this experiment was designed to study the effect of changing soil moisture on CO₂ flux, average soil moisture within the cores did not change between these sampling dates. The same held true when only surface soils (0-8cm) were considered. Soil cores were sampled at 22, 59, and 90 days for soil moisture (Figure 2-11).

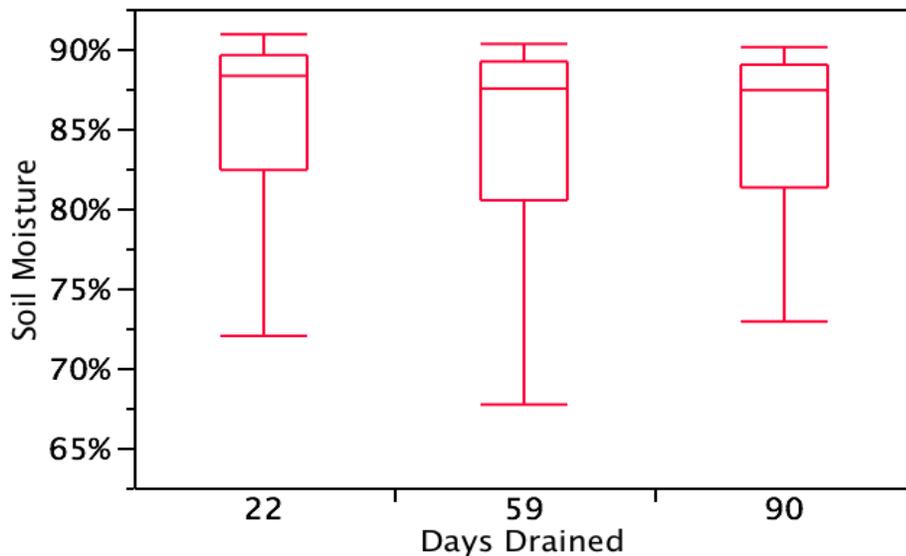


Figure 2-11. Soil moisture (%) after draining. P-value = 0.7128, n=3

Discussion

Insitu Study

When the water table was below the surface of the marsh, CO₂ flux became high and quite variable. Some of this variability in drained CO₂ flux could be explained by the fluctuating nature of soil temperature during the summer months. A study by Inglett et al. (2011) showed that temperature, vegetation, and their interaction was a significant factor in determining greenhouse gas flux (aerobic and anaerobic) for soils in the same general region in Florida. The current study also demonstrates the importance of

temperature on aerobic respiration; Q_{10} values of over 4 were found for potential CO_2 flux. It is also interesting to note that according to a studentized t-test ($\alpha=0.05$), average water level did not differ significantly between any pair of field sites. Because the soil elevation does differ between sites, this lack of difference in water levels indicates either inaccuracies in measurement, or that the potentiometric surfaces at these sites mirrors their respective soil elevations. This lack of difference in average water levels was seen despite the fact that the transects were set up for the express purpose of studying the effect of differing water levels on CO_2 flux.

Wetlands are generally very productive ecosystems, and can have significant impacts on the global carbon cycle. While under normal hydrologic conditions, wetlands are considered to be net carbon sinks, however, they can become sources when drained (Schedlbauer et al. 2010; Kechavarzi et al. 2010; Jauhiainen et al. 2008; Armentano et al. 1986). Primary productivity in wetlands has been estimated to be between $1300 - 3200 \text{ g C m}^{-2} \text{ yr}^{-1}$ (Miller and Fujii 2009). Therefore, using the CO_2 and CH_4 fluxes found in this study, the SJMCA would undergo rapid oxidative soil loss (roughly $6-8 \text{ kg C m}^{-2} \text{ yr}^{-1}$) if drained, but would be in approximate equilibrium while flooded. This relates to an average net flux of carbon from the SJMCA to the atmosphere of $0.97 \text{ Tg C yr}^{-1}$ under drained conditions during the study period (not a full year).

Over the course of the study, methane contributed a relatively small amount to overall carbon flux ($7.2 - 30 \text{ gC m}^{-2} \text{ yr}^{-1}$). Lowest methane fluxes were seen when the water table was below the surface, while highest fluxes occurred when the wetland was flooded (Figure 2-4). While drained CH_4 fluxes were consistently low, flooded CH_4 flux

was quite variable. Although factors like temperature and nutrient status certainly contributed to some of this variability, measurement methods are also of concern. When CO₂ flux is measured, the observer only needs to approach the chamber once to initiate the experiment, then does not return until the measurement is completed. As methane flux was measured using gas samples taken at 5-10 minute intervals, the observer must approach the chamber a number of times, possibly disturbing the soil, releasing excess methane to the chamber. Care was taken to ensure this source of error was kept to a minimum, but some level of disturbance was inevitable. Given that the SJMCA was only flooded for about 25% of the year during this study, CO₂ flux should be considered the main gaseous loss of carbon from this wetland.

Methane fluxes found in this study represent the difference between the processes of methanogenesis (production) and methane oxidation within the soil-plant-air continuum. Methanogenesis is the result of a series of microbially mediated reactions of which the final step is the actual production of methane. Methane oxidation is also a microbial process that occurs at the aerobic-anaerobic interface within the soil, often oxidized portions of the rhizosphere (Segers 1998; Altor and Mitsch 2008). Thus, the gross methane flux in the SJMCA is almost certainly much higher than the net fluxes shown here. A 1990 study of Everglades peat soils showed that 91% of the methane produced in soils is oxidized before it can be lost to the atmosphere. This oxidation occurs in the aerobic portion of the soil/floodwater system (King et al. 1990).

As seen in Figure 2-4 (c,d), the inflection point for both CO₂ and CH₄ fluxes did not line up exactly with a water depth of 0 cm as expected. CO₂ flux began to decrease and CH₄ flux began to increase as the water table reached about -10 cm. Therefore, even

though the potentiometric water level was well below the soil surface, capillarity action was sufficient to generate anaerobic conditions at the soil surface. This information could be very useful in managing the SJMCA, it indicates that soil loss could be kept to a minimum while still drawing the water table to 10 cm below the soil surface. This water table drawdown would allow the wetland to approximately store an additional 1.4×10^7 m³ of water, which is vital given the importance of the flood control mission of the wetland.

Laboratory

Subsidence study

Total subsidence was found to occur at a potential rate of 4.7 cm yr^{-1} . This rate includes primary and secondary subsidence, which were calculated to account for 36.5 and 63.5% of total subsidence respectively. A study by Vepraskas and Ewing (2006) found primary and secondary subsidence to account for 69 and 31% of total subsidence respectively. Their study compared subsidence rates between soils drained for 15, 20, and 30 years. As time drained increased, primary subsidence became more influential relative to secondary subsidence. Therefore, as the soils in the current study were drained for less than a year, it would follow that secondary subsidence would be the main cause of soil elevation change, which was the case. Nevertheless, the SJMCA has been chronically overdrained since the mid 20th century. If this drainage time is taken into account, it would then be expected that primary subsidence would dominate relative to secondary subsidence based on previous studies. The findings in this study could thus be interpreted as being somewhat unconventional in that secondary subsidence was higher than primary subsidence for soil that has been hydrologically altered for a significant period of time.

As was seen in the water table study (and in many previous studies (Inglett et al. 2011; Kirschbaum 2006; Chapman et al. 1996)), temperature is an important factor in determining the rate of organic matter oxidation. Thus, any differences in temperature could introduce a large source of error into this subsidence model. The average air temperature in the study area is 22.4 °C, and the air temperature in the laboratory was between 21 and 24 °C. This is a maximum difference in temperature between field and lab of 1.6 °C. A study of BCM soils (Inglett et al. 2011) found Q_{10} values to be in the range of 1.4 to 2.1. If Q_{10} is assumed to be the mean of these values, 1.8, CO_2 flux would only be a maximum of 11% different between the field and lab. While temperature differences are only a source of minimal error, there are certainly other factors that could cause the potential subsidence rate to differ from the actual rate, including vegetation, fauna, aeolian soil loss, fire, etc. Although these factors cannot be accurately accounted for using this subsidence model, it should still be relatively accurate in predicting soil elevation change over time.

Water table study

Figure 2-9 shows CO_2 flux increased linearly from 441 to 923 g C m⁻² yr⁻¹ as the water table decreased from +1 to -20 cm. All CH_4 fluxes while drained were essentially negligible, but increased to 30 g C m⁻² yr⁻¹ when flooded. A least squares model showed that water depth and temperature were both significant predictors of CO_2 flux, but not of CH_4 flux. The inclusion of water temperature increased the model reliability markedly for CO_2 flux, but decreased reliability slightly for CH_4 flux (Adj R^2 increase of + 148% and - 19% respectively). This is consistent with literature reports that aerobic decomposition is more temperature sensitive than anaerobic decomposition (Chapman et al. 1996).

An interesting finding of this study was that the temperature sensitivity (Q_{10}) was greater when the water table was low, and decreased when flooded. Q_{10} values were between 1.48 and 4.37. Values in the range of 1.5 to 3.5 are typical for soils in this region according to previous studies by Inglett et al. (2011). Although determining temperature sensitivity was not part of this study, this finding offers valuable insight into the factors effecting soil oxidation in the SJMCA. This finding indicates that the season in which the water table is drawn down is very important. During winter months, the depth of water in the wetland should have a small influence on the CO_2 flux, while during summer months, the flooding status will be critical. Another possible explanation for this observed relationship is that increasing recalcitrance of OM is related to greater temperature sensitivities (Conant et al. 2008). Thus, as time progresses after draining, the peat becomes more recalcitrant, and possibly more sensitive to temperature changes.

Drain study

After draining, CO_2 flux was initially very high, but quickly decreased to a steady rate of between 600 and 1200 g C m⁻² yr⁻¹ within the first 10 days. These anomalously high initial CO_2 fluxes are most likely a result of ebullition of gas bubbles from the recently drained soil. While the soil cores were flooded, gas accumulated in pore spaces, so when the cores were drained, this gas was released.

CO_2 flux was expected to increase to a maximum while soils drained to optimal levels, then to decrease as the soil became too dry for respiration. This trend was not seen, possibly because soils never dried out fully. Figure 2-12 shows that the water content of the soil cores did not change significantly between days 22, 59, and 90 after draining. As described in the methods, a black plastic tarp was draped over the cores

during this experiment to inhibit the growth of plants. This tarp may have had the unintended effect of also limiting water vapor transport from the cores to the atmosphere.

Concluding Remarks

The objective of this research was to quantify the effect of current hydrologic management schemes on both primary and secondary subsidence. This study indicated that under current management practices, soil subsidence will continue to be a factor in the SJMCA. While drained, the potential subsidence rate was 4.7 cm yr^{-1} . When the soils are allowed to drain for approximately 1 month per year or more, total subsidence (primary and secondary) will exceed sediment accumulation, causing net soil loss. Furthermore, the finding that secondary subsidence exceeded primary subsidence is somewhat novel, and indicates that the legacy of drainage in the SJMCA might be different than in other wetlands with similar hydrology.

While vegetative type may not be particularly critical in determining CO_2 flux rates, temperature and water depth are key to predicting carbon mobility. During both field and laboratory studies, hydrology exerted a strong influence on CO_2 flux; rates were low when the soil was flooded, and high and variable when the water table was below the surface. CO_2 Flux when drained was roughly 6 times higher than when flooded. Similarly, CH_4 flux showed a strong trend with water level. When drained, flux was near zero ($7.2 \pm 6 \text{ g C m}^{-2} \text{ yr}^{-1}$), but became high and variable when flooded ($30 \pm 14 \text{ g C m}^{-2} \text{ yr}^{-1}$). Following draining, CO_2 flux was initially high and variable, but quickly (within 10 days) approached a steady rate of between 600 and $1200 \text{ g C m}^{-2} \text{ yr}^{-1}$.

Overall, hydrologic management appears to have a strong influence on carbon dynamics in the SJMCA. Because wetlands are characterized by intermediate water

table conditions (water level at or near the surface), small changes in hydrology can have significant effects on carbon cycling. As current hydrologic management results in a water table drawdown during the summer months, when temperature is high, oxidative carbon loss will continue to be a problem in the SJMCA.

CHAPTER 3 NUTRIENT EXCHANGE IN THE SJMCA

Opening Statement

Wetlands can effectively function as sinks or sources of nutrients to downstream water bodies. The nutrients of most concern in Florida are carbon (C), nitrogen (N), and phosphorus (P). When wetlands are flooded, the diffusion of oxygen into the soil is greatly diminished, restricting the ability of aerobic microbes to compete for organic matter. When oxygen is not available, other electron acceptors (NO_3^- , Fe^{3+} , SO_4^{2-} etc) are used by anaerobic microbes to break down organic carbon (Mclatchey and Reddy 1998; White and Reddy 1997). In the case of nitrogen, when NO_3^- (aq) is reduced during the oxidation of organic matter, gaseous species are formed (N_2O and N_2). This process is termed denitrification, and can result in a permanent removal of nitrogen from the wetland. In freshwater systems, denitrification can lead to a nitrogen flux from the wetland to the atmosphere of up to $0.5\text{g N m}^{-2}\text{ day}^{-1}$ (Knowles 1982; Bowden 1987; Hansen et al. 1993). If not denitrified, this nitrogen would be transferred downstream where it could contribute to water quality degradation.

In addition to providing this biogeochemical sink for nitrogen, wetlands may also retain phosphorus. Under aerobic conditions plant litter can be fully decomposed, allowing all organic phosphorus to be mineralized to PO_4^{3-} . Wetlands, on the other hand, are characterized by anaerobic conditions where SOM is not fully mineralized. In this case, phosphorus within the SOM is protected to a greater extent from microbial activity. Therefore, wetlands can serve as a sink for phosphorus, sequestering this nutrient along with carbon in thick layers of peat (Mclatchey and Reddy 1998, Olila et al. 1997).

While under flooded conditions, wetlands can restrict the flow of nutrients from upstream lands to downstream water bodies, upon draining, this mechanism is lost. When wetlands are drained, oxygen can penetrate into soil organic matter that is highly reduced. This has profound effects on the fate of soil C, N, and P. While flooded, the major end products of decomposition are CH₄ and DOC. When drained, porewater DOC that accumulated during anaerobiosis is rapidly mineralized to CO₂, and lost to the atmosphere (Ellis et al. 2009; Freeman et al. 1996; Leiros et al. 1999). Ellis et al. (2009) found that a water table drawdown of 12 weeks resulted in a 12.6% decrease in porewater DOC. This aerobic DOC mineralization brings a concomitant flux of P and N. A 1996 study by Freeman et al. demonstrated that water table drawdown increased phosphatase (enzyme responsible for hydrolysing P containing substrates) activity by 31-67%. This increase in phosphatase activity can result in a large flux of PO₄²⁻ into the water column upon re-flooding. As no common gaseous form of phosphorus exists, the P mineralized during draining remains in place within the OM until reflooding. Olila et al. (1997) found that draining lake Apopka sediment for 6 weeks resulted in 334 mg P m⁻² d⁻¹ being released to the water column.

Similar to phosphorus, NH₄⁺ (ammonium) that was accumulated in the soil when flooded can be nitrified (microbially oxidized to NO₃⁻) when drained. A study by Malecki et al. in 2004 showed that sediments in the Lower St Johns River (LSJR) fluxed significantly more ammonium into the water column when the soil was anaerobic than when aerobic. This ammonium is oxidized to nitrate through the process of nitrification. Under flooded conditions, this nitrate would be rapidly used by denitrifying microbes,

and released as N₂ gas. Nitrate remains in the soil when drained, and is lost to the water column when re-flooded (Bowden 1987).

While nitrogen and phosphorus are necessary nutrients in all biological systems, they can lead to imbalances in water chemistry when in excess concentrations (eutrophication). This nutrient imbalance can cause the proliferation of nuisance species of algae and macrophytes (Maher 1997; Malecki et al. 2004, Lorenzen 2001). These nutrients can come from point and non-point sources, both of which are of significant concern in the St Johns River watershed. According to a report by Maher in 1997, the loads of nitrogen and phosphorus in the lower St Johns River are dominated by contributions from upstream flow. As this upstream component of nutrients originates in the marshes in and around the SJMCA, it is vital to understand how these nutrients move through the wetland and how water management may be utilized to reduce nutrient loading (deterring subsidence of organic soils).

In many oligotrophic aquatic systems phosphorus is a limiting nutrient, and if added in excess of a critical amount, can lead to the development of HABs or crashes in dissolved oxygen. While phosphorus can be assimilated into wetland soils effectively when flooded, it is rapidly mineralized and can be lost to the water column when the soil is drained and subsequently flooded (Olila et al. 1997). As the current management practices result in severe water table fluctuations (flooded in the winter and drained in the summer), yearly phosphorus fluxes during the summer months are frequent. Interestingly, these P-fluxes from the USJRB can initiate a positive feedback cycle in the lower SJR. According to a study by Malecki et al. (2004), algal blooms caused by nutrient loading can cause decreases in porewater O₂, and subsequent anoxia. This

anoxia increased sediment to water P-flux by a factor of 40, further enhancing eutrophication.

According to a ^{210}Pb nutrient accumulation study by Brenner et al. (2001), the USJRB has experienced increased P-loading over the last 70 years. This phosphorus loading in conjunction with seasonal water table fluctuations has created a situation where DO crashes, fish kills, and harmful algal blooms (HABs) are a common occurrence on the lower St Johns River, a water body that is critical to Florida's economic and environmental well being. Clearly, more needs to be known about the effect of water table dynamics on nutrient biogeochemistry so the wetlands of the USJRB can be best managed to prevent floods without causing water quality degradation.

Nutrient mineralization and subsequent eutrophication of downstream water bodies is a serious problem in the SJR watershed. As such, it is vital to know how major nutrient biogeochemistry (cycling of nitrogen, phosphorus, and sulphur) is affected by soil drying and rewetting under altered hydrologic conditions. The objective of this research is to determine how various water management schemes affect nutrient losses from the SJMCA. Potential nutrient mineralization rates will be determined using both laboratory and field studies.

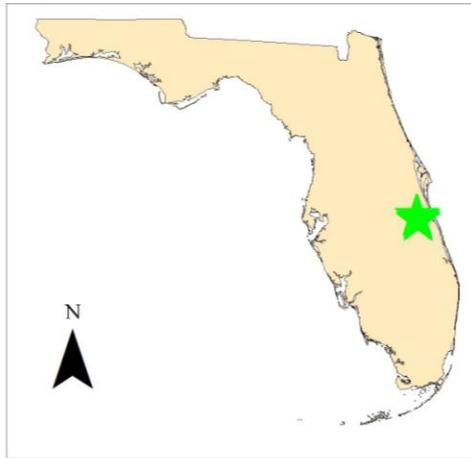
Methods

Site Description

All of the field sites for this study are located within the SJMCA (Figure 3-1). The location of these sites were chosen to maximize elevation and vegetation gradients. Two transects of approximately 0.5 km each have been established along the main C-40 canal. Each Transect consists of three sites: one close to the canal, one at a

moderate distance, and one far away from the canal, with the objective of creating a profile of wet to dry across the wetland. Four more sites were set up on the north side of the marsh where multiple vegetative communities existed in close proximity. The 4 vegetation communities chosen were: Willow (*Salix caroliniana*), Sawgrass (*Cladium jamaicense*), Cattail (*Typha spp.*), and slough. One last site (SJM3-CTR) was created on the western side of the SJMCA where soil was characteristically dry. These sites can be seen in Figure 3-2.

St Johns Marsh Conservation Area



SJMCA Field Sites



Figure 3-1. Location of study site.

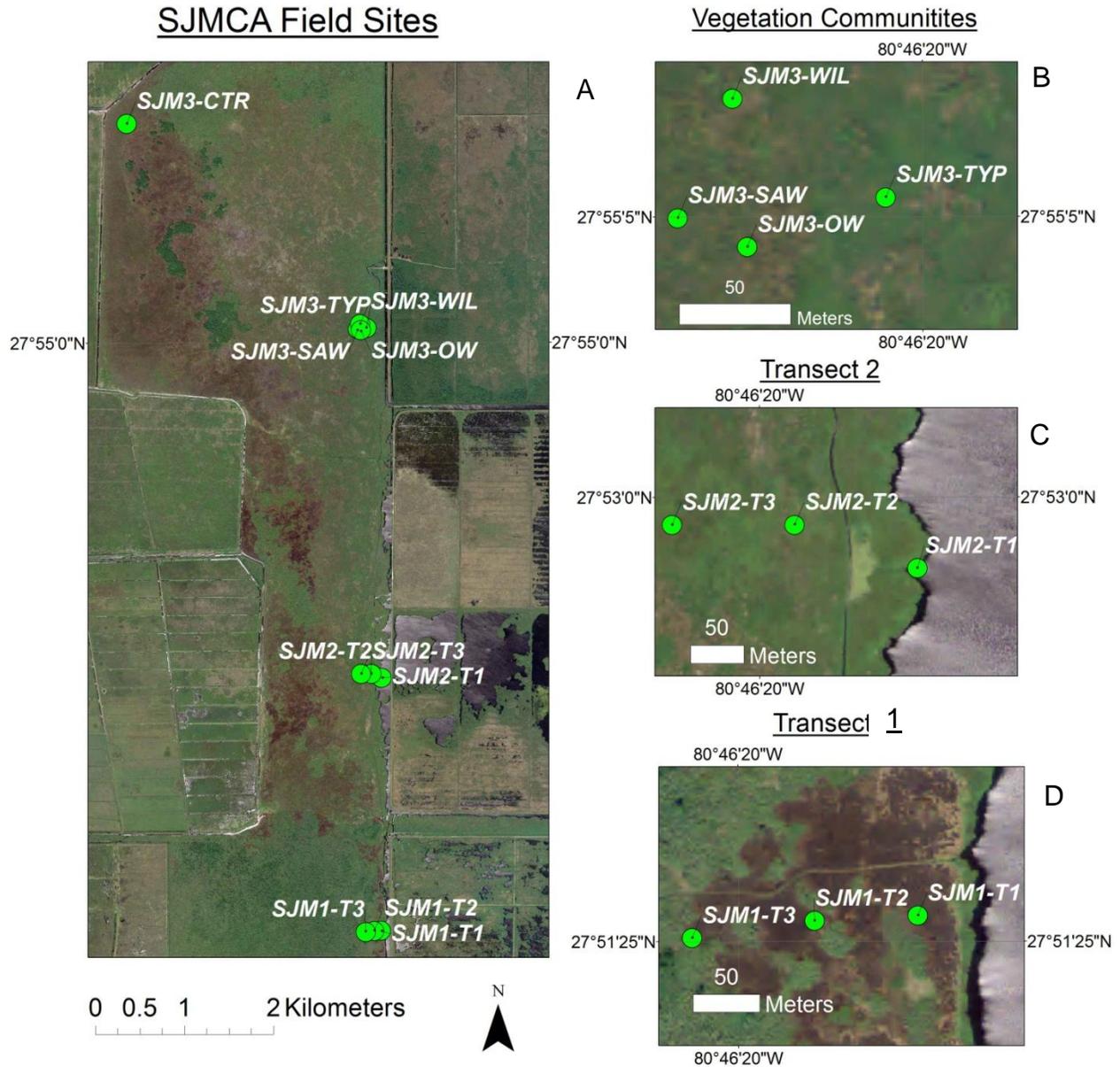


Figure 3-2. Map of field sites. A) Overall field sites in the SJMCA. B) Vegetation communities. C) Transect 2. D) Transect 1.

Thus, a total of 11 sites were established in the SJMCA. At each site, 20 cm diameter PVC collars and 10cm diameter PVC pipes (1-1.5 m tall) were installed in triplicate, as well as a single shallow (~1.5m) monitoring well. Each well was made by drilling a series of holes into a 5 cm diameter, 1-1.5 m long PVC pipe. At the center of each field site, an auger just under 5 cm in diameter was used to excavate soil to allow

installation of the wells. Depth to water in the wells was measured using a tape measurer equipped with a water depth sounder. A diagram and a picture of a field site can be seen in Figure 3-3

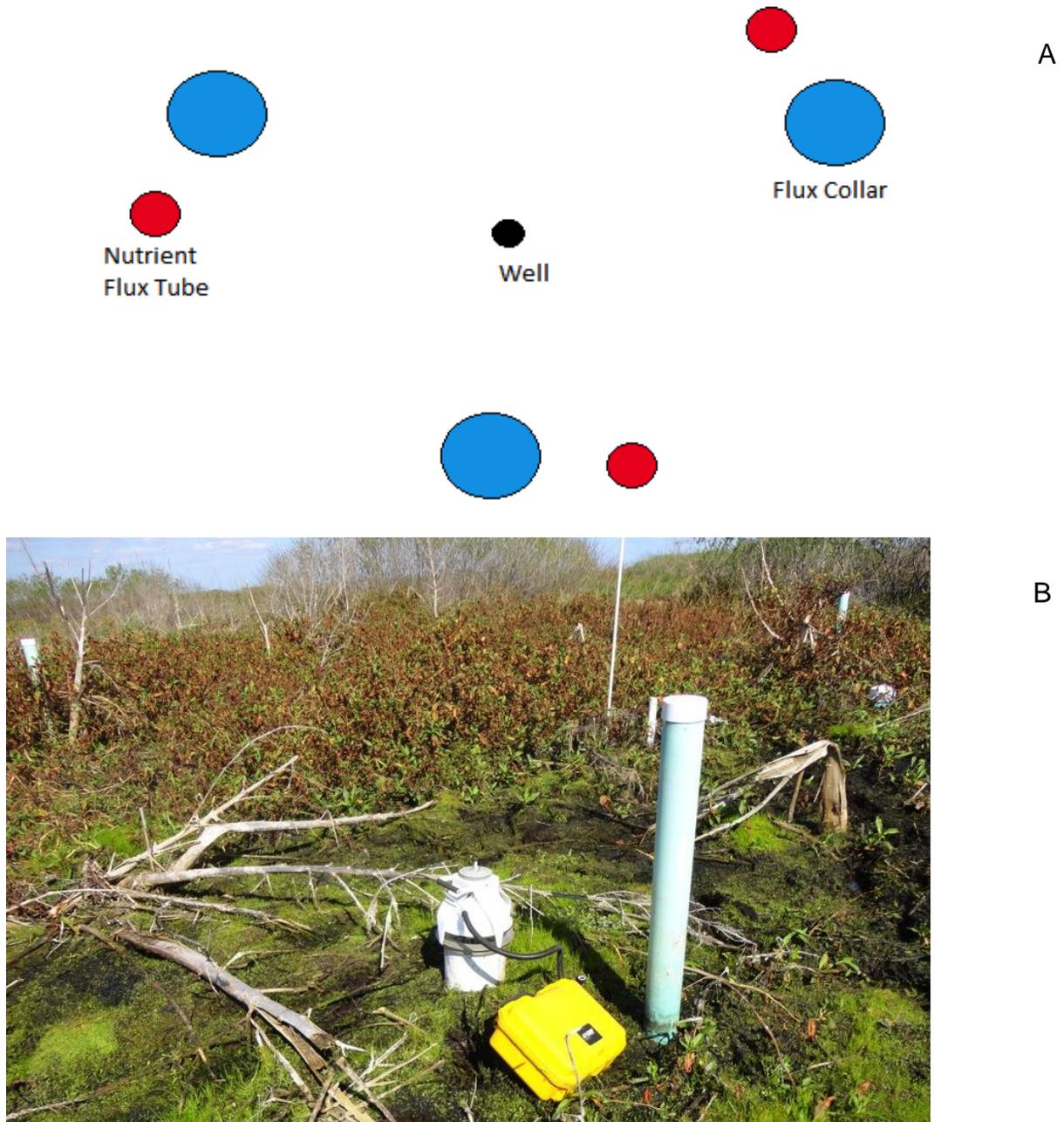


Figure 3-3. Diagram and picture of field site layout. A) Individual site diagram. B) Photograph of one site (SJM2-T3). Central well, nutrient flux pipes, and gas flux collars can be seen. Photo courtesy of Bryce Van Dam.

Soil Chemistry

Soil cores taken from the SJMCA were sectioned by depth and analyzed for total C, N, P, and S in this study. One core (5 cm diameter by 20-30 cm in length) was taken from each of the 11 field sites described in the site description section. Due to soil compaction, it was not possible to fill some cores to the top. These cores were extruded by hand and cut into 2 cm segments. This soil was then dried at 105 °C for 24 hrs, sieved, then ball milled for 15 minutes prior to analysis. Total carbon and nitrogen were determined using a Leco Tru-Spec combustion analyzer. Total sulfur was determined by HCl extraction/ICP (Thermo ICP6500). TP was determined by atomic emission on a PerkinElmer 5300DV (EPA 200.7). For this and all other studies, statistics were analyzed using JMP 8.0.2. All comparisons were made using one-way ANOVA or studentized t-test at $\alpha=0.05$.

Laboratory Nutrient Flux

To reduce variability often found in field flux studies and look at more controlled water level effects, 37 soil cores were taken from SJMCA on 16 March 2011 (9 cores from each of 4 vegetation communities) and brought back to the laboratory where they were placed into buckets and flooded to 10cm over the soil surface with DI water. The soil at this region had been drained for approximately 3 months prior to removing these cores. All of these cores were cut down to 15 cm in length to ensure the soil volume for each sample was the same. While site water could have been used to flood the cores, the use of DI water offered 2 distinct advantages for this study. DI water is more representative of rain water, and reduces a significant source of excess nutrients that could cause error in nutrient flux calculations. The 4 vegetation communities sampled were: Willow (*Salix caroliniana*), Sawgrass (*Cladium jamaicense*), Cattail (*Typha spp.*),

and open water. One last site (SJM3-CTR) on the western side of the SJMCA was used where the soil was chronically overdrained. Before flooding, 3 small holes were drilled into PVC caps placed on the bottom of the cores. These holes allowed water to enter the soil from the bottom, so as to not move nutrients down the soil profile. These cores were kept in a dark tub in an indoor laboratory (seen in Figure 3-4). A plywood cover was placed over the cores to prevent light entry. The ambient air temperature was regulated by an air conditioning unit, and was generally between 21-24°C. After 9 days, water standing above the soil was sampled, filtered (0.45 µm), acidified (to a pH of 2.0 with H₂SO₄), and analyzed for DOC, TDP, and TDKN. Sampling was accomplished by lowering a small tube half way between the water surface and the soil, and pulling water into a 60µL syringe. Standing water was not mixed prior to sampling as to prevent soil material from contaminating water samples. DOC was found using the high temperature combustion method with a Shimadzu TOC5050A CE (SM5310B).



Figure 3-4. Soil cores in laboratory. Container is insulated fiberglass; a plywood lid was placed over the bath when not being sampled to prevent light from entering. Soil cores were placed randomly into each of 13-5 gal buckets. Cores were flooded by pouring DI water into buckets outside of the cores.

TDP was determined spectrophotometrically (Bran and Lubbe AA3 auto-analyzer) using the ascorbic acid method (SM4500P F). TDKN was determined using a sulfuric acid digestion, with analysis on a Technicon AA2 auto-analyzer (O'Dell 1993) (EPA351.2). After sampling, the cores were allowed to drain under gravity for 215 days. Following this period, the cores were again flooded with DI water to an average of 13cm above the

soil surface (total subsidence during the 215 days was 3 cm on average). After 9 days, surface water samples were again taken, filtered (0.45um), acidified (H₂SO₄), and analyzed for DOC, TDP, and TDKN as before (TDKN was determined on an Astoria Pacific A2 segmented flow analyzer). Nutrient flux was calculated as in the field methods.

Field Nutrient Flux

PVC pipes were installed in triplicate at each of the 11 field sites. These pipes were 10 cm in diameter and were sunk about 30 cm into the soil (Figure 3-3). The 10 cm diameter PVC pipes had one small hole drilled in the side to allow pressure equilibration upon flooding. Additionally, a loosely fitted PVC cap was placed on the top to prevent contamination by rainwater, insects, and birds from entering the nutrient flux tube. The purpose of these pipes was to allow sampling of soil pore water without contamination from surface or rain water. Once the water table reached the surface, pore water fluxed from the soil into the inside the columns was removed from the tubes using a peristaltic pump. It could not be determined exactly when the field sites became flooded, but effort was made to sample as soon as possible after flooding. Water was not mixed within the tube in order to prevent soil from contaminating water samples. Water samples were analyzed by the St Johns River Water Management District Laboratory for DO, DOC, NH₄, TP, TDKN, Turbidity, and pH (along with other analytes that are not immediately relevant to this study). Nutrient flux was calculated as g m⁻² using the pipe diameter and depth of water at each location.

Results

Soil Chemistry

Results from soil chemical analysis are shown in full in Figure A-2 in the Appendix, and are summarized in Table 3-1. Soil total carbon increased with depth from 40.0 ± 1.8 to $51.9 \pm 1.0\%$. Total phosphorus decreased from a high of $1240.5 \pm 61.9 \text{ mg kg}^{-1}$ to a low of $216.6 \pm 29.6 \text{ mg kg}^{-1}$. Total nitrogen remained relatively constant with depth (average of $3.2 \pm 0.05\%$), but was highest at 18-20 cm. Total sulphur was also constant with depth (average of $0.29 \pm 0.01 \text{ mg kg}^{-1}$). C:N ratios (Figure 3-3) were quite variable from site to site, but averaged 14.4 ± 0.2 across depths and locations, typical for soils in this region. In general, C:N increased with depth, although site SJM2-T3 was an exception to this trend.

Table 3-1. Soil chemical characteristics (Average \pm Std Error)

Site	Average Bulk Density (g cm ⁻³)	TC (%)	TN (%)	TP (g kg ⁻¹)	TS (mg kg ⁻¹)
SJM1-T1	0.16 ± 0.01	43.1 ± 2.8	3.18 ± 0.17	0.62 ± 0.08	160.6 ± 0.03
SJM1-T2	0.14 ± 0.01	48.1 ± 4.9	3.42 ± 0.24	0.64 ± 0.39	130.4 ± 0.06
SJM1-T3	0.12 ± 0.01	48.8 ± 1.1	3.57 ± 0.06	0.55 ± 0.08	161.7 ± 0.04
SJM2-T1	0.29 ± 0.04	36.6 ± 3.6	2.76 ± 0.25	0.52 ± 0.05	399.0 ± 0.02
SJM2-T2	0.23 ± 0.03	42.8 ± 2.3	3.26 ± 0.14	0.62 ± 0.09	368.2 ± 0.03
SJM2-T3	0.16 ± 0.007	45.7 ± 1.2	3.47 ± 0.18	0.82 ± 0.11	315.6 ± 0.02
SJM3-SAW	0.17 ± 0.08	47.8 ± 1.0	3.13 ± 0.08	0.76 ± 0.14	364.4 ± 0.02
SJM3-TYP	0.15 ± 0.004	45.5 ± 1.1	3.20 ± 0.04	1.26 ± 0.10	355.6 ± 0.02
SJM3-OW	0.17 ± 0.01	49.5 ± 5.5	3.26 ± 0.33	0.56 ± 0.11	317.4 ± 0.04
SJM3-WIL	0.13 ± 0.004	50.1 ± 1.3	3.19 ± 0.06	0.63 ± 0.13	393.8 ± 0.02
SJM3-CTR	0.17 ± 0.02	52.4 ± 1.0	3.00 ± 0.08	0.49 ± 0.08	241.1 ± 0.03

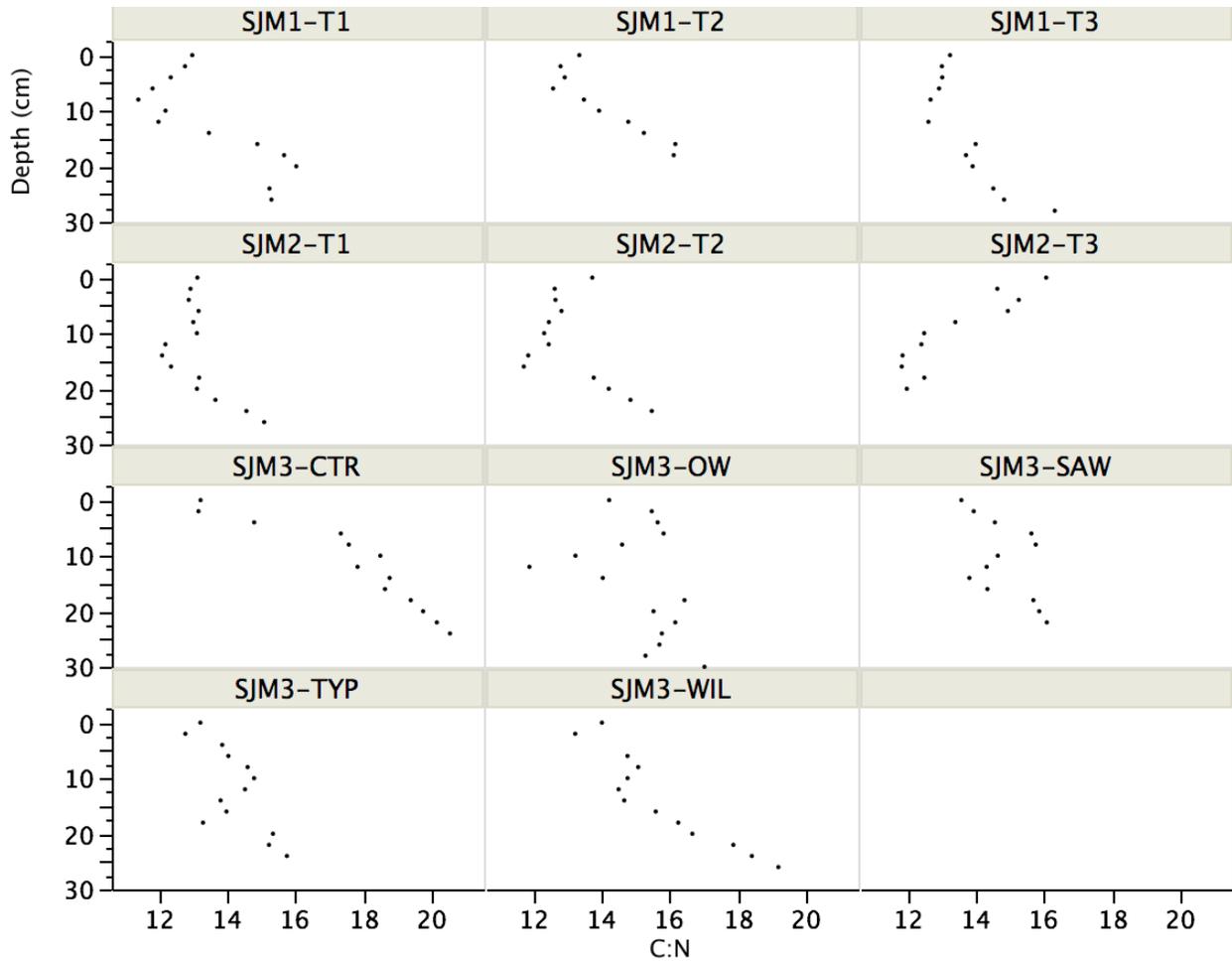


Figure 3-5. C:N and depth by field site.

Lab Nutrient Flux

As seen in Table 3-2, both TDP and TDKN fluxes were significantly higher (students t-test, $\alpha=0.05$) after cores were allowed to drain for 215 days. DOC flux was slightly lower after draining, but not significantly so.

Table 3-2. Nutrient fluxes (g m^{-2}) given as mean \pm SE (n=37). Asterisked p-values are significantly different ($\alpha=0.05$)

	Post-field	Post-drain (215 days)	p-value
DOC	0.42 \pm 0.020	0.37 \pm 0.03	0.24
TDP	0.021 \pm 0.002	0.030 \pm 0.004	0.028*
TDKN	0.062 \pm 0.004	0.20 \pm 0.020	<0.0001*

Differences in nutrient fluxes were also calculated as % change from post-field flux, and were compared between vegetation communities. These results are shown in Figure 3-6; values for % change greater than 0 indicate that nutrient fluxes were higher after draining than before. For example, TDP % change for the Slough vegetation community was approximately 100%, reflecting a two-fold increase in phosphorus flux following draining.

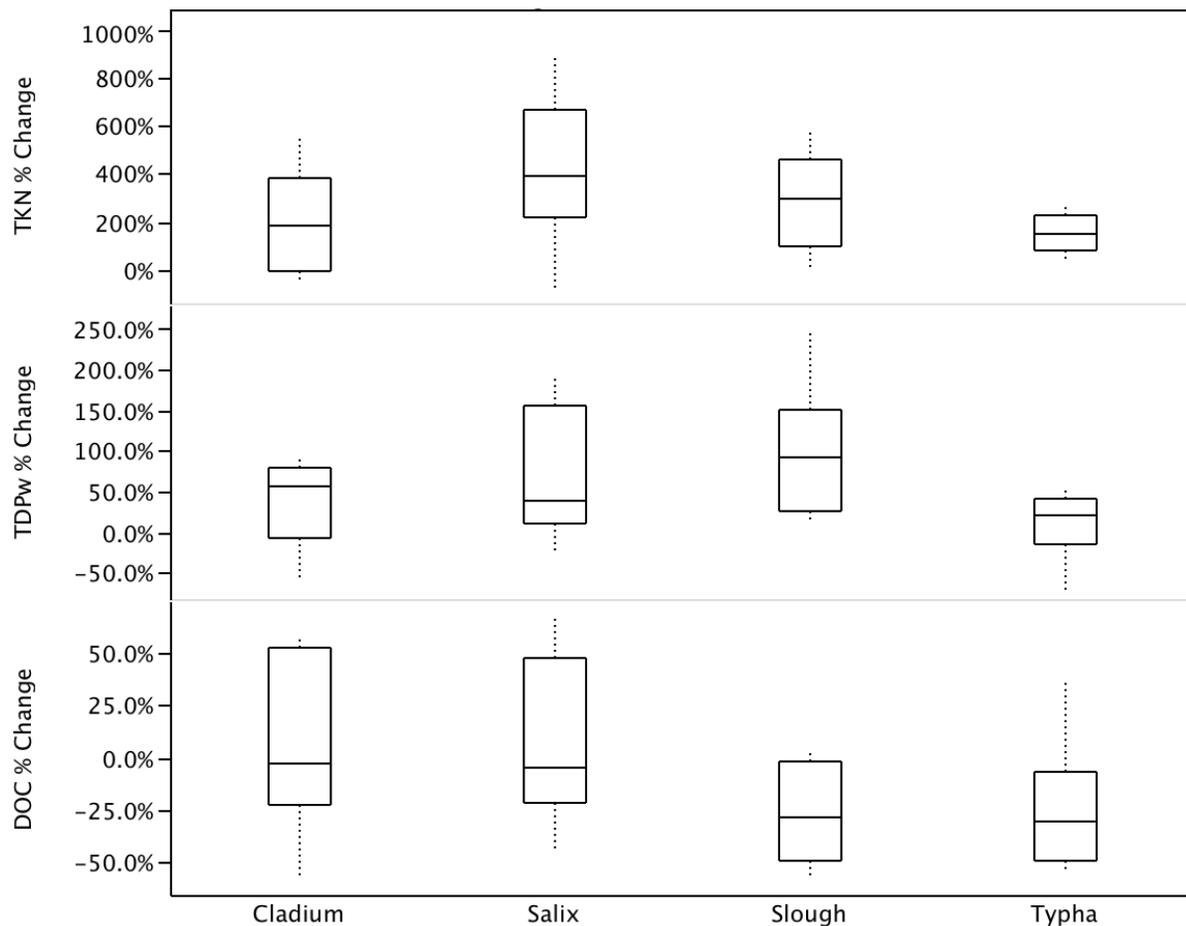


Figure 3-6. Nutrient flux % change by vegetative community (n=9).

Interestingly, the site that had the highest soil TP values (Typha or SJM3-TYP) exhibited the lowest change in P-flux following draining, and the site with the second lowest TP concentrations (Slough or SJM3-OW) showed the highest change in P-flux.

Figure 3-7 shows that soil TP was very highly correlated ($R^2 \text{ Adj} = 0.998$) with TDP % change. It is important to note that soil cores measured for TP were not the same ones used for the nutrient flux experiment, although they did come from the same field site (less than 10m apart).

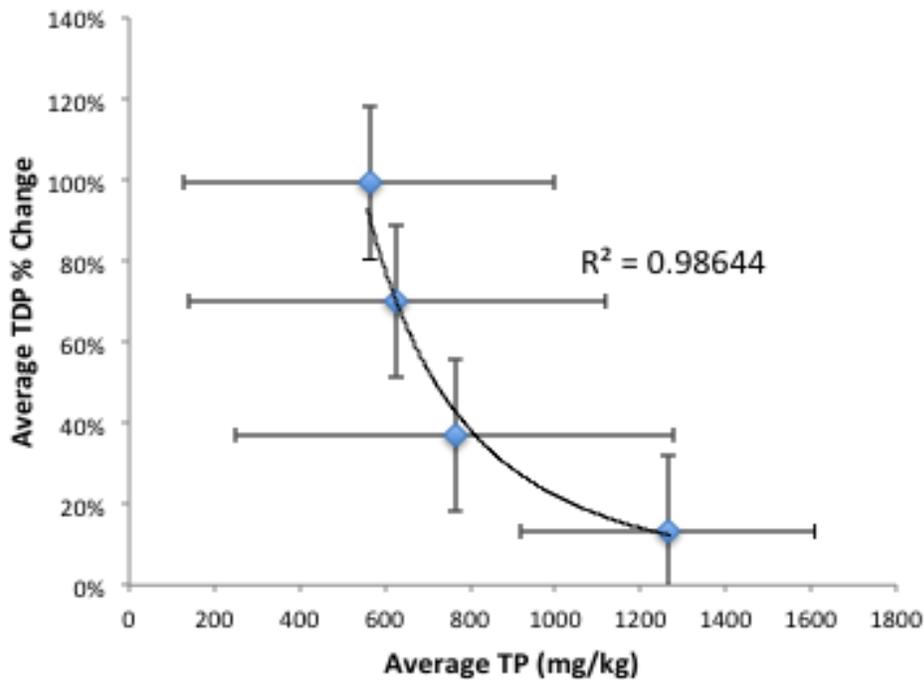


Figure 3-7. Average TP (mg kg^{-1}) and TDP% change. Reciprocal fit. $R^2 \text{ Adj} = 0.998$, $n=4$

Field Nutrient Flux

Overall data from the field nutrient flux study are shown in Figure A-1 and A-3 in the Appendix. As seen in Figure 3-8, nutrient fluxes in the field ranged from 25-50 g m^{-1} for DOC, from near 0 to 1.75 g m^{-1} for TP, and from 2-12 g m^{-1} for TDKN. Notably, between vegetation communities, TP flux was highest in the willow (*Salix caroliniana*) dominated site (Figure 3-8), and lowest in the open water (or slough) site. DOC and TDKN fluxes were also highest at the willow dominated site. The *Cladium* dominated site released the least DOC and TP, but had the second highest TDKN flux (one anomalously high value raised the mean).

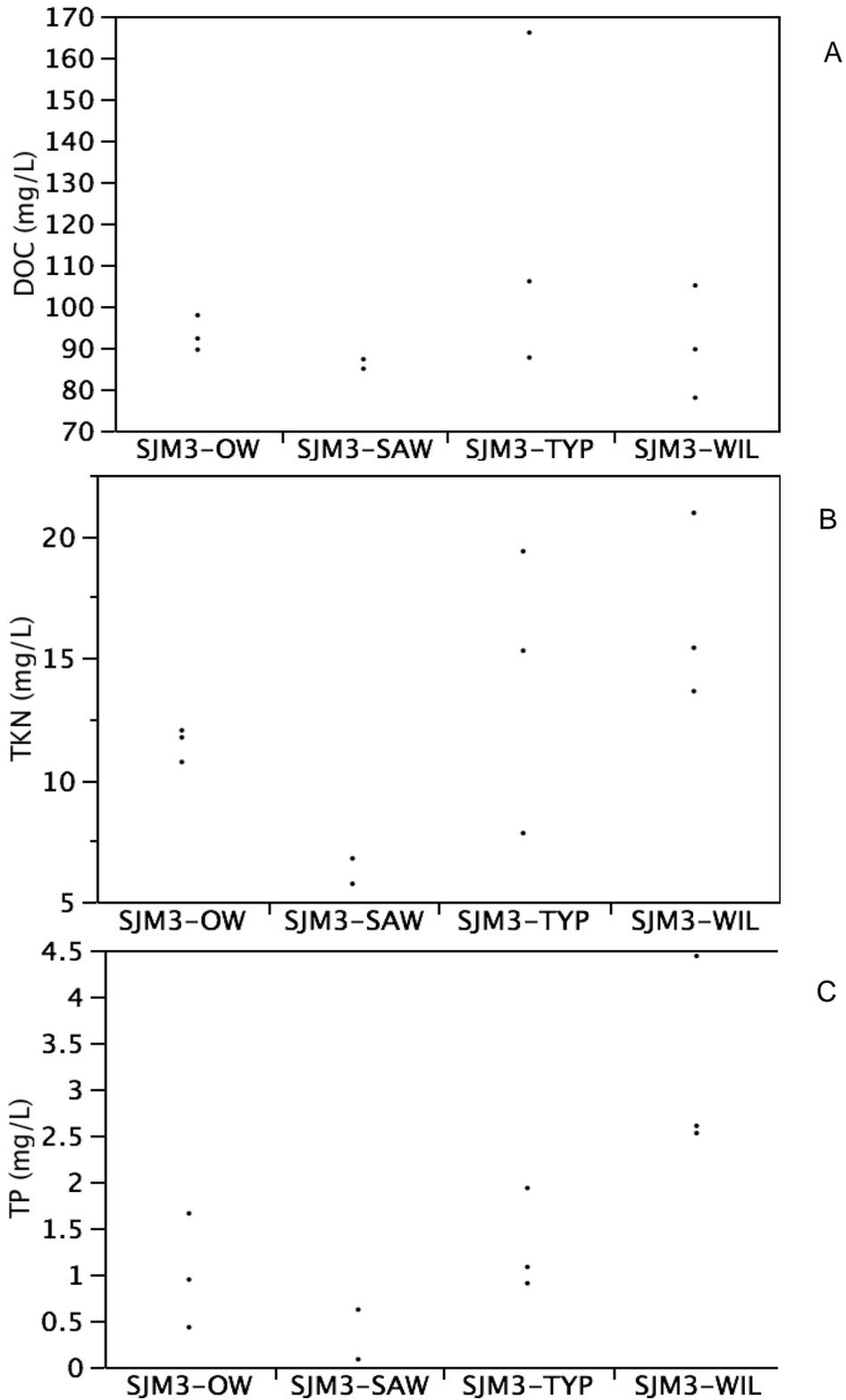


Figure 3-8. Field nutrient fluxes by site. A) Field TDKN fluxes (g/m^2). B) DOC fluxes (g/m^2). C) TP fluxes (g/m^2). ($n=3$).

Least squares regression analysis suggests, DOC, NH₄, and DOC concentrations were the only significant predictors of TDKN flux. For DOC flux, TP, NH₄, DOC, and TDKN were all significant predictors. Only PO₄ and DOC were significant predictors of TP flux (Figure 3-9).

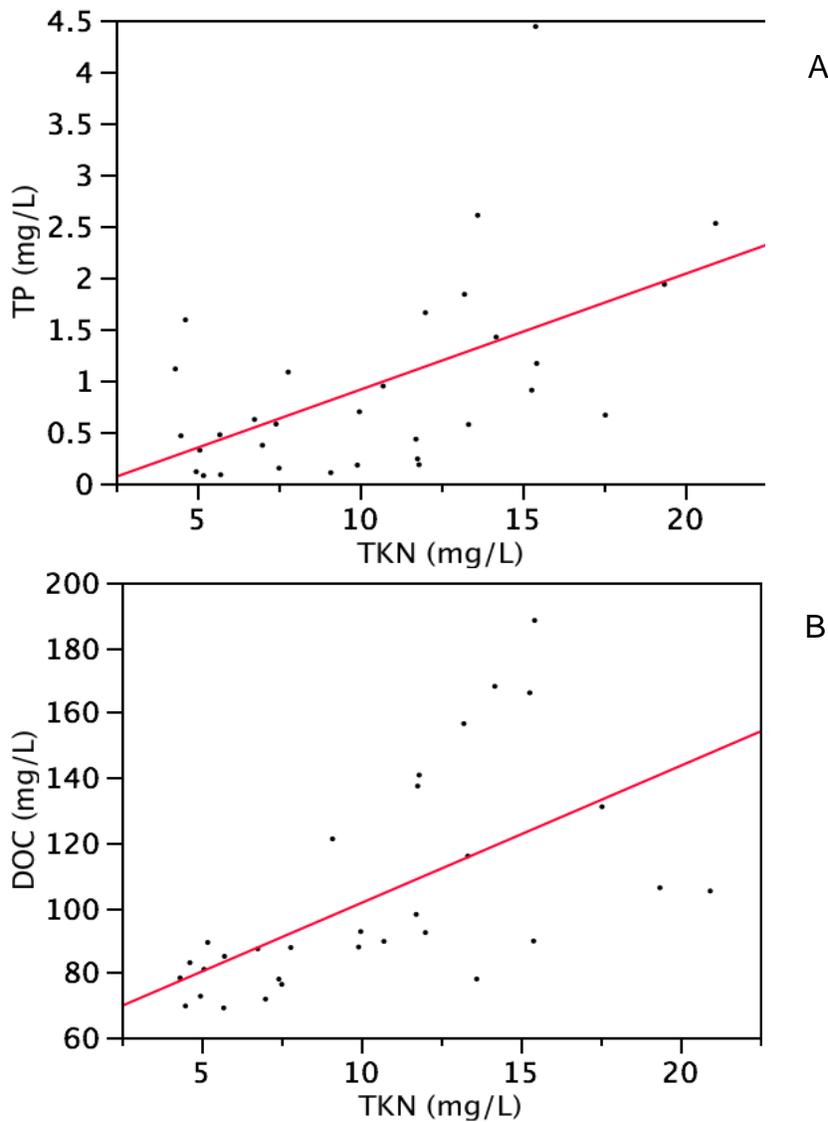


Figure 3-9. DOC and TP fluxes regressed against TKN. A) TKN (mg L⁻¹) and TP(mg L⁻¹) (R² Adj = 0.27, n=31). B) TDKN (mg L⁻¹) and DOC (mg L⁻¹) (R² Adj = 0.34, n=31). TDKN outliers of 40.2 and 50.6 mg L⁻¹ were removed.

During the laboratory nutrient flux experiment, it was possible to impose an equal level of flooding for all soil cores. This way, all samples would be diluted to the same

level and any differences in nutrient flux would be a function of soil processes, not experimental error. In this field experiment it was not feasible to sample with all soils flooded to the same level. Consequently, all field water samples were diluted to different levels based on their relative elevations. Therefore, in order to compare nutrient fluxes between the laboratory and field studies, fluxes were normalized by depth of flooding and area sampled to create Figure 3-10. The units for this Figure are grams per m² of soil per cm of floodwater (g m⁻² cm⁻¹). In general, nutrient fluxes in the field were approximately 10 times greater than in the laboratory (both initial and post-drain).

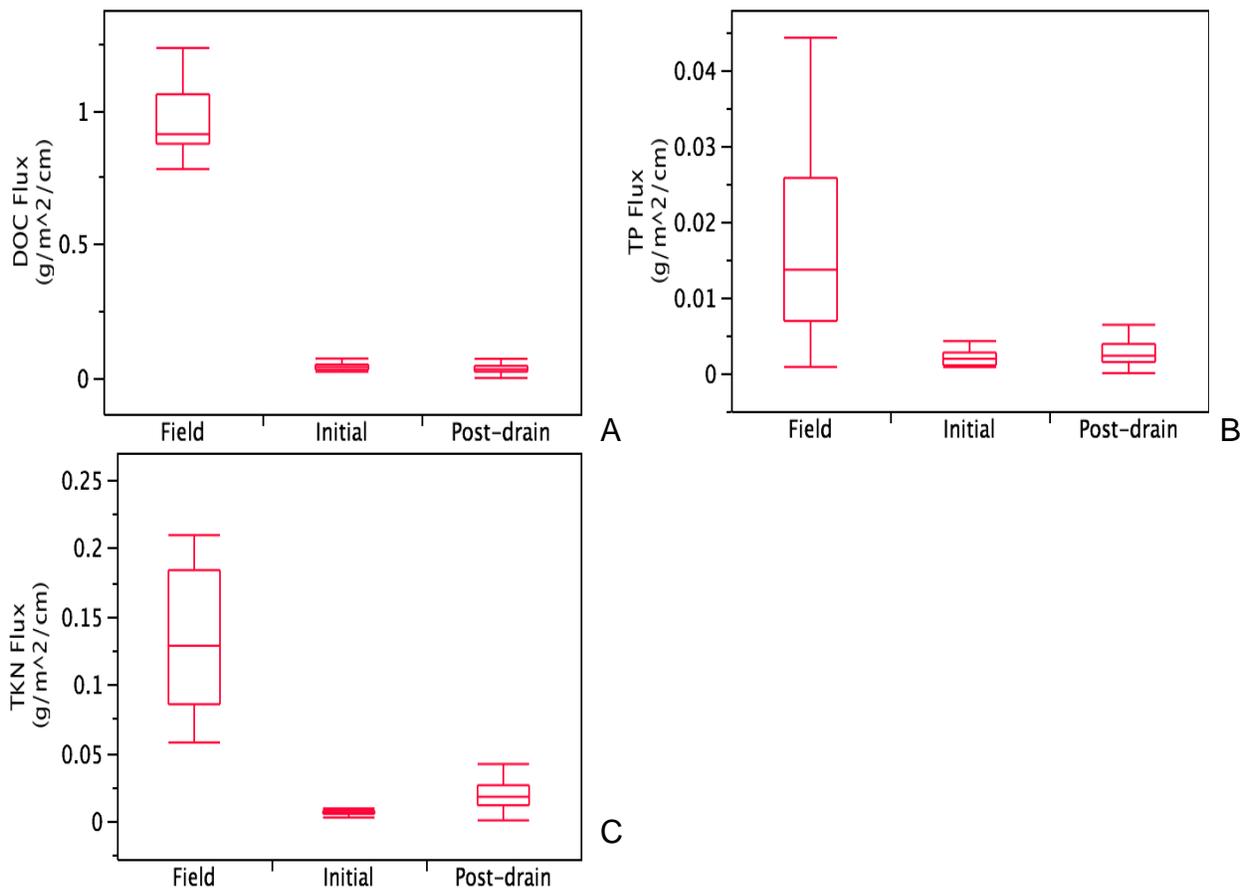


Figure 3-10. Field nutrient fluxes compared with laboratory nutrient fluxes. A) DOC fluxes (g m⁻² cm⁻¹). B) TP fluxes (g m⁻² cm⁻¹). C) TDKN fluxes (g m⁻² cm⁻¹) for field and lab studies

Discussion

Soil Chemistry

Soil TP was relatively consistent between the transect sites, but was more variable between the different vegetation communities. Soil TP at the Typha dominated site (1.26 mg kg^{-1}) was significantly higher (students t-test, $\alpha=0.05$) than at all other sites. This finding was expected as typha is known to become established where phosphorus is found in excess (Lorenzen et al. 2001; Maher 1997). Both TC and TN were much more consistent from site to site, ranging from 36.59-52.75% and 2.76-3.57% respectively. Because these soils are highly organic, high TC and TN values were expected. Only the slough site (SJM3-OW) showed significantly higher TN and TC than any other vegetation communities.

In soil profiles from all sites, TP was highest in the surface and decreased in concentration with increasing depth. This is characteristic for soils that have experienced nutrient loading in their recent history. Another possible explanation for the high TP seen in surface soils could be the presence of benthic algae. As TP analysis does not exclude phosphorus associated with microbial biomass, any benthic algae in the soil will contribute to a higher final TP value. As opposed to larger plants that need a high carbon structure for support, algae have no such need, therefore have much lower C:P ratios (Brenner et al. 2001; Reddy et al. 1999). Nevertheless, the fact that surface soils at all sites were P-enriched indicates to me that allochthonous sources of P are the primary culprit.

At the C:N ratios seen in this study (14.4 ± 0.2), ammonification (mineralization of organic nitrogen to NH_4^+) and subsequent N loss to pore/flood water can be expected during both flooded and drained conditions. Additionally, this low C:N ratio indicates that

upon draining, initial organic N mineralization will occur relatively rapidly. This is because microbial N demands (for biomass buildup) are quickly satisfied, allowing subsequent N to be released to the soil solution (Reddy and DeLaune 2008). Interestingly, the one site that highest surface C:N ratios (and a trend with depth opposite of the other sites) also had by far the lowest pH when flooded.

Both TN and TC exhibited a negative relationship with bulk density, with higher nutrient levels corresponding with lower bulk densities. As soils with more organic carbon generally have lower bulk densities relative to more mineral soils, this relationship between TC and density is consistent. It was expected that soil bulk density would exhibit a strong relationship with depth, but interestingly any association was weak ($\text{Adj } R^2 = 0.23$, $n=136$). TC and TN were good predictors of bulk density ($\text{Adj } R^2$ of 0.29 and 0.20 respectively) while TP and TS were not. Average N:P ratios (top 10cm) were not significantly different between the various vegetation communities in this study, although TP was significantly higher in the typha site than in the other vegetation communities. This could very well be due to the fact that only one core was taken from each site and sampling was only conducted once.

It is generally accepted in the literature that invasive species like cattails (*Typha* spp.) gain dominance in wetlands when phosphorus limitations are decreased (Lorenzen et al 2001; Wright and Reddy 2001; Reddy et al 1993; Liu et al. 2010). As the sites chosen for this study were intended to examine the chemical differences between soil with invasive and native vegetation, we expected to see more P-limitation at the native sites (*Cladium* and slough/open water) compared to the sites with invasive species. Nevertheless, average N:P ratios (top 10cm) were not significantly different

between the various vegetation communities in this study, indicating that all surface soils were similarly P-limited. While this seems contradictory, it should be noted that vegetation observed during sampling is only representative of conditions during recent years. As the SJMCA has undergone severe changes in hydrology over the last few decades, soil deeper in the profile was most likely produced by vegetation different from that observed during this study. Thus, it is not surprising that soil chemical profiles generally appear to converge as depth increases. For comparison, the high and low TP values found in this study (1240 to 216 mg kg⁻¹) are similar to impacted and normal Everglades soils respectively (Osborne et al. 2011).

Lab Nutrient Flux

Both TDP and TDKN fluxes increased after soils were drained, while DOC flux decreased. This difference can be explained in part by the C:N ratios found previously. As soil C:N was 14.4, and microbial biomass is generally between 8 and 12 (Cleveland and Liptzin 2007), it would be expected that upon draining, soil carbon would be lost to the atmosphere as CO₂, and nitrogen would be microbially cycled within the soil, causing the C:N ratio to decrease, approaching 10.

A previous study by Olila et al. (1997) found that floc sediments drained for 6 weeks exhibited a 10-fold higher phosphorus flux to the water column after flooding relative to cores drained for only 3 weeks. This study demonstrates that for peat soils, a similar relationship exists; increased drainage time results in higher P-fluxes to the water column. Using average nutrient flux values (after draining), nutrient loads for the entire SJMCA are 51.3, 4.2, and 27.8 metric tons for DOC, TDP, and TDKN respectively.

Field Nutrient Flux

The willow dominated site exhibited the highest TP flux upon flooding. This is interesting because the willow site also had relatively low soil TP values. A similar trend was seen during the laboratory study where TDP flux increased more after draining for low TP soils than for high TP soils (seen in Figure 3-7). Higher (not significantly) DOC and TDKN fluxes were also seen for the willow site. These high nutrient fluxes from the willow site were not expected, but could be due to a few different factors. First, the rooting structure of the willow community in the field site was very thick; adventitious roots were common and very dense. These root structures are produced by the plant for the purpose of oxygen introduction to the rhizosphere, which can then be used to mineralize phosphorus by microbes (Lorenzen et al. 2001). Additionally, prescribed burns are commonly used by the SJRWMD to manage the spread of willow with mixed success (Lee et al. 2005). As a result of fire, phosphorus that is normally unavailable is mineralized and can be leached into the water column. In 1991, Spencer and Hauer observed that soluble reactive phosphorus (SRP) in streams increased between 5 and 60 times during wildfire occurrence. While this 1991 study shows that fire can mineralize P, the effects on P-flux may only be relevant for short periods of time after the fire. Liu et al. (2010) demonstrated that the first 24 hours after burning were most crucial to the fate of mineralized phosphorus. Therefore, if fire (wild or prescribed) has occurred in the salix community, it is reasonable that P-flux would be high.

It was expected that field sites with the highest CO₂ fluxes would produce the largest nutrient fluxes when soils flooded, however, this trend was not observed. In all cases, nutrient fluxes in the field were roughly 10 times as high as those in the lab. While it was expected that soils in the field would release more nutrients than in the lab,

this magnitude of difference was somewhat surprising. Many factors may have led to the relatively high flux of nutrients from the field sites. First, there is significantly more soil in the field; using a metal rod, the peat was found to be at least 3-4m deep at most sites (soil cores were only 15cm long). While this difference in total soil volume is certainly a concern, a study of USJRB soils by Reddy et al. (2006) showed that carbon mineralization is highest in the top 20cm; beneath this depth, mineralization decreases significantly. Second, flora and fauna may contribute to nutrient cycling in the field, while they are excluded from the lab study. Additionally, temperatures in the field are much more variable than in the laboratory. A study by Inglett et al. (2011) shows that both temperature and vegetation are significant predictors of aerobic/anaerobic respiration.

Concluding Remarks

The objective of this study was to determine how water management affects nutrient movement within the SJMCA. Hydrology does appear to have an influence on nutrient movement through and from the wetland. When soils were drained and re-flooded, fluxes of dissolved nitrogen and phosphorus increased significantly. Additionally, soils that were populated with invasive plants like cattails and willow generally released more nutrients (not always significantly) than soil colonized by native species like sawgrass.

While both the insitu and laboratory studies measure similar factors, their respective utilities to management in the SJMCA are different. Figure 3-10 shows that the two studies produced somewhat disparate results. As the laboratory nutrient flux study occurred in a much more controlled way (cores flooded and sampled at same time), its results should be more useful in predicting the rate of nutrient loss after water table drawdown as a function of vegetation type. The primary utility of the insitu study is

to show the influence of environmental factors on nutrient flux. The finding that insitu nutrient fluxes were much greater than those seen in the laboratory indicates that factors like vegetation, temperature, and macrofauna have a significant influence on the loss of nutrients from the SJMCA.

Given that nutrients fluxed from the SJMCA will be translocated downstream to the SJR, where they can contribute to HABs and fish kills, care must be taken when the water level is allowed to decrease. This study has shown that phosphorus in particular responds very strongly to water table drawdowns, increasing from 0.021 to 0.030 g m⁻² after drainage. As phosphorus is the nutrient most often linked with the occurrence of HABs and DO crashes (Chapman et al. 2001; Paerl 2007; Steidinger et al. 1998), and summer storms can rapidly flush this phosphorus downstream where it can cause ecological harm; water tables during the summer months particularly should be kept at a level high enough to maintain anaerobic conditions in the surface soils. This management scheme would allow for much of the flood control properties of the wetland to be maintained, while still limiting the loss of nutrients from the SJMCA.

CHAPTER 4 SUMMARY AND CONCLUSIONS

The primary research objectives of this study were to answer the questions: 1) what are the major causes of soil subsidence, and 2) how can the current hydrologic management scheme best be modified to reduce or eliminate organic soil subsidence and allow the SJMCA to meet its requirements of: flood protection, environmental enhancement, water supply, and recreation. Multiple laboratory and insitu studies were undertaken in order to answer to these questions.

This study indicates that under current management practices, soil subsidence will continue to be a factor in the SJMCA. When the soils are allowed to drain for approximately 1 month per year or more, total subsidence (primary and secondary) will exceed sediment accumulation, causing net soil loss. Although some previous studies have suggested that soils drained for multiple years experience more primary subsidence than secondary, this study provides data to the contrary. Here, it was shown that secondary subsidence exceeded primary subsidence for the SJMCA, a wetland that has been chronically overdrained.

While vegetative type may not be particularly critical in determining CO_2 flux rates, temperature and water depth are key to predicting carbon mobility. Both CO_2 and CH_4 fluxes demonstrated a strong relationship with hydrology. CO_2 flux in the field was only $417 \text{ g C m}^{-2} \text{ yr}^{-1}$ when flooded, but increased to $2595 \text{ g C m}^{-2} \text{ yr}^{-1}$ when the water table fell below the surface. CH_4 flux was $30 \text{ g C m}^{-2} \text{ yr}^{-1}$ when flooded, but decreased to $7.2 \text{ g C m}^{-2} \text{ yr}^{-1}$ when the soil was drained below the soil surface. During the course of the study, the water table was beneath the soil surface for roughly 8 months out of the year. Thus, aerobic respiration of soil OC is the most important factor regulating carbon

cycling in the wetland, and should be considered when manipulating the water table in the SJMCA in order to avoid excessive soil loss.

The C:N ratios found in this study indicate that the carbon quality of surface soils in the SJMCA is relatively high. This implies that upon water table drawdown, significant mineralization of SOC should be expected, with concomitant oxidative carbon losses. The high carbon quality of surface soils in the SJMCA could also partially explain the unexpectedly high fraction of total subsidence caused by secondary factors. High levels of TP in surface soils could also theoretically help explain the high level of secondary subsidence, although no relationship was found in this study between phosphorus and CO₂ flux.

Conversely, the high C:N ratios seen lower in soil profiles indicate lower quality carbon deeper in the soil. This refractory carbon source generally yields a lower average respiration rate relative to more labile substrates (Conant et al. 2008; Kechavrzi et al. 2010), although a 2011 study by Inglett et al. indicates that both anaerobic and aerobic CO₂ fluxes are not correlated with carbon quality. As seen in the insitu subsidence study, once the water table dropped below approximately -20 cm, the rate of CO₂ flux did not change significantly. This suggests that allowing the deeper soils in the SJMCA to become exposed to oxygen does not have a significant effect on secondary subsidence. The implication for management here is that the negative effect of wetland drainage is limited to the first 20 cm of water table drainage; beneath this level, additional drainage has little to no effect on gaseous carbon loss.

While the gaseous loss of carbon is clearly a significant cause of soil subsidence, soil material can be lost through processes besides oxidation. Aqueous carbon lost from

the soil while flooded could also be considered a contributing factor to subsidence. This study showed that roughly 1 g C m^{-2} was lost from the soil for every centimeter of floodwater. Although this loss of carbon is small relative to secondary subsidence, it is still a loss that is not currently assessed.

This study also shows that hydrology also has an influence on how nutrients move through the wetland. When soils were drained and re-flooded, fluxes of dissolved nitrogen and phosphorus increased significantly. Soils that were populated with invasive plants like cattails and willow generally released more nutrients (not always significantly) than soil colonized by native species like sawgrass. Areas colonized by cattails also had higher levels of soil TP, possibly allowing for higher phosphorus fluxes when flooded. Given the current pattern in the SJMCA of drainage during the warm summer months and flooding during the winter, persistent soil loss will continue. As drained CO_2 flux was found to be 6 times higher than flooded flux, and these rates are temperature dependant, draining of the SJMCA during the summer is the worst possible case for carbon stability within the wetland. Additionally, extreme seasonal nutrient fluxes can occur, leading to the algal blooms and fish kills that blight the St Johns River annually. In light of both of these factors, soil carbon loss concomitant with nutrient flux, the soils in the SJMCA must be better managed to protect the services it provides to Florida's citizens and environment.

It would seem that sites with high nutrient loss upon flooding would also exhibit high CO_2 fluxes because microbial decomposition rates influence both factors. Nevertheless, this relationship between nutrient flux and CO_2 flux was not seen. In both laboratory and insitu studies, TDP, TDKN, and DOC fluxes predicted less than 5% of the

variability in average field CO₂ fluxes (R^2 Adj < 0.05 for all comparisons). This indicates that the factors controlling nutrient losses in the SJMCA are different than those governing CO₂ flux.

While water table drawdowns during the summer may be necessary to prevent flooding in the surrounding urban areas, the unintended consequences of oxygenating the soil are severe. Over time, the soil that retards the flow of water during high rainfall events will be lost due to primary and secondary subsidence.

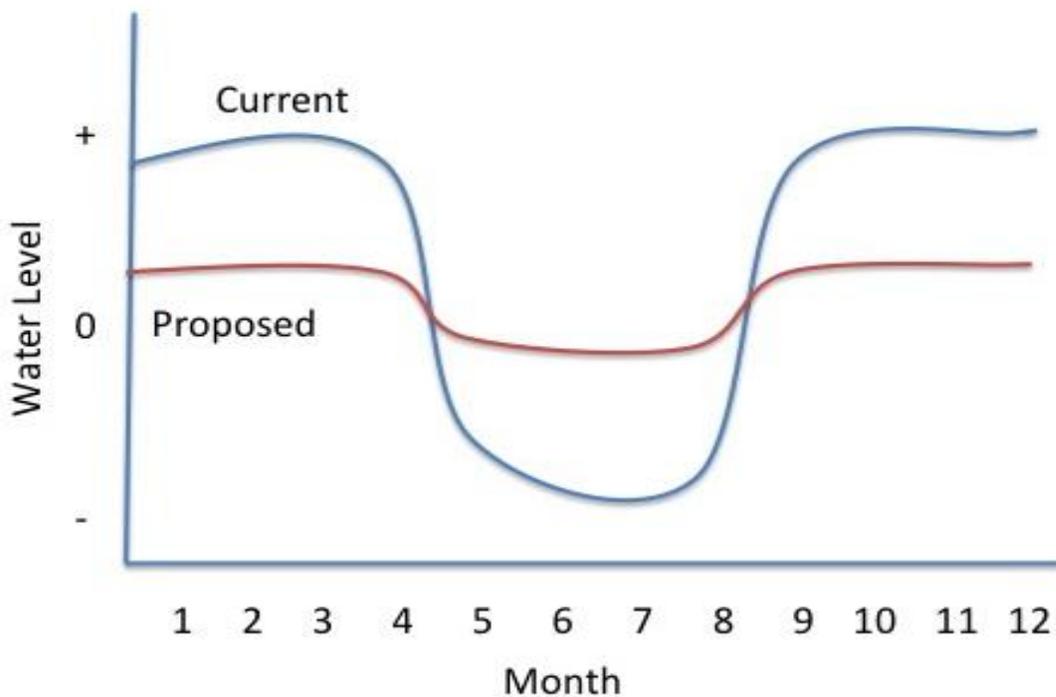


Figure 4-1. Sketch of current and proposed water table regime for SJMCA

Therefore, if the water table can be kept high enough to maintain anaerobic conditions during the warm summer months, oxidative carbon loss (as well as subsequent nutrient mineralization) could be diminished substantially. While keeping the water level at the soil surface would be ideal, an additional drawdown of 10 cm would only have a small impact on the loss of soil carbon through oxidation. Thus, the SJMCA

should not be drained below -10 cm. Average yearly water tables would be kept the same, but would be lower in the winter and higher in the summer (Figure 4-1). This would not only help to reduce soil and nutrient losses, but would also go some way to restore natural wetland hydrology to the SJMCA.

APPENDIX
ADDITIONAL FIGURES

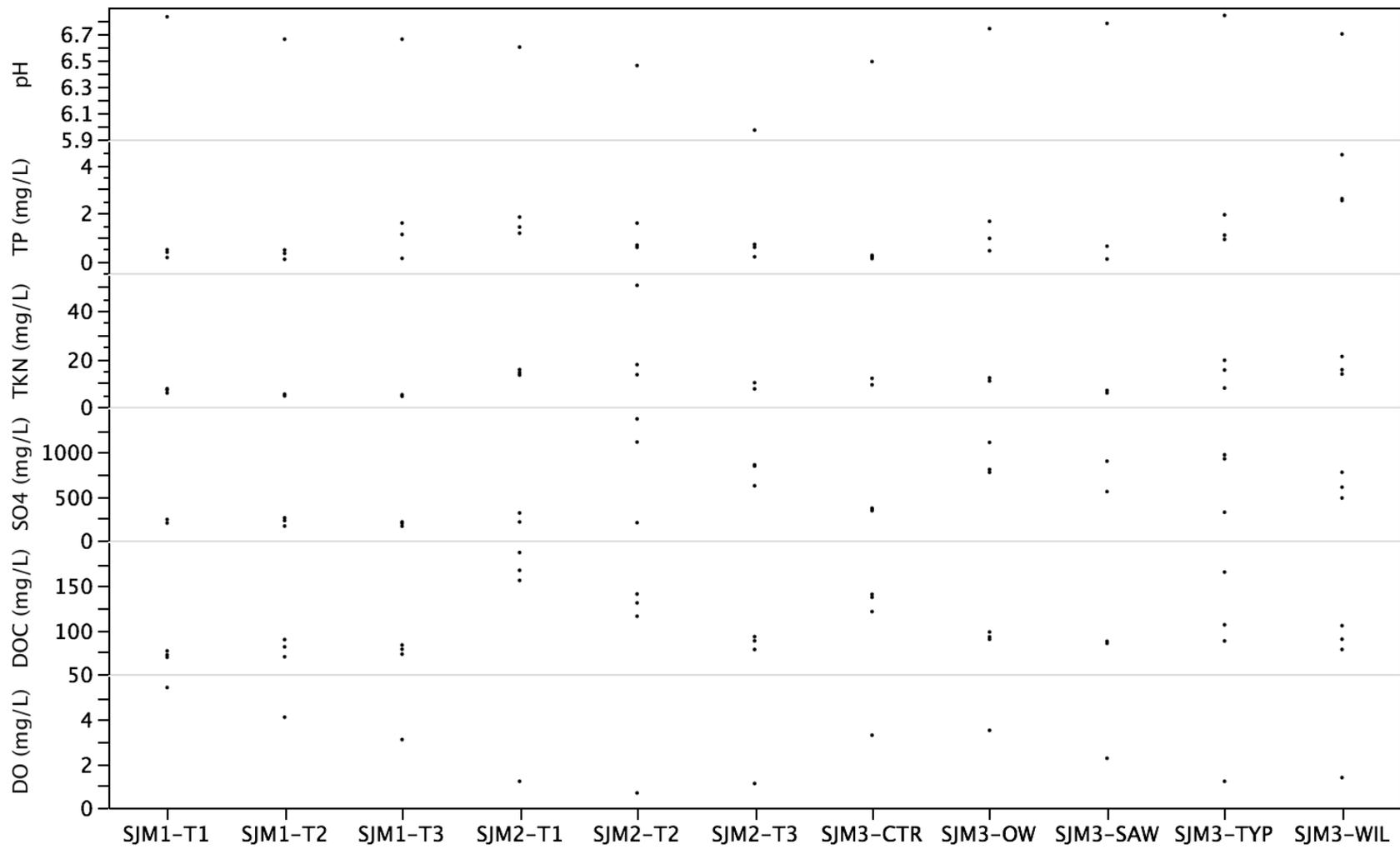


Figure A-1. Field DO (mg L^{-1}), DOC (mg L^{-1}), NH_4 (mg L^{-1}), TDKN (mg L^{-1}), TP (mg L^{-1}), and pH. n=3

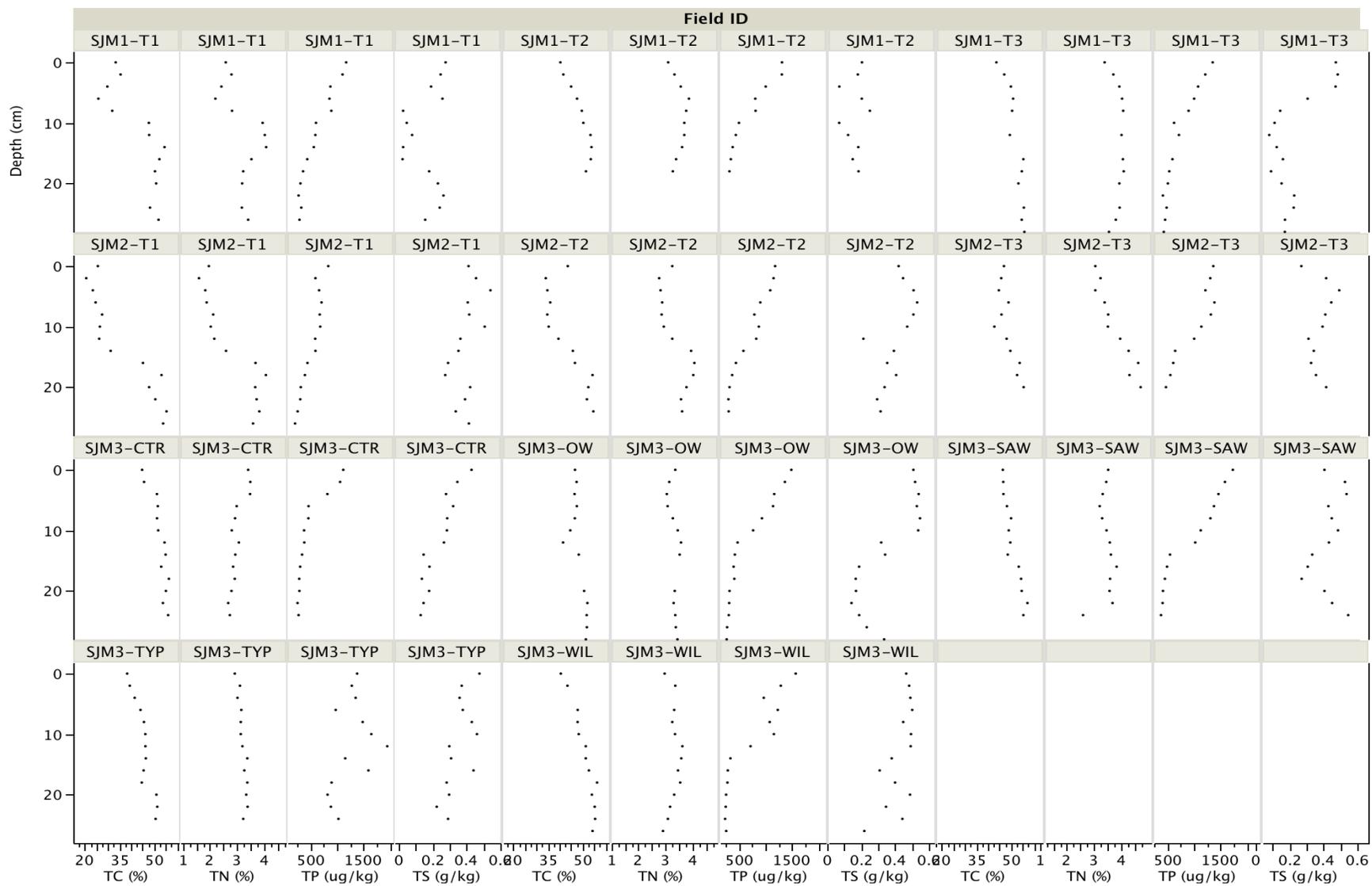


Figure A-2. Depth profiles for TC (%), TN (%), TP ($\mu\text{g kg}^{-1}$), and TS ($\mu\text{g kg}^{-1}$) by site. n=1

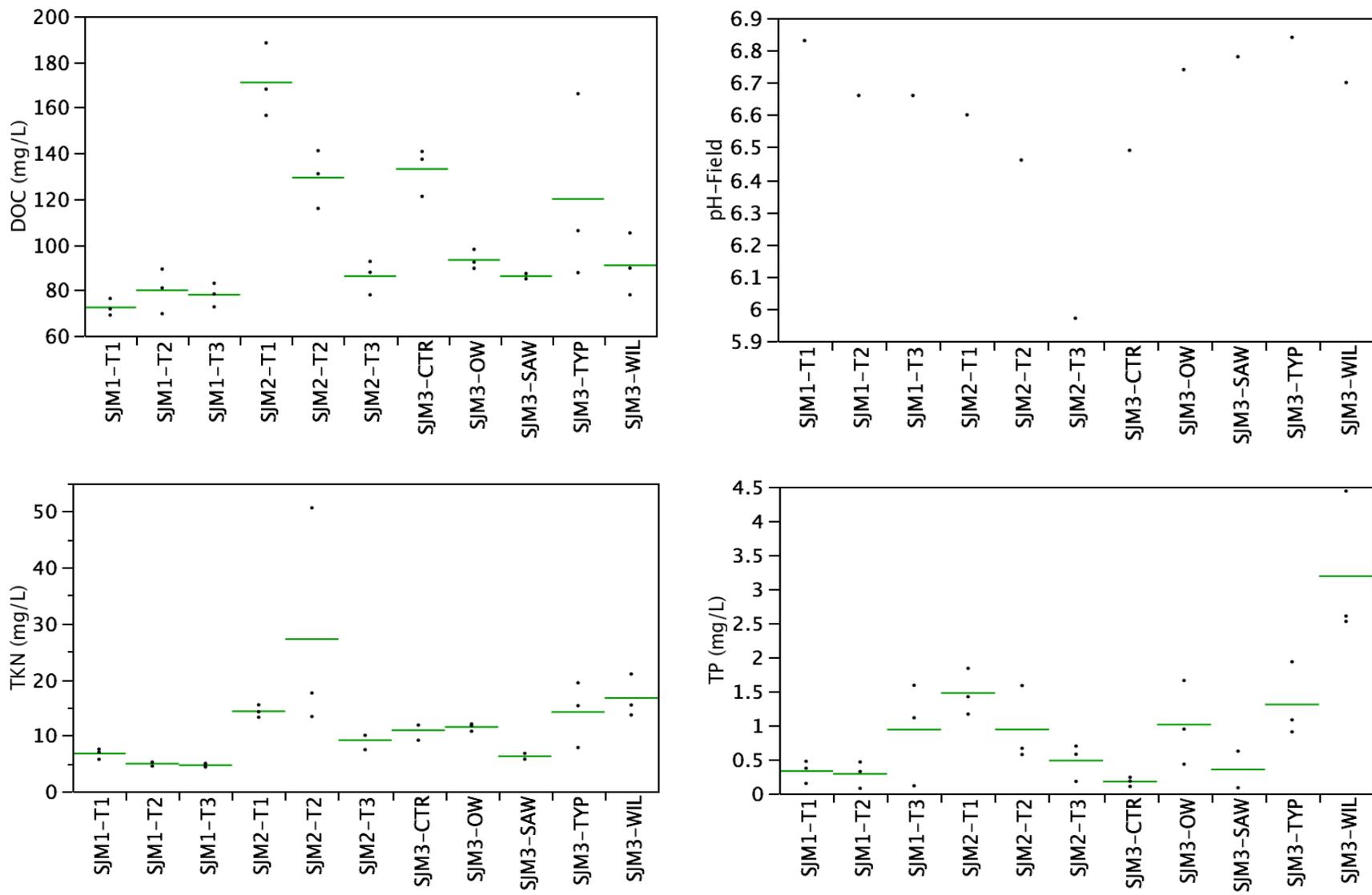


Figure A-3. Major nutrient fluxes (mg/L, n=3) and pH (n=1) for all field sites

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BIOGRAPHICAL SKETCH

Bryce Van Dam was born in Tallahassee, FL in 1987. After finishing high school, Bryce enrolled at the University of West Florida in 2005. In 2010, he graduated with a Bachelor of Science in environmental science/natural science, and a certificate in geographic information science. In 2012, Bryce completed a Master of Science degree under Dr. Todd Osborne in the Soil and Water Science Department with a focus on wetland sciences at the University of Florida.