SPECIAL PUBLICATION SJ2007-SP5

INFLUENCE OF WATER LEVELS ON SUBSIDENCE OF ORGANIC SOILS IN THE UPPER ST. JOHNS RIVER BASIN

Influence of Water Levels on Subsidence of Organic Soils in the Upper St. Johns River Basin

Final Report
Contract SH45812

September 2006

Final Report Submitted to:
St. Johns River Water Management District
4049 Reid Street, P.O.Box 1429
Palatka, Fl 32178-1429

Prepared By:

K. R. Reddy, T. Z. Osborne, K. S. Inglett, and R. Corstanje
Wetland Biogeochemistry Laboratory
Soil and Water Science Department
University of Florida-IFAS
Gainesville, Fl 32611

Table of Contents

Executive Summary

The accumulation of the characteristic organic soils of the Upper St. Johns River Basin (USJRB) has taken several thousand years. The stability of these soils is highly dependent on hydrology as their formation is due to the historically extended hydroperiod of the USJRB floodplain wetlands. In some regions of the USJRB, the hydrology of the floodplain has been dramatically altered, resulting in substantial loss of organic soil. Knowledge of the critical water depth at which accelerated soil loss occurs is needed to refine estimates of marsh water levels which are protective of the soils of the region. The Environmental Water Management Plan for The Upper St. Johns River Basin (1996) calls for a "mean depth and inundation frequency of the central critical elevation such that there will be no net loss of organic soils through oxidation." This is one element of a set of hydrologic criteria that includes frequency of inundation, maximum depth, magnitude of annual fluctuation, timing of fluctuation, and water level recession rates. Environmental hydrologic criteria that numerically describe each of these characteristics are currently being developed and refined. These criteria will ultimately be used to direct the operation of project structures when water levels are below established flood control regulation schedules.

The main goal of this research was to determine the minimum water levels in wetlands needed to prevent net loss of organic soils that eventually leads to subsidence of soils in the Blue Cypress Marsh (BCM) of USRB. This was achieved by investigating the effect of water drawdown on soils with different vegetation type, temperature and nutrient levels. The effect was determined by measuring microbial activities such as aerobic and anaerobic respiration, CH_4 production, phenol oxidase activity and β glucosidase enzyme activities. The objectives of this study were to:

- review current extent of literature on the subject of soil subsidence with special focus on organic soils
- determine relative proportion of labile, moderately labile, and refractory pools in soils of BCM
- determine soil organic matter loss rates as a function of water level relative to soil surface
- determine the influence of inundation frequency on soil organic matter decomposition
- determine the influence of dominant vegetation type, temperature, and soil nutrient levels on soil organic matter decomposition.

The first objective was to review the pertinent literature available on the subject of soil subsidence with special focus on organic soils in subtropical climates. We found the following:

- Most literature available focuses on Everglades soils and agricultural organic soils of the Everglades Agricultural Area and are out of date

- Organic soil subsidence is the result of over drainage and involves three major processes; oxidation of organic matter, loss of soil buoyancy and shrinkage of organic soils when water is removed
- Loss of soil due to microbially mediated oxidation of soil organic matter under aerobic conditions is the most damaging effect of over drainage as the loss of organic matter is irreversible
- Major factors influencing organic matter oxidation are organic matter lability, soil moisture, oxidation/reduction potential, nutrient availability, temperature, and microbial communities/activities

The second objective was to characterize the BCM soil organic matter from five sites with various dominant vegetation and nutrient enrichment status by qualitative and quantitative means. Investigation of soil physical and biogeochemical parameters, as well as, organic matter lability were used in this characterization.

- Sites dominated by *Typha* and *Salix* (nutrient impacted) where the two highest sites observed with respect to soil phosphorus
- All sites were similar in value with respect to LOI, pH, and TC. TN was variable within sites and nutrient impacted sites were not the highest values observed
- Bulk density was lowest in *Typha* and *Nymphea*, and highest in *Salix* site, suggesting occasional drawdown events in the *Salix* area
- Fractionation of soil organic matter into labile, moderately labile, and recalcitrant/refractory pools indicated *Typha* and *Nymphea* sites had the largest pools of labile and moderately labile organic matter. *Cladium* and *Panicum* sites, were indicative of unimpacted BCM soils with high levels of recalcitrant organic material. It is expected that *Typha* and *Nymphea* dominated sites would be the most susceptible to oxidation in the event of drawdown.

The third objective of this research was to determine the effects of various water levels on oxidation of BCM organic soils. Microbial biomass carbon, enzyme activities, CO_2 production, and redox potential were used to evaluate oxidation processes under various water table scenarios.

- Rates of CO_2 evolution and thus oxidation of soil organic matter increased as water table decreased. This relationship was constant throughout the experimental period, with highest rates being from the cores with lowest water tables and lowest being from the flooded cores. Evaluation of the data suggests that the top 10 to 15 cm is the most reactive area with respect to microbial oxidation. Therefore, this layer of reactive soil is most susceptible to oxidation during drawdown events. Below 15 cm the CO_2 flux does not increase as dramatically with each increment of water table drawdown.
- Redox potential (Eh) indicated that capillary fringe existed above the water table sufficient to maintain anaerobic conditions above the water table in most cores.
- Annual secondary subsidence (subsidence due to oxidation of organic matter) was calculated to be 1.2mm for flooded soils; 2.4 mm for surface inundation; 2.9 mm when water table drops to -2 cm; 3.6 mm at -5cm; 4.8 mm at -7.6 cm; 5.4 mm at -15 cm; 5.4 mm at -20 cm; and 5.7 mm at -30 cm water table.

- Phenol content was high in flooded and saturated soil cores. Drained soil cores showed a decline in total phenolics content with the lowering of water table. There was no correlation found between soil moisture content and their total phenolics content
- Phenol oxidase activity was higher in drained cores than in flooded cores, however there was no correlation between the water level drawdown and the enzyme activity. Presence of this enzyme activity was detected in soils as deep as 30 cm.
- β glucosidase activity was present in all drained and flooded cores, but did not show any increasing or decreasing trend with the water table drawdown.. Relatively higher β-glucosidase activity was observed at the depth of 7.6 to 10 cm in soil cores. BGA was also correlated with the moisture content in drained soils.

The fourth objective of this research was to determine the effect of frequency of inundation on soil organic matter oxidation by comparing CO_2 flux measurements from three different flooding regimes over the course of many months.

- Repeated drawdown and reflooding events significantly reduced the amount of organic matter oxidation in comparison to continual drawdown events for similar time periods.
- Frequencies of flooding and draining at intervals of 10 days or less are suggested for protection of organic soils in the BCMCA.
- Experimental results suggest secondary subsidence (subsidence due to organic matter oxidation) is estimated to occur at a rate of 1mm a year under flooded conditions, 2.5 mm yr^{-1} when flooded and drained at a frequency of 10 days, 2.7 mm yr^{-1} at a frequency of 25 days, 3.5 mm yr^{-1} at a frequency of 50 days, and 5 mm yr^{-1} when continually drained to a depth of 15 cm.

The fifth objective of this study was to evaluate the effects of different vegetation communities (OM source), temperature, and soil nutrient status on aerobic and anaerobic decomposition of soil organic matter.

- Organic matter decomposition rates in all soils were higher during aerobic conditions and increased with temperature. Rate of CO_2 evolution during aerobic incubations was (3-10 times) higher than that observed during anaerobic incubations. Temperature coefficient values (Q_{10}) varied within soils from different vegetation community regions in the BCMCA. Q_{10} values were 1.5-2 times higher for temperature ranges of (20-30ºC) than those determined at 10-20ºC.
- *Nymphaea* & *Eleocharis* soils, representative of slough ecotypes, evolved more CO_2 than other vegetation types under both aerobic and anaerobic conditions. CO_2 evolution from *Panicum* soils, representative of wet prairie ecotypes, was less than that in *Nymphaea* & *Eleocharis* soils but higher than other soils. In contrast, *Cladium* soils, another wet prairie vegetation community, had the lowest decomposition rates under anaerobic conditions.

- *Typha* dominated region represents high nutrient levels and long hydroperiods. Aerobic decomposition rates of SOM from this region were lower than other vegetation communities except at 30ºC where it was higher than all other vegetation communities. A similar stimulating effect of temperature was also observed in CH_4 production.
- Production of CH_4 was high in *Salix* soils and *Nymphea* & *Eleocharis* soils, again suggesting the importance of nutrients and labile organic matter in decomposition under aerobic and anaerobic conditions.

These combined studies indicate that organic soils in the BCMCA are subject to impact with respect to water level drawdown. Carbon dioxide flux studies suggest that the surface 10 cm of soil is the most reactive and requires protection from subsidence due to oxidation when water levels are low. Soils in the BCMCA are also characteristically variable with respect to soil lability, dominant vegetation type, and site nutrient status, therefore responses to subsidence inducing low water events is site dependant. Microbially mediated oxidation is the primary driver of organic soil subsidence, and while shrinkage and compaction due to dewatering can have an effect, the long term losses of organic carbon due to oxidation are the most critical threat. This study provides evidence that suggests any drawdown even resulting in water levels below the soil surface can result in increased soil organic matter losses to oxidation and that these losses will be variable across the landscape given variation in soil nutrient availability and organic matter quality.

CHAPTER 1

REVIEW OF LITERATURE RELEVANT TO SOIL SUBSIDENCE

1.1 Introduction

Background and history of subsidence

Pressure to reclaim peat soils for agriculture has been constant throughout history. Once organic soils have been drained, the irreversible process of subsidence starts (Stephens et al.,1984), which can only be blocked by re-saturating the peat (Salmah et al., 1994). Estimates of global peatland area coverage range from 388 to 408 million ha (Maltby and Immirzi, 1993) to more than 420 million ha worldwide (Clymo, 1987), of which 36 million ha are found in the tropics and subtropics (Andriesse, 1988) an estimate which Maltby and Immirzi (1993) recently increased to 49 million ha. Population pressure and the relative agricultural value of peat soils have led to substantial development of these areas (Wösten et al., 1997). Globally an estimated 25 million ha have been drained and developed for forestry or crop production (Armentano and Verhoeven, 1988). For example, the Everglades were developed for agricultural prospecting in the early 1900's. Subsequently in the 1950's, extensive water control structures were implemented that permanently altered hydrology and accelerated soil subsidence. The first subsidence transect was established in 1913 (Shih et al., 1978; Stephens, 1984) and thus began the measurement of soil loss to subsidence in south Florida. The overall loss of peat soils in this area has been estimated from 1.4 cm yr^{-1} to 2.54 cm yr^{-1} (Shih et al., 1997, 1988). Overall subsidence rates in New Zealand where found to be 3.4 cm yr^{-1} (Schipper and McLeod, 2002). Other examples of subsidence in reclaimed peat soils can be found (Figure 1) in the

Netherlands (Schorthorst, 1977), Malaysia (Wösten et al., 1997), Canada (Silins and Rothwell, 1998).

Soil subsidence on agricultural lands greatly increases the potential for flooding and increases drainage costs (Rojstaczer and Deverel, 1995). Furthermore, on a global scale, subsidence as a result of oxidation can contribute significantly to the CO_2 levels in the atmosphere (Armentano, 1980, Maltby and Immirzi, 1993, Rojstaczer and Deverel, 1993, Berglund, 1995).

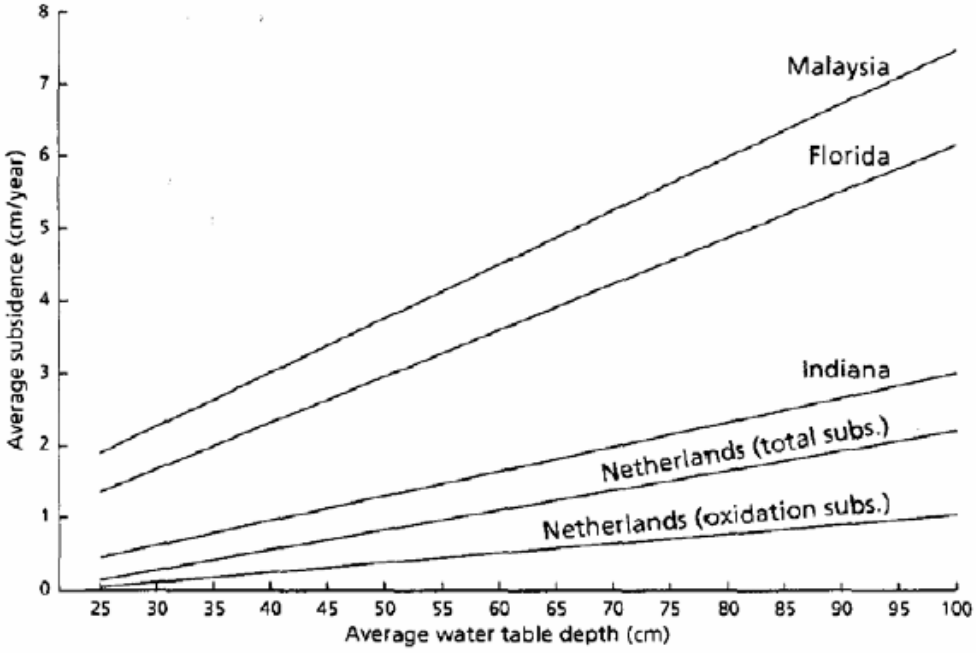

Figure 1.1 Subsidence as a function of water table depth in different geographic areas (figure from Wösten et al. 2007).

1.2 Subsidence Processes

Subsidence has been defined as the lowering of surface elevation after drainage due to causes other than erosion (Jongedyk et al., 1950). Wösten et al., (1997) defined subsidence as the total sum lowering of the soil surface. Rates of subsidence tend to vary strongly and depend on a variety of factors, such as type of peat, rate of decomposition, density and thickness of peat layers, drainage depth and climate (Schothorst, 1977, 1982; Stewart, 1994). The processes that result in subsidence can usually be divided into three categories (Parent et al., 1982; Wösten et al., 1997):

1.2.1 Consolidation. Compression of the permanently staturated peat layers situated below the groundwater level. Parent et al. (1982) estimate this process to be active over the first 5 to 10 years. Factors that affect this process are drainage depth, peat thickness and density (Nesternko, 1976; Ilniki, 1977). Schothorst (1977) was able to determine that for peat soils in the Netherlands, 35 % of the subsidence could be attributed to consolidation whilst the remaining 65 % was a result of the oxidation and shrinkage.

1.2.2 Oxidation. Introduction of oxygen in the peat layer resulting in the physical loss of organic matter due to increased rates of decomposition. This is often estimated using decomposition models, Volk (1972) reports that the microbial oxidation of peat soils can contribute from 58 to 73 % of the total subsidence. Stephens and Speir (1970) estimated that 75 % of long-term subsidence is due to organic matter oxidation.

1.2.3 Shrinkage. Reduction in the peat volume due to desiccation.

The relative contributions of each of the processes are a function of water management (Tan and Ambak, 1989). Total subsidence may be better described as a series of phases. There is an initial period in which the soil consolidates, after which there is a significantly slower oxidation and shrinkage phase.

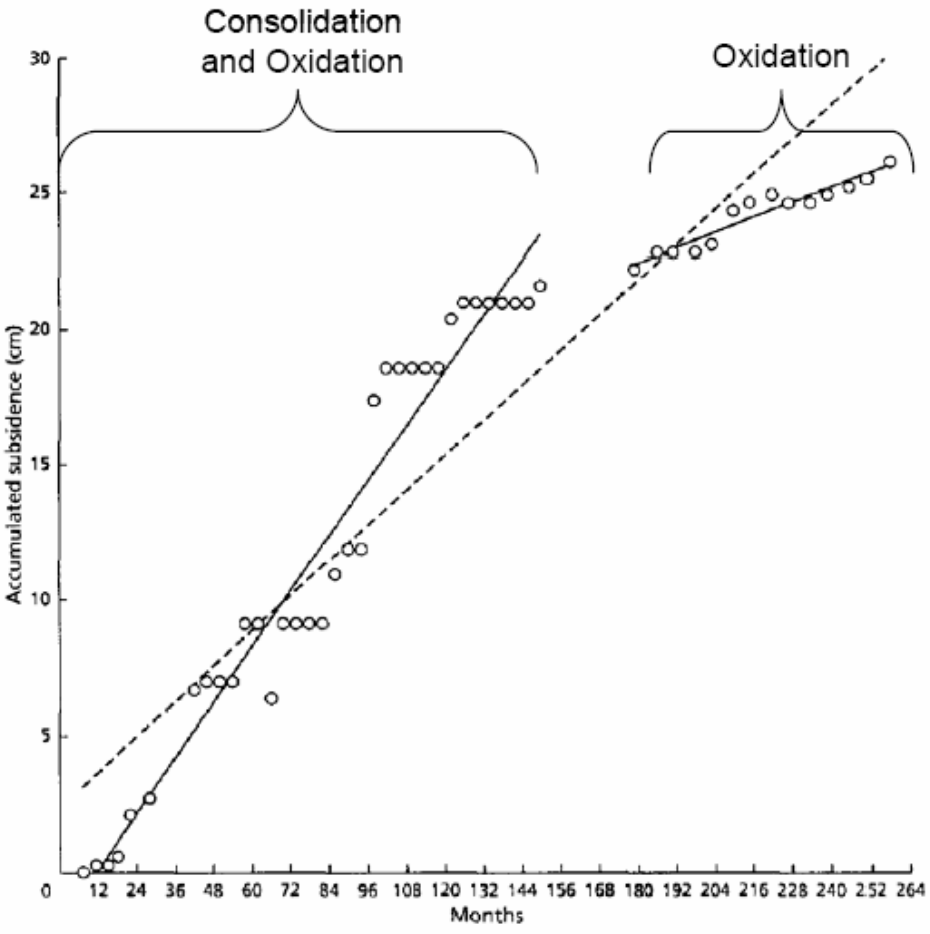

Figure 1.2. Accumulated subsidence data from Malay soils identifying the different phases of subsidence, shrinkage is assumed to occur primarily in the second phase, consolidation is assumed to predominate in the first phase (figure from Wösten et al. 2007, modified).

Wösten et al, (2007) used data from Stephens and Stewart (1976) and Schothorst (1977) to describe subsidence rates as a function of groundwater levels (Fig 1.1). Climate related differences in environmental parameters such as soil temperature and seasonal periodicity result in the differences seen between the Florida and Malay soils described in those studies and those from Indiana and the Netherlands.

Other factors that can influence subsidence other than the water table are fire (Maltby and Immirzi, 1993) and wind erosion (Parent et al., 1982). Both factors will result in significantly more subsidence than would be predicted by water table levels alone.

1.3. The physics of peat subsidence

A direct result of a drawdown is loss of buoyancy. For each cm drop in groundwater level, the overburden pressure of the overlying soil layer increases 0.01 kPa (Wörsten et al, 1997). Consolidation can be calculated using the change in bulk density. Changes in bulk density can be approached using x= 0.005665*y-3.52 in which x is the depth below ground level and y is the bulk density (Salmah et al., 1994). Subsidence as a result of consolidation at the beginning of the process can be as high as 20-50 cm per year (Welch and Nor, 1989). Armentano and Menges (1986) estimated that for New Zealand soils under pasture, these initial rates could be as high as 20 cm yr^{-1}. Wösten et al. (2003) found that the initial subsidence rates ranged between 4.6 cm yr^{-1} and 2 cm yr^{-1}, they judged the lower bound to a better estimate for their area of study (Malaysia).

Drainage of peat soils results in reduction of macropores, increased bulk density, and reduced permeability for air and water (Eggelsman, 1972). After seven years of draining a Canadian peatland, Silins and Rothwell (1998) found that the mean bulk density increased by > 60 % and this was primarily associated with the loss of pores > 600 μm in diameter in a peatland drained for forestry in Alberta. At > 0.5 MPa water potential, a saprist organic soil may have 50% more pore volume when compared to a mineral soil (Farnham and Finney, 1965). Silins and Rothwell (1998) suggest in their discussion that the subsidence of surface layers is associated with the collapse of these macropores. Silins and Rothwell (1998) postulated that the changes in soil pore distribution and bulk

density are not likely to be reversible. Drainage of peat soils has also been shown to affect the cation exchange capacity, which increased 32-38 % over a 27-year period, and base saturation, which decreased 4 % over the same period (Braekke, 1987).

1.4. The biogeochemistry of peat subsidence

Hydrology, specifically the presence, quantity, quality and timing of water controls many wetland characteristics. The hydrological regime affects the predominant vegetative communities, generates most of the hydric soil characteristics and attracts wildlife characteristic to wetland ecosystems. The hydrological regime prevalent in a system determines the soil moisture, oxygen content, pH and redox potential. Oscillations in the hydrology can effect soil accumulation, subsidence rates and nutrient availability (Schothorst, 1977;

Reddy et al., 1990; Newman and Pietro, 2001). Hydrology (both water level and soil moisture), soil quality and chemical redox state are interrelated in wetlands in a pattern of mutually dependent plant and microbial communities in which all of these factors play a significant role in modulating subsidence of organic soils (Chanway et al., 1991). However, research in the Sacramento-San Joaquin Delta, another area undergoing soil subsidence (Stephens et al., 1984), indicates that under drained conditions, soil moisture has little effect on the oxidation rates and temperature is the primary controller (Rojstaczer and Deverel, 1995).

Cycles of water level drawdown/reflood occur naturally in wetlands, introducing oscillations of soil oxygenation. A sustained drawdown in wetlands can result in soil organic matter mineralization and subsequent nutrient releases from the oxidized soils and sediments (De Groot & Van Wijck, 1993; Qui and McComb, 1994, 1995; Baldwin, 1996; Olila et al., 1997; Mitchell and Baldwin, 1998; Fisher and Reddy, 2001). Organic matter decomposition and nutrient regeneration has been summarized as a function of i) substrate quality (Fenchel and Jorgensen, 1977; Godshalk and Wetzel, 1978) (ii) hydrology due to its control on the supply of electron acceptors (e.g., O_2, NO_3^-, SO_4^{2-})(DeBusk and Reddy, 1998; McLatchey and Reddy, 1998) and iii) environmental factors such as pH and temperature (D'Angelo and Reddy, 1994; Wright and Reddy, 2001). The release of nutrients is therefore a function of the microbial group activities and substrate composition (Fenchel and Jorgensen, 1977; McLatchy and Reddy, 1998). Decomposition rates under aerobic conditions has been shown to be significantly higher than under anaerobic conditions (McLatchey and Reddy, 1998; Wright and Reddy, 2001), DeBusk and Reddy (1998)

found that anaerobic mineralization of wetland soils occurs at 1/3 the rate of aerobic soils.

The aerobic versus anaerobic respiratory activities in this study were not initially significantly different; only towards the end of the experimental period did the anaerobic respiratory levels drop to about 1/3 of the aerobic decomposition rates, presumably as a result of high nitrate levels initially present in the floodwaters. Freeman et al., (1996) suggested that the enhanced decomposition following draw downs may also be due to the reactivation of the extracellular enzymes that are responsible for organic matter decomposition.

The intermittent flooding and draining of wetland soils results in considerable temporal variability in soil redox potentials, the spatial variability is further compounded by rhizosphere oxygenation by wetland macrophytes (Lorenzen et al., 2001). Due to high environmental variability in redox state, wetland soils therefore are relatively rich in functional microbial communities that are capable of utilizing a large range of electron acceptors, such as O_2, NO_3^-, Fe(III), SO_4^{2-} and CO_2 (McLatchey and Reddy,1998; Wright and Reddy, 2001). The absence of O_2 in saturated soil conditions and the presence of alternate electron acceptors is the main control on the rates of microbially mediated organic matter mineralization (McLatchey and Reddy, 1998). Furthermore, the redox status of a soil has a significant effect on the N cycling (Reddy and Patrick, 1975), effecting not only the rates on organic N mineralization (White and Reddy, 2001) but the also nitrification and denitrification processes. Release of organic N can enhance organic matter lability by providing a source of N that subsequently enhances decomposition.

The effect of redox potential on organic matter mineralization has been documented in soil slurries (McLatchey andReddy, 1998; Wright and Reddy, 2001). Nutrient release experiments in marsh and lake sediments have been done both on soil slurries (Moore and Reddy,1994) and with intact cores (Holdren and Armstrong, 1980; Olila et al., 1997), in which the depth of the overlying water is controlled. Martin et al. (1997) conducted an experiment in which peat columns where drained to 0, 10, 20 and 35 cm depth and the leachate collected at set intervals 30 d over 5 mo. The water depth treatments did not influence total organic carbon or NH_4^+ losses from the columns. They did, however, find a significant treatment effect in NO_3^{2-} and phosphorus release from the soil columns. Drying and rewetting processes are expected to generate a sequence of alterations to microbial communities due to changes in electron acceptor availability (Tate, 1980a,b), concurrent with and affecting the changing soil properties as different microbial communities vary in their ability to utilize substrates. Morris et al. (2004) conducted an experiment in which temporally varying water table depths were related to soil organic matter oxidation. They found that 1 week of soil saturation followed by 14-day drainage to 50 cm did not affect the potential for organic matter oxidation when contrasted to a fully drained control (measured as $^{14}CO_2$ production), oxidation was decreased further if the soil was only allowed to drain 33 cm deep. Minimal oxidation occurred at a water table at 16 cm depth (equivalent potential to fully flooded).

The important controls on organic matter decomposition and C mineralization are soil temperature, plant community structure, position of redox boundaries associated with the water table, and the chemical composition of

detrital plant tissue and peat (Bubier et al., 1993, 1995; Whiting and Chanton, 1993; Yavitt et al., 1997). In a spatial explicit survey of the Sacramento-San Joaquin Delta, Rojstaczer and Deverel (1995) found that organic matter content of the soil was the single factor that best predicted subsidence levels. The higher the organic matter content, the higher the levels of subsidence. Soil texture and compositional qualities can also be affected by significant drawdowns, for example transforming organic forms of carbon, phosphorus and nitrogen into inorganic forms. The enhanced rates of organic matter mineralization results in the release of phosphorus (Reddy and Rao, 1983; Olila et al., 1997; Pant and Reddy, 2001) and nitrogen (Newman and Pietro, 2001) into the water column upon reflooding. Carbon mineralization under drained conditions generally results in the release of CO_2 and as noted before, does not seem to affect DOC (Martin et al., 1997).

Microbial activities (aerobic respiration) and extracellular enzyme activities (sulfatase, β–glucosidase and phosphatase) were significantly stimulated by an experimental drawdown of a peatland (Freeman, 1996). Drawdown has also been shown to suppress microbial activity in certain biofilms in a peatland (Freeman et al., 1995). Further, the effect of drawdown and reflooding events on extracellular enzymes is uncertain (Shackle et al., 2000). One particularly important component of biochemical oxidation of peatlands involves the aerobic breakdown of phenolic complexes (Pind et al. 1994, Freeman et al. 1996). The degradation of phenolic compounds occurs enzymatically through phenol oxidases in the presence of oxygen. Tate (1979, 1980a) found that flooding of muck soils resulted in significant inhibition of salicylate oxidation, an aromatic ring structure that requires phenol oxidase. Concomitant

to the inhibition in salicylate oxidation activity, he found a reduction in general microbial activity. Generally, as a result of flooding, microbial respiration of aromatic ring structures became insignificant while anaerobic and facultative metabolism of carbohydrates and amino acids became more important (Tate, 1980 a,b). Evidence suggests that controlling factors in phenol oxidase activity include oxygenation, temperature, pH (Ladd 1978), and plant species composition (Kuprevich and Shcherbakova 1971, Tate, 1980b, Duxbury and Tate 1981). In some cases, authors found phenol oxidase activity to be insensitive to changes in water table (Freeman et al. 1996), specifically in peatlands that are suboptimal for this enzyme because pH is too low (Pind et al. 1994). Williams et al. (2000) found no response in phenol oxidase activity in response to a summer drought in *Sphagnum* and *Carex* dominated wetlands (water table at 10 and 21 cm's respectively over an 81 day period).

Research into the relationships between water table depths and subsidence has primarily been focused on optimizing agricultural production on organic soils whilst minimizing subsidence. Usually the type of crop requires a certain degree of oxygenation in the root zone. Okruszko (1989) found that for a fibric and hemic peat, a water table depth of 96 and 130 cm respectively, were the critical depths to ensure little subsidence during growing season.
McAfee (1989) recommended a drainage depth of 40 cm for a fen peat soil to ensure crop growth and minimize subsidence, which is similar to the results that Breglund (1995) obtained for hemic (fen) soils. The more fibric soils (moss peat) required a deeper water table to ensure aerobic conditions in the root zone (60-70 cm depth).

1.5. Modeling Subsidence

Wösten et al. (2003) found that linear subsidence models were poor predictors of subsidence over time (over 21 years). However, regression has been a tool used to approximate the rates of subsidence once the initial consolidation phase is no longer as influential (Shih et al., 1979). Other, comprehensive models used in subsidence have generally been an approximation to the exponential model (Mathur and Lévesque, 1977), in which ultimately long term subsidence is reduced by water table limits. A more complex model was developed at the Wageningen Agricultural University and models the three subsidence components individually (The Peat OXidation and Permanent Shrinkage - POXAPS; Doels, 1995) and was applied successfully by Wösten et al. (2003) on subsidence data from Malaysia.

Finally, the most complex modeling effort encountered was a systems model by Browder and Volk (1978) of carbon transformations in soil subsidence. The model employs circuit and energy flow concepts (Odum, 1971) to describe organic matter decomposition to CO_2. Their model consists of two components, a biotic component and an abiotic component and represents a relatively diligent overview of the processes associated with peat subsidence. It does, however, assume that all subsidence is the result of biological oxidation, in which the water table has the most significant role in controlling subsidence.

1.6. **Subsidence and natural wetland systems**

Urban development in Florida and elsewhere will focus increasing pressure on natural aquatic ecosystems to meet water supply demands. Increasingly water resources are becoming an important commodity and supplies have to be allocated amongst many users. As a result, the pressure on aquatic ecosystems will only increase in the near future (Haakh, 2002). Inversely, increasing recognition of the importance of wetland functions has led to many restoration efforts on previously developed agricultural lands (for example, the Everglades Nutrient Removal Project – ENRP; Martin et al., 1997, the Mississippi delta; Templet and Meyer-Arendt, 1988 or the Sacramento-San Joaquin Delta, Rojstaczer and Deverel (1995). Restoration and maintenance of aquatic ecosystems highlight the need for careful management and allocation of natural waters for municipal and agricultural use.

Hydrology is a dominant factor controlling wetland plant community development and structure. This is exemplified in the zonation of adult plants, seeds, and seedlings in a number of different wetland types, including tidal freshwater marshes (Davis 1940, Whigham et al. 1979, Simpson et al. 1983, Mitsch and Gosselink 2000). Duration, frequency, and depth of inundation can also influence the species composition of wetland vegetation (Baldwin et al. 2001). Casanova and Brock (2000) found, in a study where seed bank samples from an intermittently flooded wetland were subjected to 17 hydrologic regime treatments varying in water depth and frequency and duration of flooding, that continuous flooding yielded the lowest species richness (about 6/sample), nonflooded (but moist) conditions produced the greatest richness (about 18/sample), and intermittently flooded treatments produced intermediate richness. Johnson *et al.* (2000) found

that lowest species richness occurred in a continuously flooded treatment, highest richness in a continuously nonflooded (but moist) treatment, and intermediate richness in treatments of flooded followed by nonflooded and viceversa. Because hydrology, or the lack there of, dives subsidence, the effects of drawdown events and the subsequent change of soil elevation can alter plant community structure by changing the physical soil elevation. This change in soil elevation, when reflooded, alters the duration and depth of inundation, which in turn can influence plant communities by exceeding tolerance zones or thresholds of previously established vegetation.

Because wetland vegetation and thus portions of ecological functions derived from these wetlands are evolved from hydrologic regimes, alterations to these regimes result in changes to functional and ecological value of impacted systems. The water requirements of wetland systems therefore do not only respond to soil loss processes, but are also as a function of the wetland communities desired in these systems. In terms of maintaining certain wetland ecosystem structures, timing, as well the overall quantity of water, is critical to the system. Whilst the physical and biogeochemical processes that occur in organic soils as a result of drainage are well understood and documented, we found that a great majority of these studies where executed in soils developed for agriculture with the purpose of maintaining ideal conditions for a particular crop yet minimizing subsidence due to oxidation (e.g., Berglund, 1995; Morris et al. 2004). The effect of a drawdown or drainage in natural systems is not well understood nor is the relationship between critical water table depth and drawdown duration to soil subsidence in natural systems. What does increasingly seem to be critical, both in natural systems as in the studies done on agricultural soils, is not only mean annual water table depth, but

timing of watertable changes. This in turn requires management strategies to focus on the priority usage or maintenance of these aquatic systems and associated organic soils with respect to water level management and delivery.

CHAPTER 2.

CHARACTERIZATION OF SOIL ORGANIC MATTER INTO LABILE AND REFRACTORY POOLS

2.1 Introduction

Organic soils are generally made up of detrital materials from dominant vegetation types deposited over long periods of time. Plant tissues are often similar at the coarse scale in that the main constituents of all plants are hemicellulose, cellulose and lignin. These are the main carbohydrate building blocks of terrestrial plants and differ only in their complexity of structure, all having the same atomic building blocks of $C_6H_{12}O_6$ oriented in a six carbon ring formation indicative of most sugars. Hemicellulose is the precursor to cellulose and is different from cellulose in its structural bonding. As plant biosynthesis continues, hemicellulose molecules are cross-linked through bonding structures that increase its structural integrity and begin to closely resemble cellulose (the most prominent bio-polymer on earth). This process continues in plants that require more rigid structural support in that cellulose is further cross linked and condensed to form lignin, a bio-polymer that is very recalcitrant to microbial degradation due to its molecular structural complexity.

These plant tissue constituents are deposited as detritus and are subsequently subjected to long term microbial decomposition at the soil surface. Over time, the relative amounts of theses fractions change from those found in plant tissues, due to the variable susceptibility of each to microbial decomposition processes. The lability of these fractions is highest in hemicellulose and lowest in lignin, with cellulose being intermediate. For this reason, it is expected that organic soils will be found to have

relatively high amounts of recalcitrant materials (lignin) and lower amounts of the more readily decomposable materials such as hemicellulose and cellulose.

To determine the lability of the organic matter in Blue Cypress Marsh Conservation Area (BCMCA) soils, the upper 10 cm of cores from five sites with different nutrient impact status and different dominant vegetation (Table 2.1) were analyzed for labile, moderately-labile, and refractory fractions of carbon. Labile fractions are defined as all water extractable organic carbon fractions. These include sugars, starches, organic acids, proteins, fatty acids, and waxes. Moderately-labile and refractory fractions of organic carbon include hemicelluloses and cellulose, and lignin respectively. Determination of these fractions, in addition to other physical and chemical parameters used in soil characterization, is helpful in understanding the extent of decomposition these organic soils may undergo during aerobic and anaerobic decomposition. Similar techniques have been used to estimate soil organic matter lability in the Northern Everglades (DeBusk and Reddy, 1998; Osborne, 2005). This analysis is adapted from a forage and feed analysis used by Van Soest (1970) to estimate plant tissue fiber fractions and although the estimation is coarse, it is suitable for this work because it is inexpensive (relative to C^{13} NMR) and is used as a semi-quantitative descriptor of organic matter for comparison of organic matter decomposition rates.

2.2 Materials and Methods

2.2.1 Study sites

The BCMCA has many different vegetation communities and for this analysis, five dominant vegetation communities with differing levels of nutrient impact were chosen to represent the majority of BCMCA soils. These sites included dominant vegetation types such as, *Typha domingensis*, *Cladium jamaicense*, *Panicum hemitomen*, *Nymphea odorata, and Salix caroliniana.* These vegetation types correspond to various ecotypes and nutrient enrichment impacts as detailed in Table 2. 1.

Table 2.1. Dominant vegetation and site characteristics of stations selected to represent the major plant communities in BCMCA.

Station	Location Lat/Lon		Vegetation Type	Ecotype	Impact Status
A	N27.69743	W80.67693	*Typha*	marsh	high
B	N27.67153	W80.68356	*Nymphea*	slough	low
C	N27.69515	W80.73538	*Cladium*	ridge	low
D	N27.69639	W80.72811	*Panicum*	wet prairie	low
E	N27.66918	W80.67660	*Salix*	swamp	high

2.2.2 Soil Sampling

Soil sampling took place on June 21, 2005 via airboat in a combined effort of Wetland Biogeochemistry Laboratory and St. Johns River Water Management District staff. Intact soil cores were taken in the field at locations listed previously with a stainless steel coring head attached to 10 cm ID PVC pipe sections 50 cm long. Five replicate cores were taken from each site in close proximity to one another, core tubes were sealed on both ends for transportation with expandable 10 cm diameter hydraulic test plugs. Care was taken to ensure that a minimum of 20 cm of site water was

maintained in the intact cores as they were transported back to the Wetland Biogeochemistry Laboratory for analysis. Upon arrival, cores were maintained at 4ºC until processed. Large detritus and live vegetation were removed from the surface of the cores before they were sectioned and analyzed for physical and chemical characteristics

2.2.3 Soil Characterization

Soil moisture content was determined by drying a subsample of soil at 70º C for three days. This soil moisture content was then used to determine bulk density based upon initial weight of the 20 cm depth core section corrected for water content. Chemical parameters were measured using standard methods in the laboratory. Briefly, total carbon (TC) and total nitrogen (TN) were determined by Flash EA 1112 series C/N analyzer (Thermo Electron Corp.), total phosphorus (TP) was determined by ashing, HCl digestion and analysis on a Bran Luebbe AA3 digital colorimeter. Soil pH was measured in a 2 to 1 slurry of water to soil with an Accumet series 50 pH meter. Loss on ignition (LOI) was determined by loss of mass after combustion at 500º C in a muffle furnace for 4 hours.

2.2.4 Fiber fractionation analysis

Soluble soil constituents (lipids, waxes, proteins, etc.), hemicellulose, cellulose, and lignin fractions were quantified by a modified sequential fiber extraction method (Ankom Technology, Fairport, NY) modified from the feed and forage analysis by Van Soest (1970) and later by Ball et al (2001). This method has been semi-automated by employing an ANKOM Total Fiber Analyzer 200 which consists of a contained sample carousel with adjustable heating and agitation. This methodology has been shown to be repeatable and consistent for vegetation analysis (Roberts and Rowland 1998). This

analysis required the use of a sequential extraction process and calculation of fiber fractions by mass loss after each extraction. Briefly, approximately 500 mg of a sample of ground soil (sieved to1mm) was placed in a permeable HDPE envelope and heat sealed, it was then extracted with a neutral detergent solution (30 g sodium lauryl sulfate, USP; 18.61 g ethylenediaminetetraacetic disodium salt, dehydrate; 6.81 g sodium tetraborate decahydrate; 4.56 g sodium phosphate dibasic, anhydrous; and 10 ml triethylene glycol in 1 L distilled water) at 100ºC with agitation, 20 g sodium sulfite, and in the presence (4 ml per 2 L detergent solution) of heat stable alpha amylase. After 75 min. of washing, samples were removed and washed 3 times with boiling water, rinsed with HPLC grade acetone for 3 minutes and dried again at 55º C. After drying, samples were weighed again and mass loss was calculated. This mass represents the soluble soil fraction (waxes, lipids, proteins, and other cellular constituents) which is equivalent to the neutral detergent soluble fraction described in the forage analysis. To determine hemicellulose content, the samples were further extracted with an acid detergent solution (30 g cetyl-trimethylammonium bromide 1 L 1 N sulfuric acid) for 1 hour at 100º C. The samples were then washed with boiling water and acetone and dried again for reweighing as described previously. The difference in mass was calculated as the loss of hemicellulose (the ADF fraction of the forage analysis). To determine cellulose, the samples were digested in 24 N sulfuric acid for 3 hours at room temperature with agitation every half hour. Samples were again washed and dried and reweighed. The mass loss was calculated as cellulose content, and the remaining material calculated as lignin and ash. Samples were subsequently combusted at 550º C for 4 hours to determine mineral content. All fiber fractions were then normalized for ash content and are hence

expressed as percent of total soil organic matter (%SOM). A complete description of materials and equipment for this method can be found at the Ankom company website: http://www.ankom.com.

As mentioned previously in the introduction of this chapter, the analysis employed here represents a coarse estimation of fiber fractions. The potential for over-estimation of fiber fractions exists depending upon soil particle size as the mesh bags can lose particles smaller than 250 um in size. This loss of ultra fine soil particles could potentially introduce error into the mass balance of sequential fiber analysis. Due to the nature of the comparisons being made here (semi-quantitative measure of soil organic matter quality in elation to carbon mineralization rates) and the relative cost effectiveness of this analysis, we feel that the data generated from the fiber analysis are significant and defendable for this work.

2.3 Results

2.3.1 Soil Physio-chemical Characterization

Results of the physical and chemical characterization efforts are summarized in Table 2.2. Bulk density was highest in the *Salix* and *Panicum* sites, and lowest in *Typha* and *Nymphea* sites. Soil pH was similar at all sites except the *Nymphea* site, which was

Table 2.2. Results of physical and chemical characterization of Blue Cypress Marsh soils and soil organic matter in the top 20 cm of the soil profile. All data represent analysis of a minimum of 3 replicate samples and ± indicates 1 standard deviation from the mean.

Station	Vegetation	BD $g\ cm^{-3}$	pH	TC $g\ kg^{-1}$	TN $g\ kg^{-1}$	TP $mg\ kg^{-1}$	LOI %	Soluble Soil %SOM	Hemi-cellulose %SOM	Cellulose %SOM	Lignin %SOM
A	*Typha*	0.070	5.8±0.1	461±4	28.1±0	622±30	93.2±0.4	16±1	21±2	35±3	28±5
B	*Nymphea*	0.069	5.4±0.1	478±4	33.3±1	584±34	95.6±0.5	11±2	15±4	36±4	39±7
C	*Cladium*	0.075	5.6	467±7	27.9±1	540±25	94.8±0.9	7±1	13±2	18±6	62±4
D	*Panicum*	0.084	5.8±0.1	468±2	37.2±1	538±73	93.1±0.3	8±2	9±1	25±2	58±5
E	*Salix*	0.091	5.8±0.1	481±9	29.7±1	851±60	94.6±0.6	5±3	10±3	30±5	55±5

noticeably low in comparison. Soil TC was similar at all sites, however, TN was more variable (range = 27.9-37.2) with the *Panicum* site being highest and the *Cladium* site the lowest. Total phosphorus (TP), as expected, was found to be highest in the *Salix* and *Typha* sites and relatively lower in the interior sites. Ash contents were similar among sites ranging from 4.4 to 6.9%.

2.3.3 Fiber fractionation of BCMCA soils

Results of fiber analysis conducted on samples of five soils from sites of different dominant vegetation within the BCMCA reveal differences in the soil fractions of lignin, cellulose, hemicellulose, and soluble soil constituents (Figure 2.1).

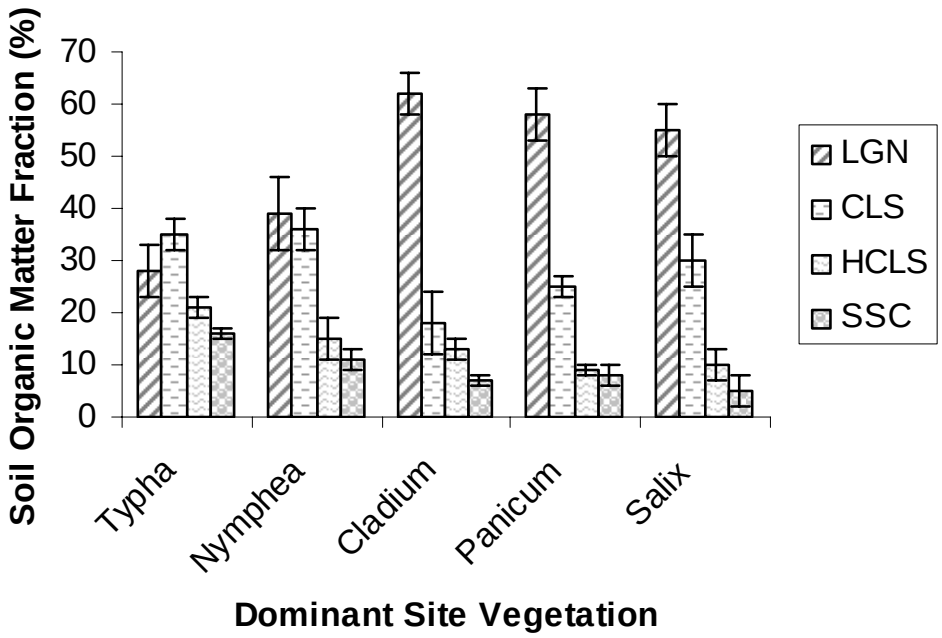

Figure 2.1. Percent of soil organic matter within each of the 4 fiber fractions described as lignin (LGN), cellulose (CLS), hemicellulose (HCLS), and soluble soil constituents (SSC) for each of the five sites defined by different dominant vegetation types. Soil organic matter fraction is presented as percent of total soil organic matter (corrected for ash content). Sampling stations are related to vegetation types by the following: Station A = *Typha*, B= *Nymphea*, C= *Cladium*, D= *Panicum*, and E= *Salix*.

These constituents appear to vary considerably between sites with different dominant vegetation types. Lignin content is highest in the *Cladium*, *Panicum* and *Salix* sites, indicating a high proportion of soil from these sites that is recalcitrant to microbial decomposition. *Typha* and *Nymphea* sites show the lowest levels of lignin respectively. Cellulose content is lowest in *Cladium* and *Panicum*, and highest in *Typha* and *Nymphea*, with *Salix* being intermediate. Hemicellulose fractions are less variable across the five vegetation types, but again, *Typha* shows the highest value, while the other sites are more similar to each other. The most labile fraction, the soluble soil constituents, are again highest in *Typha*, and *Nymphea*, the other species being significantly lower in this labile fraction, but similar to each other.

When the fiber fractionation scheme is utilized to determine pools of labile, moderately labile, and refractory organic matter, the soluble soil constituents are referred to as the most labile pool. Hemicellulose and cellulose are grouped together to make up the moderately labile pool, and the lignin and lignin-like residuals of the fiber fractionation are considered to be refractory to microbial degradation (Figure 2.2).

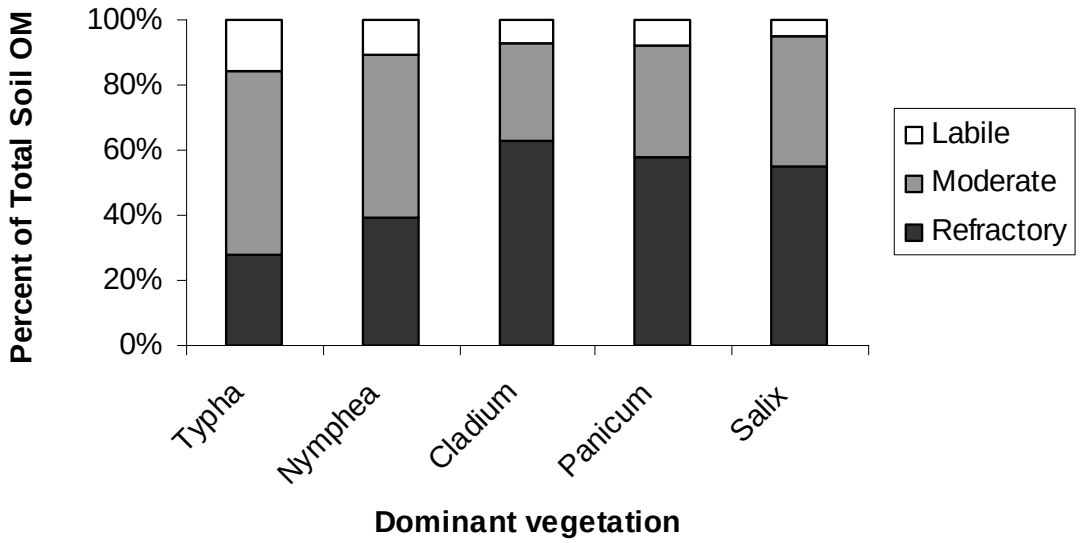

Figure 2.2. Fractionation of BCMCA soil organic matter into labile (soluble soil constituents such as sugars and proteins), moderately labile (cellulose and hemicellulose) and refractory pools (lignin and residuals).

Results of this grouping of BCMCA soil organic matter into labile, moderately labile and refractory pools indicates the highest amount of labile carbon to be found in soils from the *Typha* dominated site, with the slough site dominated by Nymphea to be similar in content. The *Cladium* and *Panicum* dominated sites show similar amounts of labile organic materials, less than *Typha* and *Nymphea* sites, but more than the *Salix* dominated site. A similar pattern was observed when cellulose and hemicellulose are combined to make the moderately labile pool. *Typha* and *Nymphea* sites have the largest pools of this material. However, *Cladium* has the lowest amount of this material with *Salix* and *Panicum* being intermediate and similar in value. The refractory pools were found to be highest in the *Cladium* site followed by *Panicum* and *Salix* respectively. *Typha* and *Nymphea* had the lowest values of refractory organic matter.

2.4 Discussion

2.4.1 Soil physcio-chemical characterization

Analysis of soil physical properties such as bulk density suggest that the *Typha* and *Nymphea* sites are both lower in density than the other soils. This is likely due in part to the nature of the plant material that is building the SOM pool. Further, the highest bulk densities observed (*Salix* and *Panicum* sites respectively) suggest that the areas with dense *Salix* and the wet prairies dominated by *Panicum* may have experienced some sort of drawdown induced compaction in the recent past.

Soil nutrient analysis suggests that all soils are similar in their carbon content, however, nitrogen is variable. It was expected that soils from the stations established in nutrient impacted areas would be higher in TP and TN. This was the case for TP but not TN. Higher microbial activity associated with higher levels of TP (a limiting nutrient) may explain the lower levels of nitrogen available as denitrification may be a significant process in the presence of elevated microbial activity. The relationship between this site, total N levels, and denitrification is not clear. The highest nitrogen levels were observed in the *Panicum* site, a wet prairie habitat, which by its nature has a shorter inundation frequency than the other interior sites. Therefore, denitrification may not be as significant at this site. The possibility of variability in plant nitrogen requirements and physiological allocation may be the cause of lower observed TN in these soils. Finally, TP levels were observed to be highest in the sites chosen to represent nutrient enriched areas of the BCMCA. These two sites have very different vegetation, *Typha*, a rapidly growing, deep water suited macrophyte and *Salix*, a woody species which grows more slowly and requires periods of drawdown to become establish. Although both are

nutrient impacted sites, the differences in hydrologic requirements for successful recruitment may be the determining factor in the differences observed in soil nutrient content.

2.4.2 Soil organic matter characterization

Analysis of BCMCA soils from sites with different dominant vegetation and thus different types of organic matter contributing to organic soil accretion, suggest that vegetation type is significant in determining the relative amounts of refractory and labile organic matter in the surface soils. Based on these results, soil organic matter from these sites could potentially react differently to decomposition and subsidence events.

Sites within BCMCA that were considered to be unimpacted by nutrient enrichment (*Cladium*, *Nymphea*, and *Panicum*) were also considered to be representative of a majority of soils in BCMCA. These soils are similar in many respects, however, the *Nymphea* sites, representative of slough ecotypes, appear to be made up of more labile organic material. This is likely a result of the nature of *Nymphea* and *Eleocharis* (another slough species) plant tissue being low in structural compounds such as lignin and thus deposition of highly refractory organic matter is decreased. There is support for this in the literature, as Osborne (2005) characterized *Nymphea*, *Eleocharis*, *Typha*, *Cladium* and *Panicum*, and reported that *Nymphea* and *Eleocharis* tissues were significantly lower in lignin and refractory compounds and higher in soluble constituents. This would suggest that the decomposition potential of slough soils would be greater than neighboring *Cladium* ridges and *Panicum* prairies. This is supported again by findings of Osborne (2005) and Lewis (2005). Further, *Cladium* and *Panicum* were shown to have higher fractions of both lignin and cellulose than that of *Nymphea* and *Eleocharis*. This

suggests that the degradation potential of soils derived from these species would be more difficult for microbial communities to decompose in comparison to slough species, given that nutrient status was not a factor. These sites were considered to be unimpacted by nutrient enrichment as they were found at the interior of BCMCA while the *Typha* and *Salix* sites were on the perimeter close to water control structures where nutrient enrichment is highest.

Areas of nutrient enrichment have been shown to have increased rates of soil and detrital decomposition and mineralization of organic carbon in the nearby northern Everglades (Davis, 1991; DeBusk and Reddy, 1998; DeBusk et al., 2001; DeBusk and Reddy, 2003). These areas of nutrient enrichment in the northern Everglades are also closely associated with the encroachment of Typha as a dominant vegetation type (Davis, 1991; Newman et al., 1996; Newman et al., 1998) and are closely linked to phosphorus enrichment. Studies by DeBusk and Reddy (2003) reported changes in soil and detrital labile and refractory pools as well as increased decomposition activity in the presence of nutrient enrichment. The application to this study is that while *Typha* has been reported to be rich in labile compounds and relatively low in refractory organics (Osborne, 2005), the decomposition rates could be higher for *Typha* due to this abundance of more labile material but also because the sites of *Typha* dominance are nutrient enriched. *Salix*, a woody species, appears to have similar levels of refractory compounds as the prairie species (*Cladium* and *Panicum*) and therefore, is expected to be relatively recalcitrant to decomposition, but Salix is also found in areas of nutrient enrichment which could cause increases in mineralization of organic matter in situations of subsidence.

2.5 Conclusions

The differences observed in labile, moderately labile, and refractory organic matter fractions in soils of BCMCA can serve as indicators of potential decomposition. If all environmental factors were equal, it is expected that *Typha* and *Nymphea* dominated regions would decompose at the highest rate, followed by *Cladium, Panicum* and *Salix* regions. High soil nutrient content of Salix dominated regions of the marsh may lead to greater decomposition rates than would be expected from vegetation types alone.. The following chapters of this report will examine the relative decomposition rates under various environmental conditions and also consider the effects of elevated nutrients on mineralization of organic matter.

CHAPTER 3

DETERMINATION OF SOIL ORGANIC MATTER LOSS RATES AS A FUNCTION OF WATER LEVEL RELATIVE TO SOIL SURFACE

3.1 Introduction

Hydrology, specifically the presence, quantity, quality and timing of water defines wetland ecosystems. Hydrology affects plant communities and generates most of the hydric soil characteristics in wetland ecosystems. Oscillations in water levels at different magnitudes, frequencies, timing, and duration are common in natural wetlands and they influence the rate of soil organic matter accretion and decomposition. Soil organic matter accumulates in a wetland system mainly because of the absence of factors that favor their decomposition. Oxidation of soil organic matter leads to C-loss from wetland promoting subsidence or loss of soil. Rate of subsidence is directly related to water table; the deeper the water table greater the rate of subsidence (Shih et al. 1997). Water level drawdown in the Everglades Agricultural Area has caused the soils to subside at a rate estimated to be 3 cm $year^{-1}$ (Stephens, 1984). This has resulted in a significant portion of the peat formed over 5000 years to be lost to biological oxidation in less than 100 years (Stephens, 1984).

Biological oxidation of organic matter refers to microbial activities in soil that enzymatically breakdown complex carbon compounds and make them available as C substrates to be utilized by microorganisms. Water logged conditions strongly influence the decomposition rates of organic matter by limiting microbial activity and restricting oxygen supply. Peatlands have been shown to accumulate organic carbon as phenolic compounds and other complex polymeric substances. Phenolics are known to be potent inhibitors of enzymes (Wetzel, 1992) and few enzymes, such as phenol oxidase, can

degrade these recalcitrant compounds. Release of carbon in these systems is controlled to a large extent by two enzymes phenol oxidase and β glucosidase (Freeman et. al, 2004). Phenol oxidase activity releases C by hydroxylation of aromatic rings (eg. lignin) ultimately leading to mineralization or humification of parent compounds. This enzyme is active under aerobic conditions, hence in water logged conditions, SOM oxidation is often limited by phenol oxidase activity. In contrast, hydrolases can remain active under anaerobic conditions (Lee et al, 1999, Nybroe et al., 1992). β-glucosidase is one of the important bacterial enzyme involved in C-cycling in soils. These enzymes hydrolyze the glucosidic bonds in large polymeric organic compounds such as cellulose (Eivazi and Tabatabai, 1988). Microorganisms without the ability to produce this enzyme depend on the glucosidase-producing organisms for their carbon sources.

Factors effecting the organic matter decomposition in wetlands include labile organic matter, high microbial numbers and enzyme activity, favorable oxidizing conditions, moisture, pH, temperature and nutrient supply. Carbon loss by microbial respiration occurs continuously in water saturated soils, but at a much lower rate than in aerobic conditions. This is partly due to reduced energy yield of other terminal electron acceptors. Several studies have shown that presence of other external terminal electron acceptors control the rates of microbially mediated organic matter mineralization (McLatchey and Reddy, 1998; D'Angelo and Reddy, 1999; DeBusk and Reddy, 2003). Enzyme substrates are other important regulators of microbial enzyme activity. Substrate quality and quantity changes with depth of soils and is reflected by the changing microbial community structure in soils.

Since hydrologic changes by increasing aeration, could enhance the phenol oxidase activity (Gorham, 1991) or not be affected by water table drawdown (Freeman et al., 1996), we investigated the relationship between water table depth and the decomposition of soil organic matter by studying the change in enzyme activity as a response to water table drawdown. Since the two important enzymes involved in the decomposition of organic matter are active under aerobic conditions and oxygen concentration declines with depth in soils (McLatchey and Reddy, 1998), we hypothesized that after a certain soil depth, there will be negligible change in enzyme activities relative to the water table drawdown. Alternately, lowering of water column in the cores would lead to soil subsidence only to a certain level immediately and then slow down. This was achieved by determining the activities of two enzymes phenol oxidase and β-glucosidase at different depths with in soil cores that were maintained at different water levels.

3.2 Material and Methods

3.2.1 Study Site

The study focused on soils within the Blue Cypress Marsh Conservation Area (BCMCA) located in Central Florida in the headwater region of the St. Johns River in Indian River County. Five soil sampling locations were established within the BCMCA to evaluate representative organic soils differing in their nutrient impact status and dominant vegetation communities. These representative sites include nutrient impacted areas dominated by *Typha domingensis* (site A) and *Salix caroliniana* (site E) as well as, non-impacted sites with vegetation communities dominated by slough vegetation such as *Nymphea odorata* and *Eleocharis interstincta* (site B). Wet prairie ecotypes are

represented with sites dominated by *Cladium jamaicense* (site C) and *Panicum hemitomon* (site D). For the purposes of studying the influence of water table changes on organic matter decomposition, soil core samples were collected from the slough site (B).

3.2.2 Soil sampling

Forty intact cores of 10 cm diameter and 50 cm depth were retrieved from the slough reference site located in the SW region of BCMA (Site B, N27.67153 W80.68356) in June 21, 2005. At the time of sampling, the water depth was approximately 50 cm. Intact cores were taken by inserting a 62 cm long x10cm ID. PVC tube fitted with a stainless steel cutting head into the soil, using a twisting motion and light downward pressure to minimize soil compaction. Cores were then extracted and capped in the core tube with 10 cm of overlying site water. The cores were transported back to the Wetland biogeochemistry Laboratory at the University of Florida for analysis and experimentation.

3.2.3 Intact core study

Intact cores were incubated in a core bath at room temperature (approximately 23ºC) for 91 days (+) at 8 different water levels, 20cm of inundation, water level at soil surface, -2cm, -5cm, -7.6cm, -15cm, -20cm, and -30cm. Four randomly chosen cores made up each treatment. Three cores served as experimental replicates for CO_2 measurement and the fourth was fitted with redox probes to monitor Eh over the course of the experiment. Redox probes were fitted at the surface of the soil and 5 cm below the respectivetreatment water levels. Platinum electrodes were used in conjunction with a Fisher Scientific Accumet AR 50 dual channel pH/ion/conductivity meter employing an Accumet 13-620-258 redox reference electrode to measure Eh throughout the

experiment. At the conclusion of experimentation and prior to destructive sampling, redox probes were removed and reconfigured to determine redox profiles in all representative cores to a depth of 10cm below the treatment water table. Over the course of the experimentation, water levels were maintained with site water initially then with distilled deionized water as needed. Cores were destructively sampled for enzyme analysis at the completion of the incubation period. Each of the 3 replicate cores in the 8 treatment sets was sectioned at 0-2cm, 2-5 cm, 5-7.6cm, 7.6-10 cm, 10-20cm, 20-30cm, and 30-40cm soil depths and stored for further analysis. These samples were also analyzed for moisture content, and microbial biomass carbon.

3.2.4 Carbon dioxide analysis

Carbon dioxide measurements were made by NaOH (0.2 M) traps (3 ml of NaOH in a 15 ml serum bottle) inserted into the core tube headspace every seven days for 24 hours. End caps of PVC were fitted over the top of core tubes to exclude gas exchange during the 24 h gas trapping and a core containing only de-ionized water with the same headspace was used as a control. After each 24 h gas collection, traps were removed, crimp capped with septa, and stored for future analysis. To analyze gas traps, 0.5 ml of 3N HCl was added to each trap to evolve trapped CO_2. Serum bottle headspace pressure was recorded and then gas sampled and analyzed on a Shimadzu GC-8 (Shimadzu,TN) gas chromatograph with a Porapak N column (6 1/8 Supelco,OR) fitted with a thermocouple detector (TCD).

3.2.5 Microbial Biomass Carbon

Microbial Biomass C (MBC) was determined by the fumigation-extraction method of Vance et al. (1987). Approximately 5 g of sample was placed into

polypropylene centrifuge tubes, 0.5 ml volume of chloroform was added, and tubes were placed into a vacuum desiccator with a beaker containing additional chloroform. Air in the desiccator was evacuated three times until the chloroform began boiling. Each time the desiccator was flushed with air. After the third evacuation, the desiccator was sealed under a vacuum for 24 h. A non-fumigated control sample set was placed on the adjacent lab bench. After 24 h, samples were removed and all samples were treated with 20 ml of 0.5 M K_2SO_4 shaken for 1 h on a longitudinal shaker and vacuum filtered through Whatman #42 filter paper. Samples were then analyzed for total organic carbon (TOC) using a Dohrman TOC analyzer (Rosemount Santa Clara, CA). Microbial biomass carbon was determined by subtracting the controls from the TOC of chloroform-treated samples. The difference in DOC of controls and fumigated samples is assumed to be microbial biomass carbon. An extraction efficiency factor of 0.37 was applied (divided), utilizing a proviso calibration for organic soils by Sparling et al., (1990). Extractable TOC was defined as the TOC from extracted non fumigated controls.

3.2.6 Enzyme analyses

Activities of β glucosidase and phenol oxidase enzymes were measured in the soil samples from the core sections collected at the conclusion of the incubation period. β glucosidase activity was measured using the fluorescent substrate methylumbelliferone3-b-glucoside (MUF-G, 500μM)(Chrost and Krambeck, 1986; Hoppe, 1993). One gram of soil sample was well homogenized by brief agitation with a Tissue Tearor model 398 (Biospec Products, Bartlesville, OK) and appropriately diluted with distilled water. In a 96 well titer plate, 50 ul of substrate was added to 200ul of sample solution and incubated at room temperature for 2 hours in dark. The reaction was stopped by adding 10ul of

glycine/NaOH buffer and fluorescence was read on Bio-Tek model FL600 (Bio-Tek Instruments Inc., Winooski, VT) set at excitation/ emission 360±40 nm/ 450 ± 40 nm. Control for each sample included negative- substrate control and negative-sample controls. Quenching curves were established for each sample to account for any fluorescence quenching by the soil samples. Enzyme activity was calculated as μmol of product g^{-1} dry soil h^{-1}. Moisture content of the soil was determined by drying the soil at 70ºC for 72 h.

Phenol oxidase activity in the soil samples were determined by a colorimetric method (Pind et al,1994). Briefly, soil slurry was prepared by diluting the soil samples and homogenizing them. Duplicate 4.5 ml aliquots of slurries were added to 4.5 ml of L-dihydroxyl -p-amine (L-DOPA, 10 mM) solution and were incubated for 1 and 3 minutes. Incubation was stopped by centrifuging the sample at 10000rpm for 5 minutes and immediately measuring the absorbance of the supernatant (filtered with GFC/C filters) at 460nm using UV-visible recording spectrophotometer model UV-160 (Shimadzu Corporation, Kyoto, Japan). The difference in the absorbance between the two time intervals was used to determine the product formation 2-carboxy-2,3-dihydroxyindole-5,6-quinone (diqc) The enzyme activity was calculated by using the Beer Lambert law and the molar absorbtivity constant of 3.7×10^4 (Mason et al. 1948).

3.2.7 Total phenolics

Total phenolics in soil samples were determined as per Box et. al (1983). Briefly, the soil samples were diluted with DI water (soil: water ratio 1:2) and shaken for 10 minutes before being centrifuged for 10 minutes at 6000rpm. The supernatant was collected and filtered through Whatman# 41. The filtrate was further filtered through

GF/C filters before being analyzed for total phenolics. To 5 ml of sample, 0.75 ml of $NaCO_3$ (200 g l^{-1}) and 1.5 ml of Folin-Ciocalteau agent was added and the solution was incubated for 1.5 hours for color development. Intensity of blue color was indicative of the phenolics in the sample and this was measured using a spectrophotometer @ 750nm. The standard curve was prepared using phenol (1- 2mg/L) (Sigma, NJ).

3.3 Results

3.3.1 Carbon dioxide flux

Evolution of CO_2 per 24 h was plotted by experimental day (every 7 days) for the 91 days of the experiment (Figure 3.1). The results are presented as carbon (ug) content of CO_2 evolved per unit area per day. Treatments with water levels below 7.6 cm tend to produce similar amounts of C as CO_2, while treatments with higher water levels tended to produce less C, but with greater difference in levels of production.

Investigation of the cumulative carbon loss per treatment over the course of the experiment reveals the lower water table treatments show similar results. These groups are the highest producers of CO_2 in comparison to the higher soil water treatments. However, the higher soil water treatments do show greater differences in C mineralization in comparison to the low water treatments (Figure 3.2). Linear regression analysis of cumulative carbon loss was employed to determine mineralization rates from each cumulative carbon loss curve (Table 3.1). These rates indicate that as water table decreases, carbon mineralization increases. However, treatments such as -15 cm, -20 cm, and -30 cm water tables do not exhibit a difference in rate comparable in magnitude with the difference of depth.

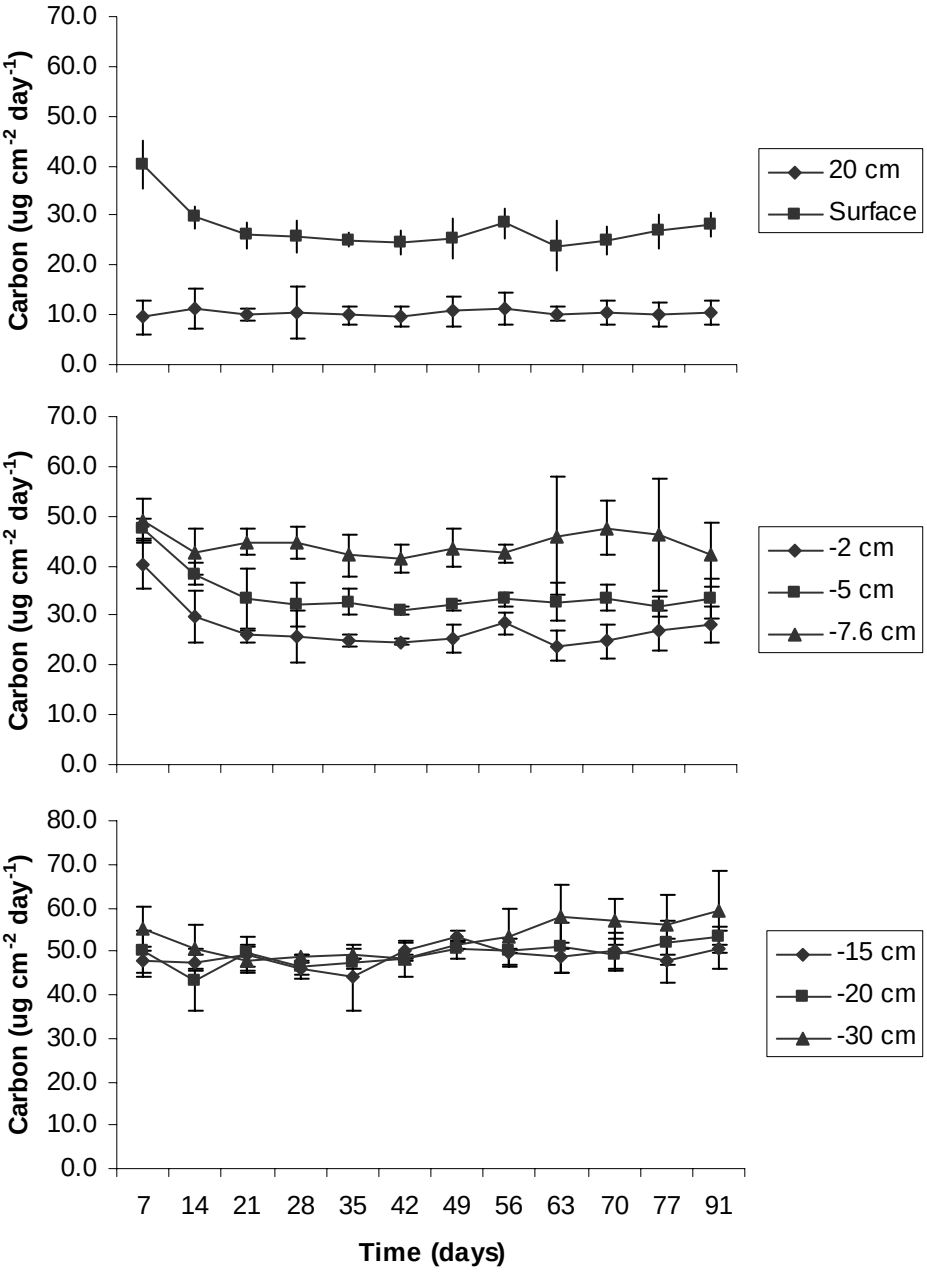

Figure 3.1 Carbon dioxide (expressed as carbon mineralized) from each treatment over the course of the experiment. Legend indicates water treatment level relative to the soil surface. Points are mean values of three replicates and error bars indicate 1 standard deviation from the mean.

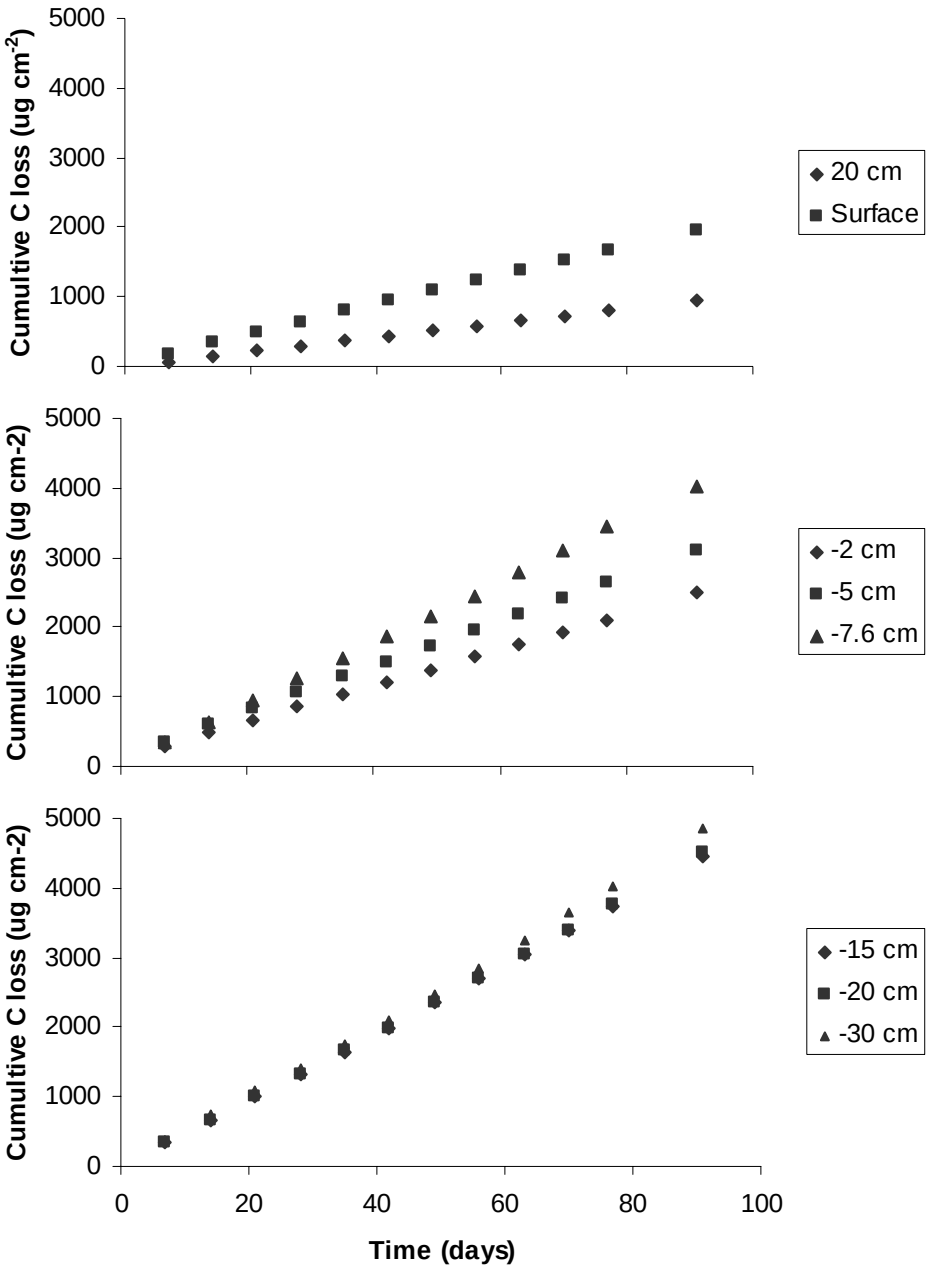

Figure 3.2 Cumulative carbon loss (measured as carbon dioxide production) over the course of the experiment. Legend indicates water level treatment. Values plotted are calculated by multiplying a 24 hr mean C value by 7 to estimate a weekly production of C.

Table 3.1. Linear equation describing cumulative carbon mineralization as a function of time (t) in days.

Treatment	Line Equation	r^2 value	k (mg m^{-2} day^{-1})
20 cm	y = 104t- 27	0.999	104
Surface	y = 213t+ 383	0.999	213
-2 cm	y = 258t + 1227	0.999	258
-5 cm	y = 328t + 1302	0.999	328
-7.6 cm	y = 436t + 364	0.997	436
-15 cm	y = 487t - 258	0.999	487
-20 cm	y = 486t - 249	0.998	486
-30 cm	y = 512t - 6	0.998	512

Plotting calculated mean rate constants (k) for CO2-C production with depth to water table treatments indicates low levels of C production within the flooded and surface saturated treatments and a dramatic increase in C production within the top 10 cm of soil when water table is located below soil surface (Figure 3.3). Lower water table treatments (-15 cm, -20 cm, and -30 cm) while producing the highest rates of C production, appear to be similar. Rates appear to plateau after treatments -15 cm and below.

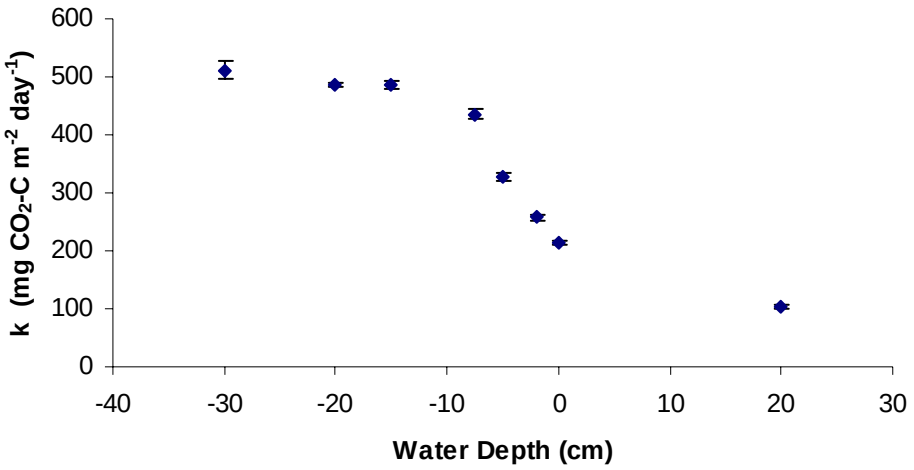

Figure 3.3 Plot of C production rate constants (k) and water treatment depth. Negative water treatment depths represent water tables below the surface of the soil. Error bars associated with rate constants represent 1 standard deviation and are less than 7 percent in all cases

Monitoring of Eh was conducted throughout the experimental period. Results of Eh measurements for soil surface and 5 cm below water table enabled the detection of anaerobic conditions for the term of the study (Figure 3.4 and 3.5). The surface soil redox measurements show increased positive measurements for deeper water table treatments. Monitoring of deep core redox at 5cm below water level treatment indicate a stable condition in most cores and a similar redox condition in all cores with the exception of treatment -7.6cm which became elevated early in the experimental phase.

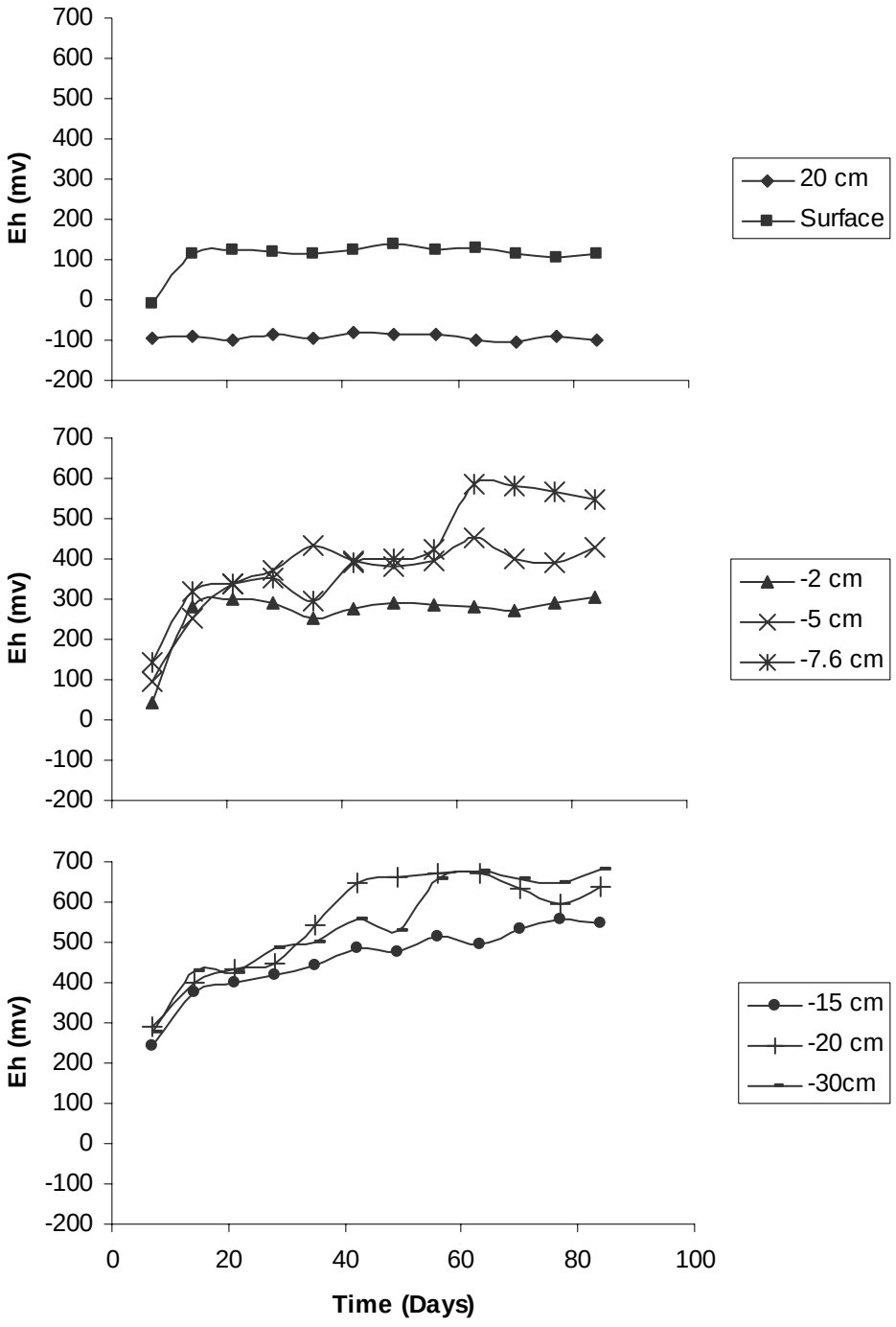

Figure 3.4 Surface soil Eh measurements for the term of the experiment. Legend indicates the depth of water level treatment in monitored cores.

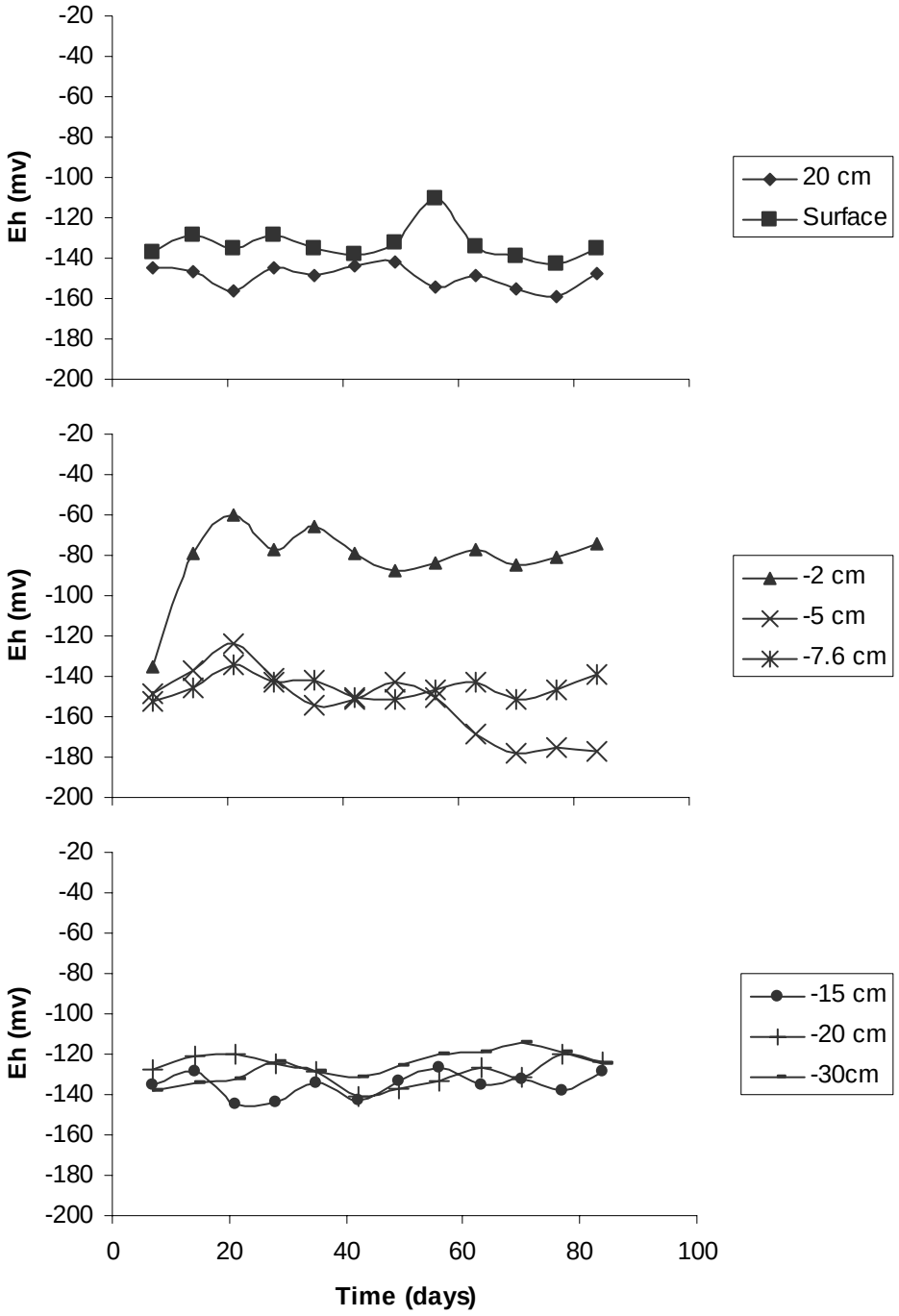

Figure 3.5 Measurements of Eh at 5cm below water table in each treatment for the first 91 days of experiment. Legend indicates the depth of water level treatment in monitored cores.

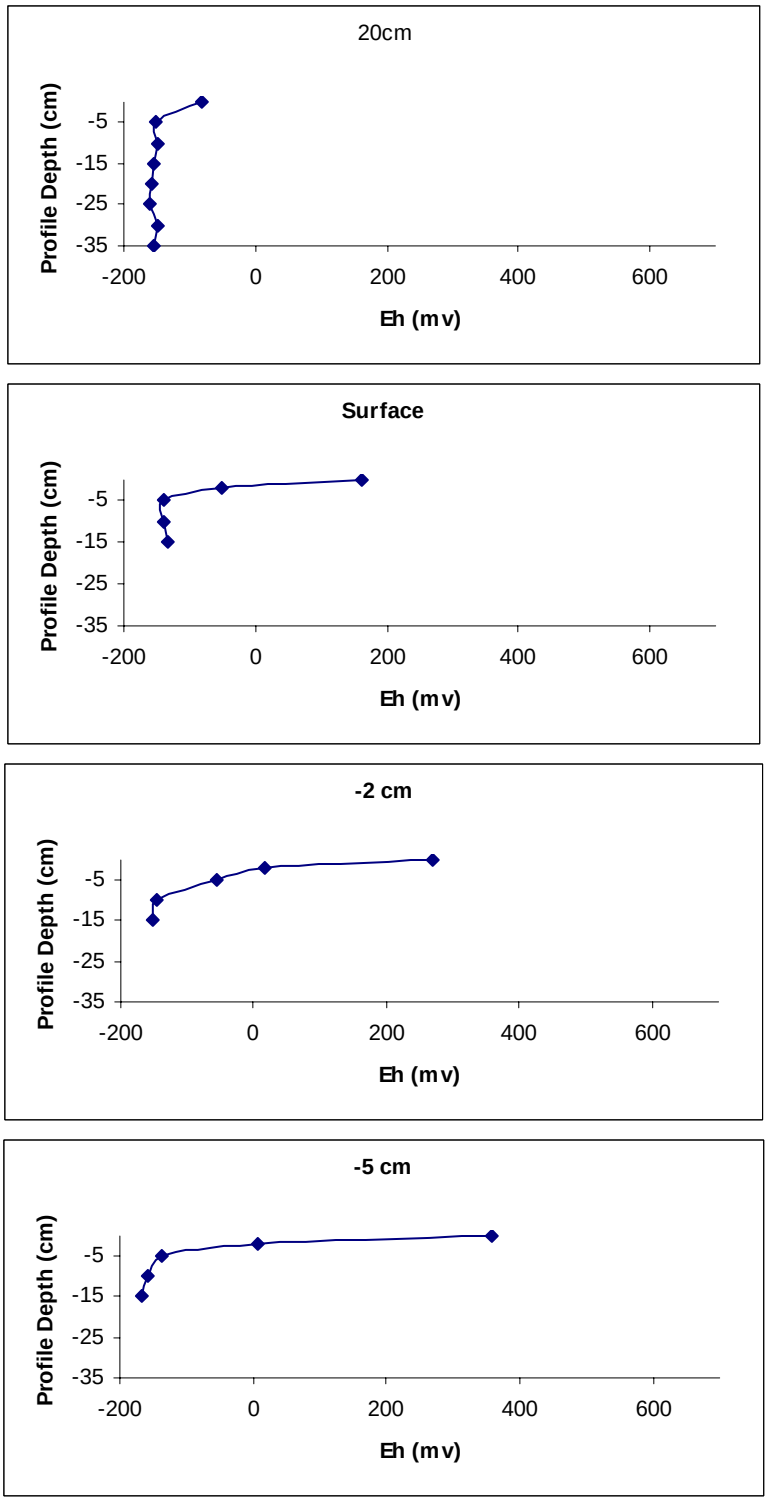

Figure 3.6 Redox profiles for treatments 20cm, surface, -2 cm and -5 cm. Profiles were taken at the termination of the study to depths of 10cm below the water table treatment prior to destructive sampling of the cores.

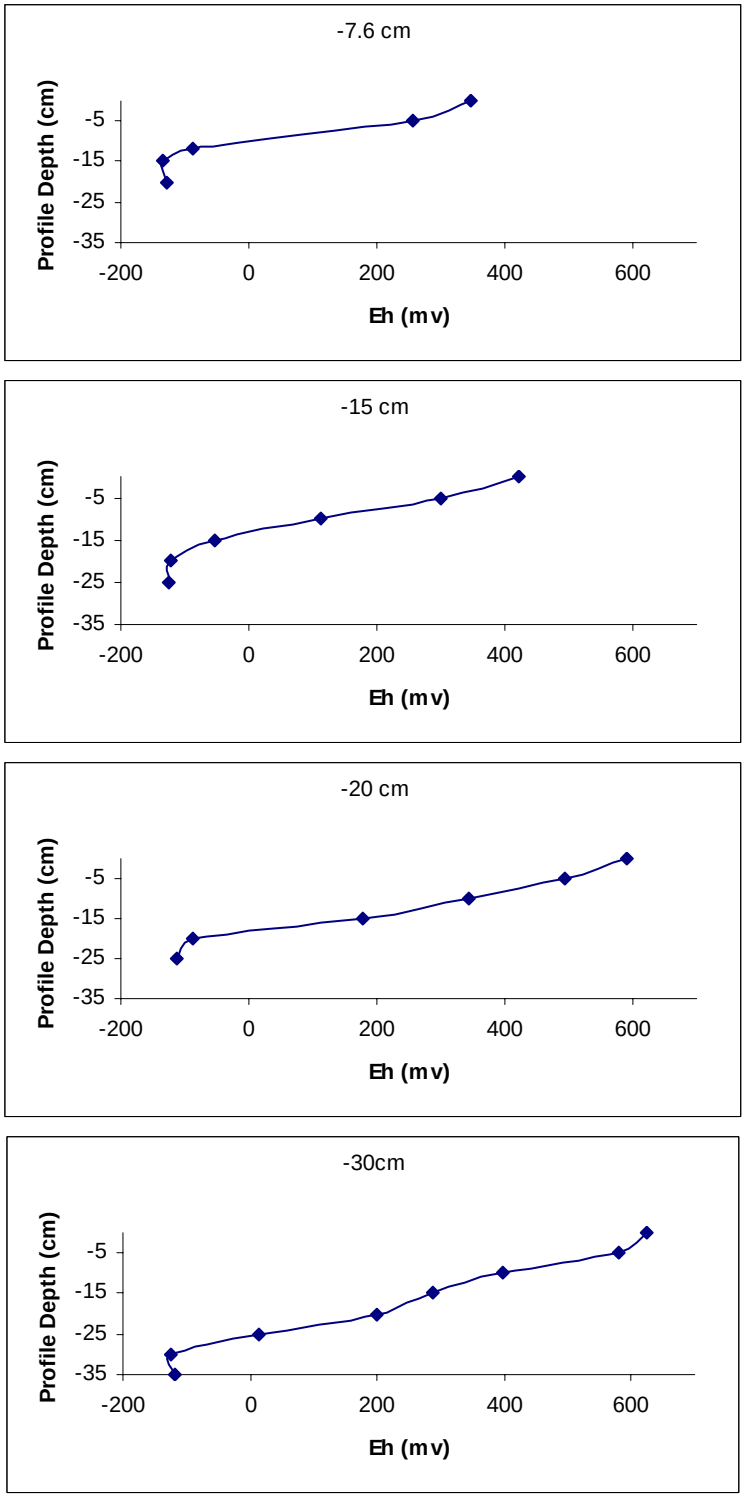

Figure 3.7 Redox profiles for treatments -7.6 cm, -15 cm, -20 cm and -30 cm. Profiles were taken at the termination of the study to depths of 10cm below the water table treatment prior to destructive sampling of the cores.

On day 90 of the experiment, redox probes were fitted to obtain Eh profiles within each of the monitoring cores to appropriate depths (Figure 3.6 and 3.7). Profiles indicate aerobic conditions in the upper portions of most cores with subsurface water tables. Values of Eh less than 200 mv can be used to infer moisture conditions within the core above the controlled water table.

3.3.2 Subsidence in soil cores

Soil surface elevation was measured in each core before and after the 91 day treatments. The percent reduction in soil height was calculated and plotted (Figure 3.8). Increase in subsidence was observed along with the lowering of the water table. There was not much difference observed in soil cores with water table -15 cm and below. Least subsidence was seen in the soil cores with water table 2 cm below the soil surface and no subsidence was observed in =20 cm flood treatment or the surface saturated treatment. Subsidence is a result of oxidation, shrinkage and consolidation which appeared to be highest in soil cores with water table -15 cm and below. Shrinkage aspects of subsidence are evidenced in the comparison of bulk densities of the two extreme treatments (Figure 3.9). This comparison demonstrates the increase in bulk density of surface soil when drying enhances compaction by removal of soil buoyancy.

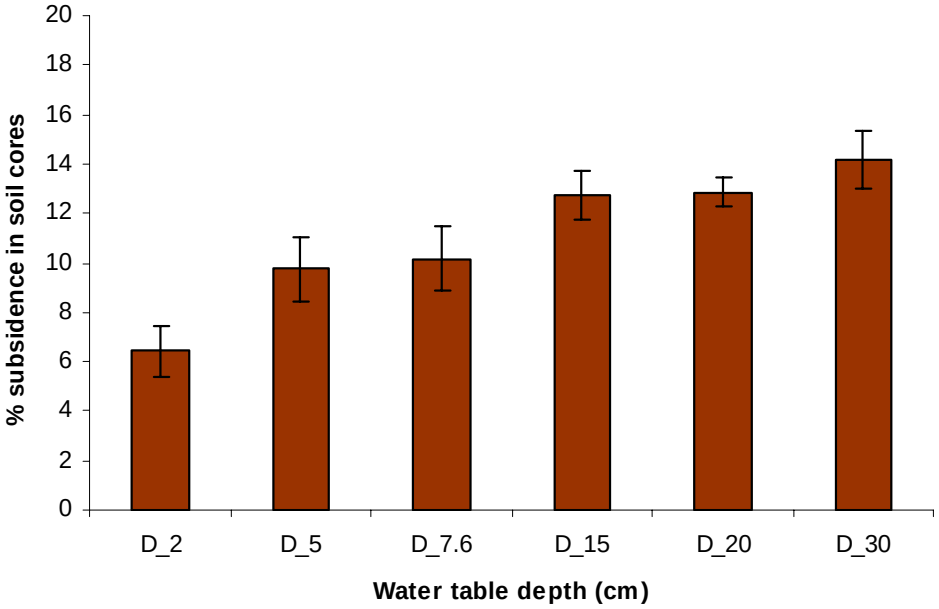

Fig 3.8 Subsidence in drained soil cores with different water table depths. Water table depths relative to the soil surface were 2cm, 5cm, 7.6cm, 15cm, 20cm, and 30 cm. The soil cores initially contained 50 cm of soil.

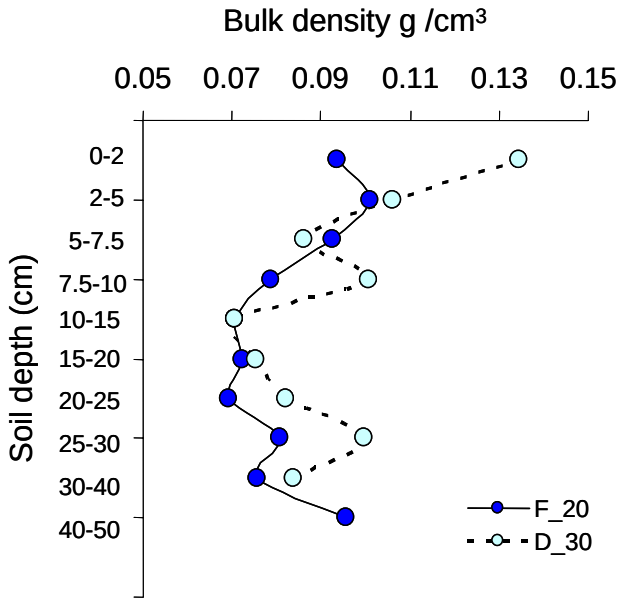

Figure 3.9 Comparison of bulk densities from +20 cm flooded (F_20) core and the -30 cm drained (D_30) treatments.

Moisture content in soil

Change in moisture content of soil with depth was observed with lowering of water table. However, below the soil depth of 10 cm, the change was negligible in all (flooded or drained) soil cores as evidenced by the moisture content recorded in soil cores with WT level of -15 cm, -20 cm and -30 cm (fig 3.10).

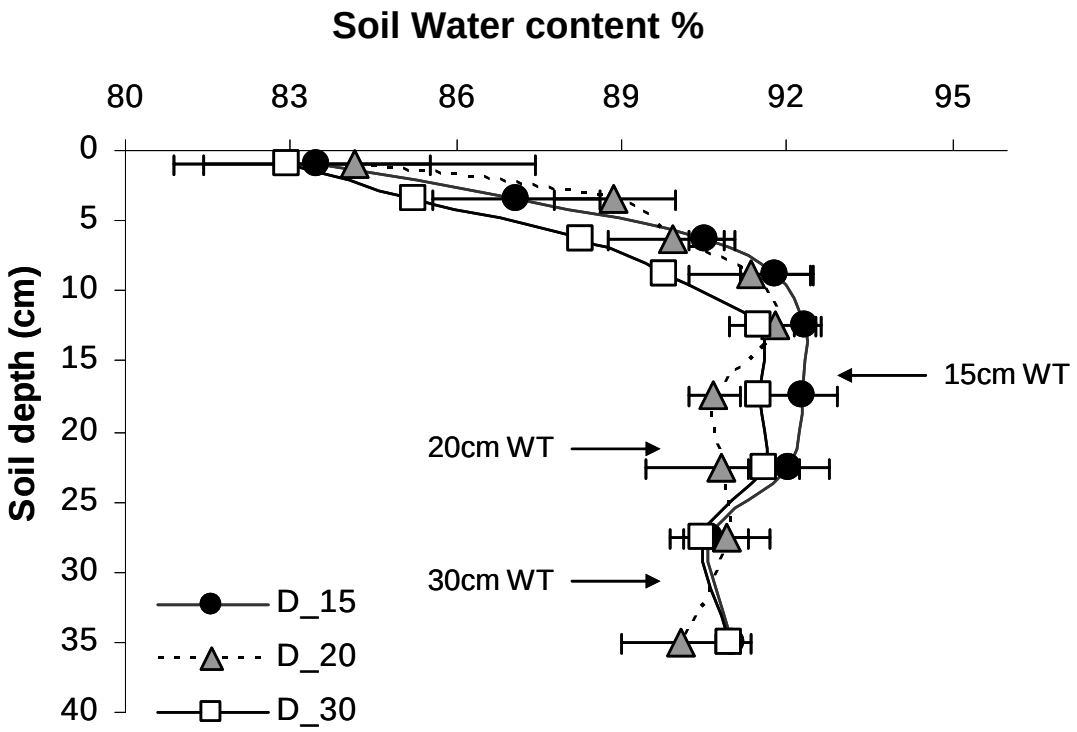

Fig 3.10 Moisture content in soil cores with water tables adjusted to the depth of 15 cm (D_15), 20cm (D_20), and 30 cm (D_30) below soil surface.

This can be explained by either due to low draining capacity of the lower soils which may have compacted over time or perhaps due to high capillary connectivity in soils or, due to water matric suction. Separate correlation curves of moisture content and water table depths for different soil depths showed higher slopes for soil samples from the upper 10 cm soil cores.

Microbial biomass

The microbial biomass carbon (MBC) in soil cores did not vary much with soil depth (Table 3.2). In drained cores, MBC did not change past 10 cm of soil depth, but was higher in the 0-2 and 2-5 cm layers of the flooded and saturated cores when compared with the drained cores. Soil moisture content in flooded and surface saturated cores was similar to that of drained cores past 10 cm of soil depth, but was lower in the upper layers of the soil cores (fig 3.11). Similar trends in soil moisture content and MBC suggested some correlation between the two.

Table 3.2 Microbial biomass carbon (MBC) along the depth in the flooded and drained soil cores. Water table was adjusted relative to the soil surface. Flooded 20 cm (F_20), surface saturated (D_0), drained -2cm(D_2), drained -5cm (D_5), drained -7.6 cm(D_7.6), drained -15cm (D_15), drained -20 (D_20), and drained -30 cm (D_30). The values reported are the averages of triplicates. Standard deviations are reported in the parenthesis.

Soil depth	**F_20**	**D_0**	**D_2**	**D_5**	**D_7.6**	**D_15**	**D_20**	**D_30**
(cm)	----------MBC g kg^{-1}----------							
0_2	15±.5.1	17.6±2.8	6.6±2.8	8.2±0.8	10.0±3	9.4±1	9.0±5.5	7.3±1.6
2_5	14.5±1.5	14.7±2.9	5.2±4.2	5.7±1.1	10.5±1	9.1±0.9	8.5±2.7	8.2±1.5
5_7.6	13.6±2.0	18.0±3.9	nd	5.6±3.4	13.3±2.4	11.0±1.3	10.5±2.2	8.7±0.8
7.6_10	13.6±2.4	16.0±0.9	nd	nd	12.9±3.8	12.6±1.7	11.8±3.3	9.9±1.2
10_20	13.1±1.2	14.9±1.6	nd	nd	nd	13.1±0.8	10.6±0.9	11.3±0.5
20_30	13.1±1.8	13.0±0.9	nd	nd	nd	nd	9.3±2.2	11.0±1.2
30_40	12.5±1.3	12.2±0.8	nd	nd	nd	nd	nd	10.3±0.8

nd: not determined.

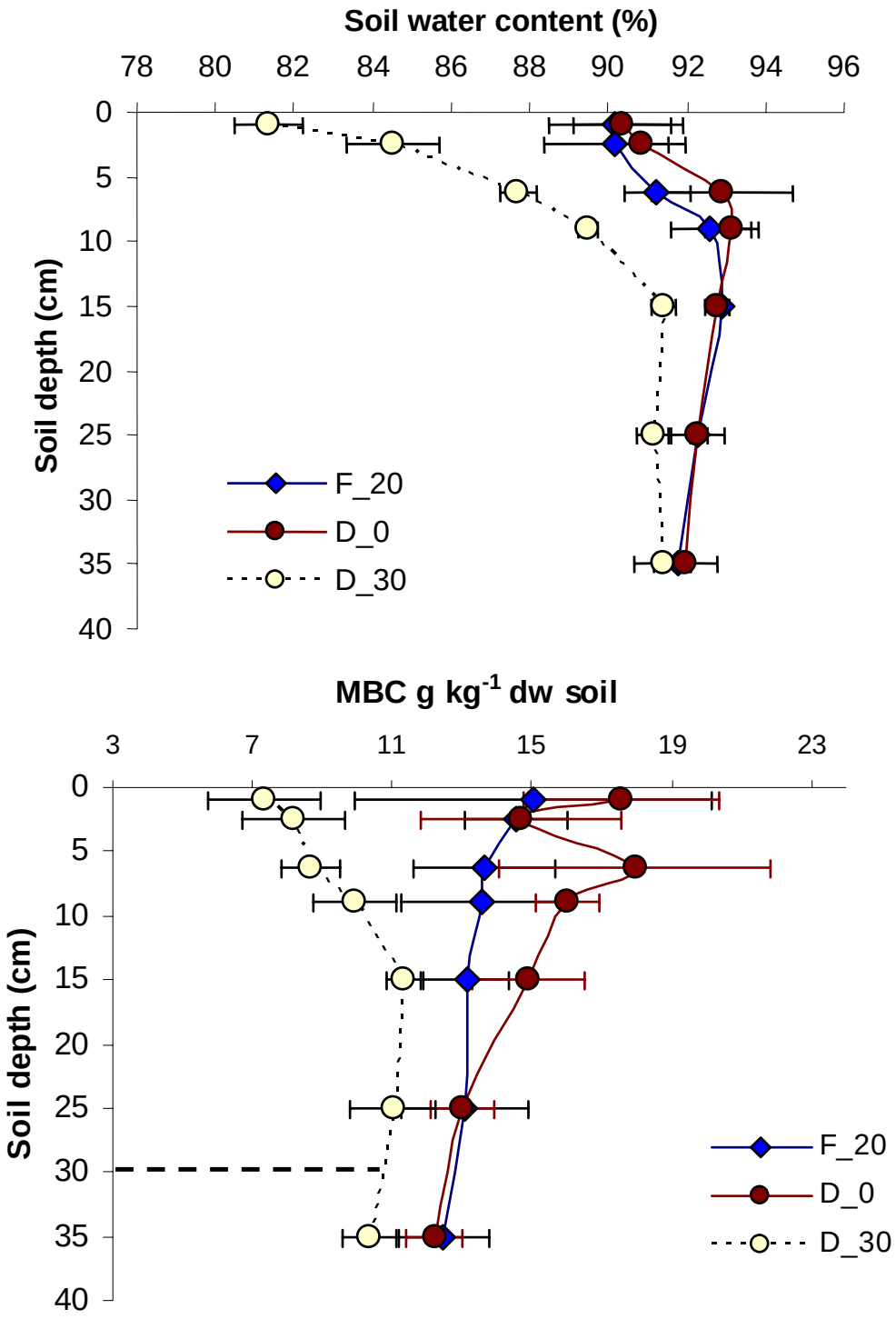

Fig 3.11 Moisture content and microbial biomass along the depth of soil cores in flooded soil with +20 cm water table (F_20), surface saturated (D_0), and drained soil with water table -30cm (D_30).Dashed line indicates the water table depth in the drained core (D_30).

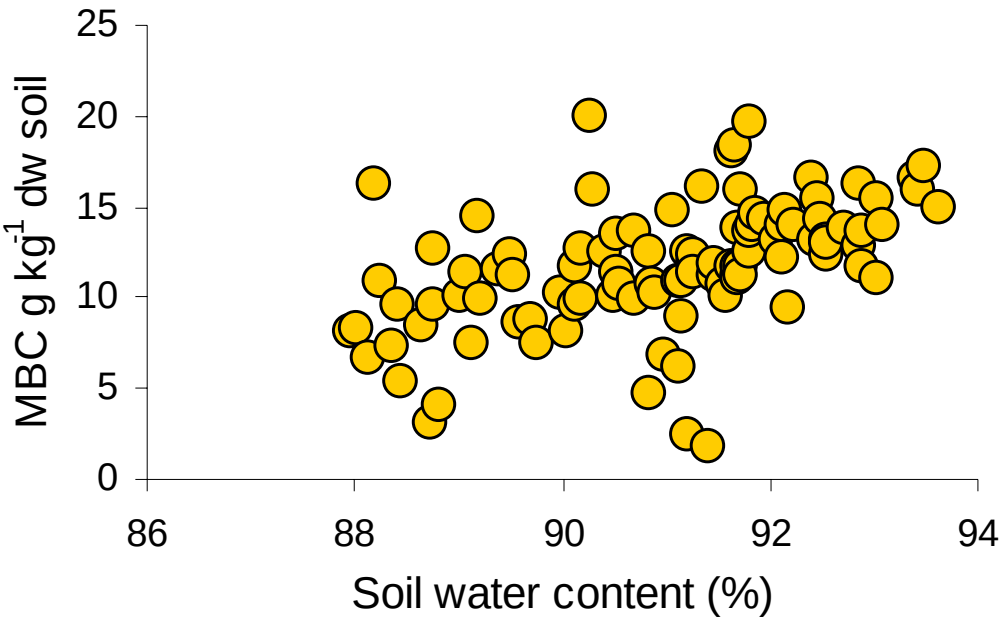

Fig 3.12 Microbial biomass carbon was positively correlated to the moisture content in cores along the soil depth.

In drained cores, MBC was positively correlated with moisture content indicating that MBC was greatly affected by the moisture content (Fig 3.11). However, this relationship was less apparent in soil cores with water table lower than 10 cm. The reason for higher MBC in flooded (F_20) and surface saturated (D_0) cores when compared with other drained cores is not clear at this time.

Total phenolic content

In the flooded core, phenolic content did not vary much with soil depth. Water table drawdown decreased the total phenolic content in the soils (fig 3.13). Oxygen availability as an effect of water table drawdown increased the degradability of phenolics in the drained core.

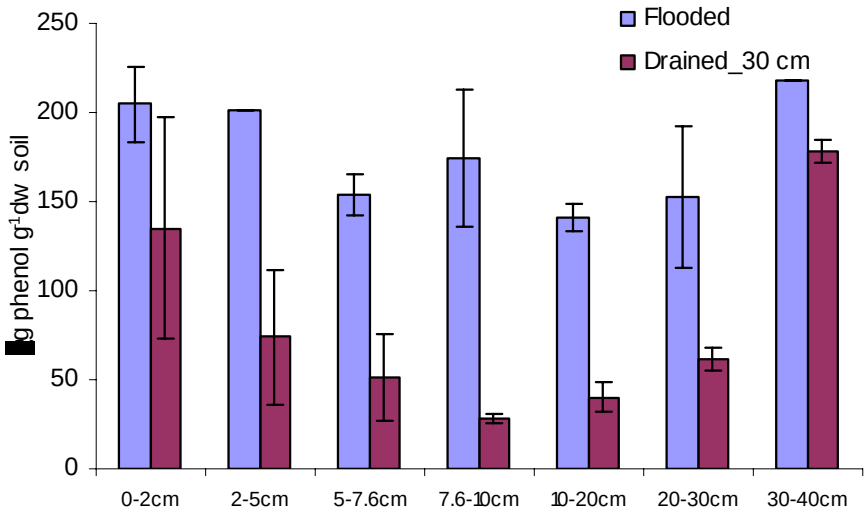

Fig 3.13 Comparison of total phenolics with soil depth within flooded and drained (30 cm) soil cores. Water table was adjusted relative to the soil surface.

Decline in phenolic content was apparent when water table was lowered to -20 cm soil depth (fig. 3.14). There was no clear declining trend observed with the lowering of water table in soil cores.

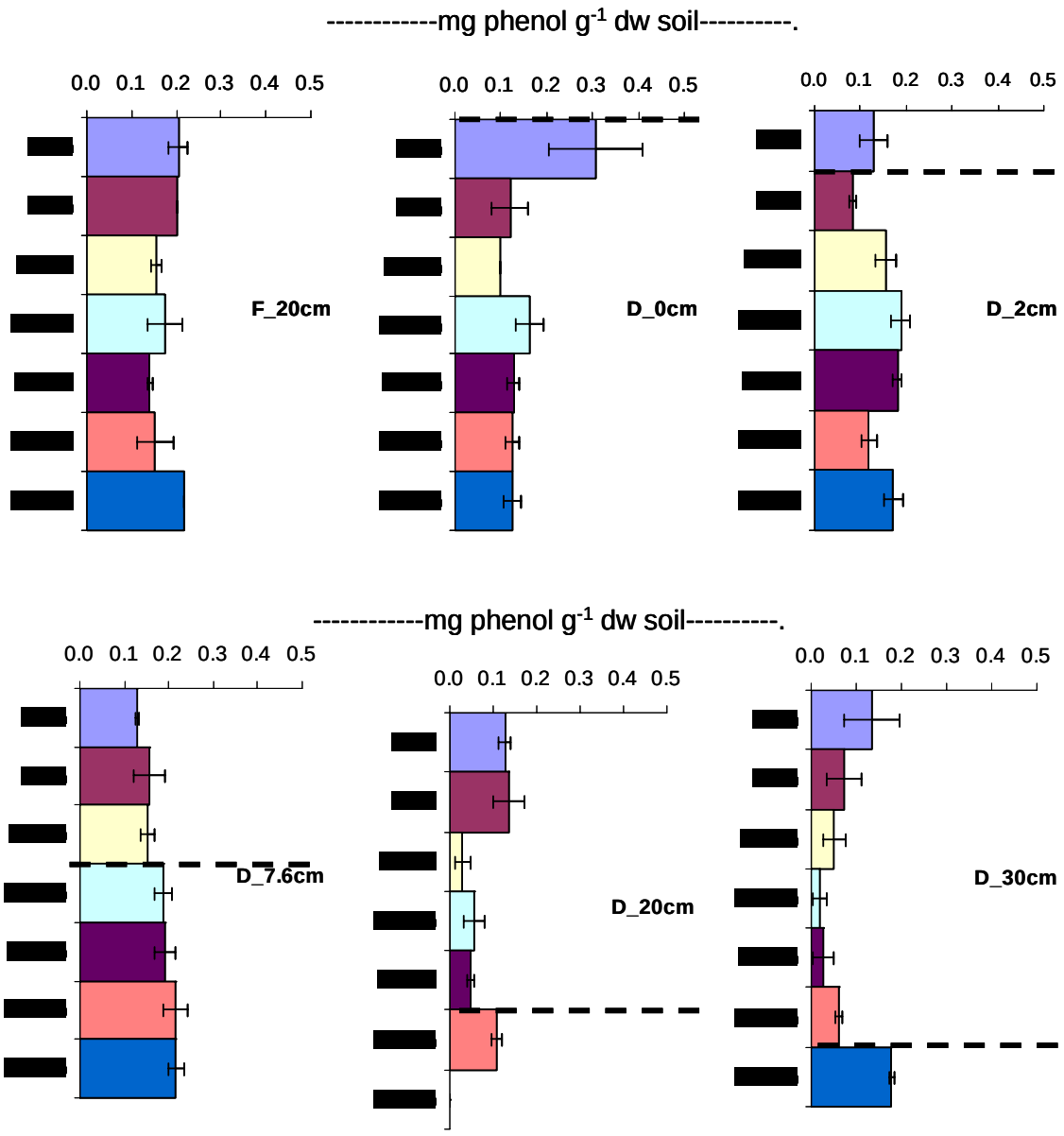

Fig 3.14 Total phenolics with soil depth in the flooded and drained soil cores. Dotted line represents the water level in soil cores. Water table was adjusted relative to the soil surface. The y-axis represents the soil depth. Flooded 20 cm (F_20), surface saturated (D_0), drained -2cm(D_2), drained 7.6 cm (D_7.6), drained -120 (D_20), and drained -30 cm (D_30).

Phenol oxidase activity

Phenol oxidase activity (POA) was measured in all cores with soil depth (Fig 3.15). There was no phenol activity detected in the flooded core. POA in the surface saturated soil core was lower than that observed in the other drained cores. Although there was no correlation found in the POA activity and the water table drawdown, increased activity was observed in the soil core with water table at -30 cm soil depth. Enzyme activity was observed at all soil depths till 30 cm.

β-Glucosidase activity

β glucosidase activity (BGA) showed a positive correlation with the moisture content in the soil (Fig 3.16). BGA was detected all along the depth of 40 cm (Fig 3.17). Cores that were flooded (F_20) and surface saturated (D_0) showed high enzyme activity in soil at 7.6-10 cm depth. A distinct peak of BGA in the 7.6-10cm section in all drained cores was observed suggesting the presence of higher substrate at that depth. Activity in the 0-2 cm sections of drained soil cores appeared to decline with the lowering of water column.

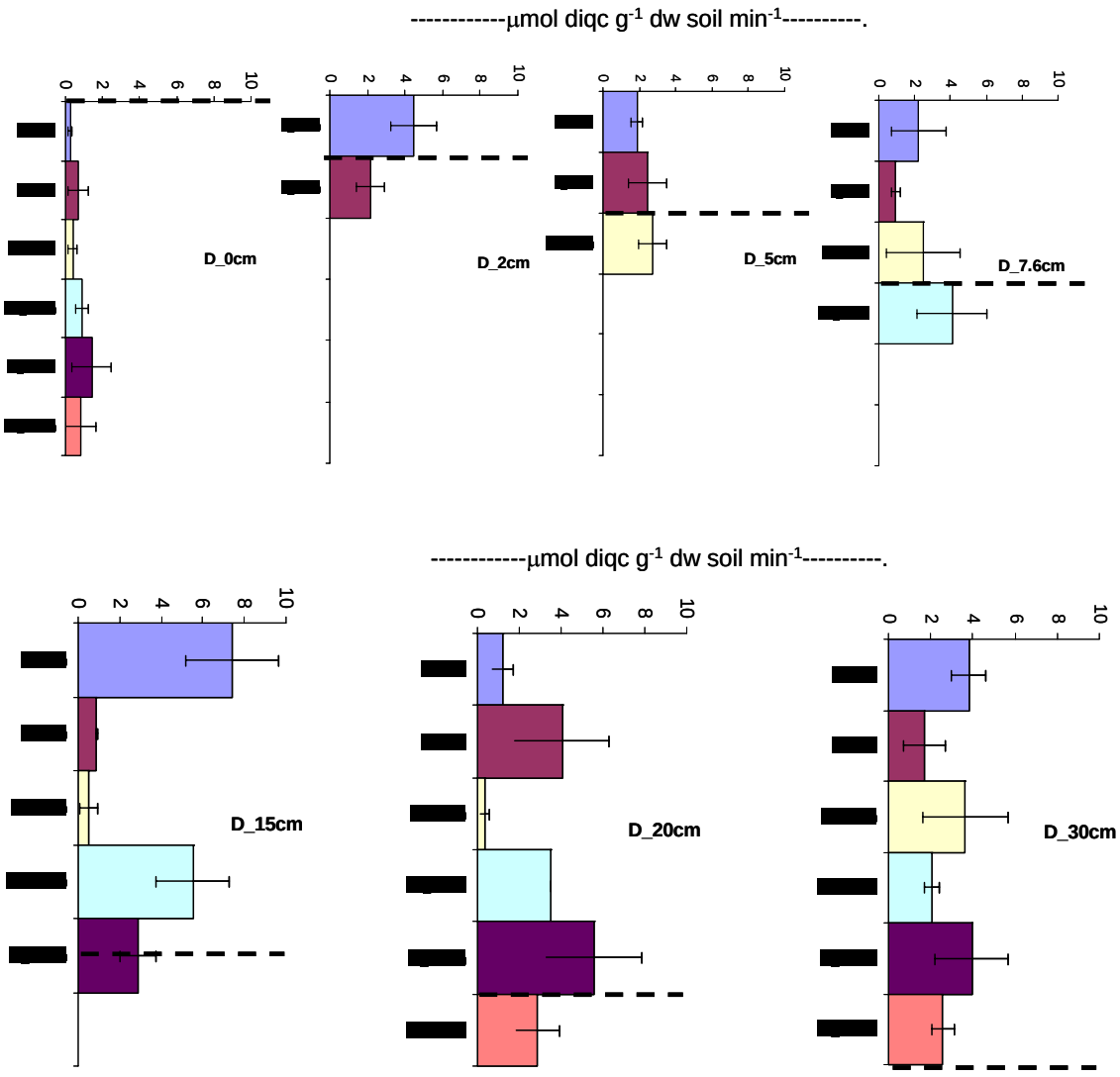

Fig 3.15 Phenol oxidase activity with soil depth in flooded and drained soil cores. Dotted line represents the water level in soil cores. The y-axis represents the soil depth. Water table was adjusted relative to soil surface. Surface saturated (D_0), drained -2cm(D_2), drained 7.6 cm (D_7.6), drained -120 (D_20), and drained -30 cm (D_30).

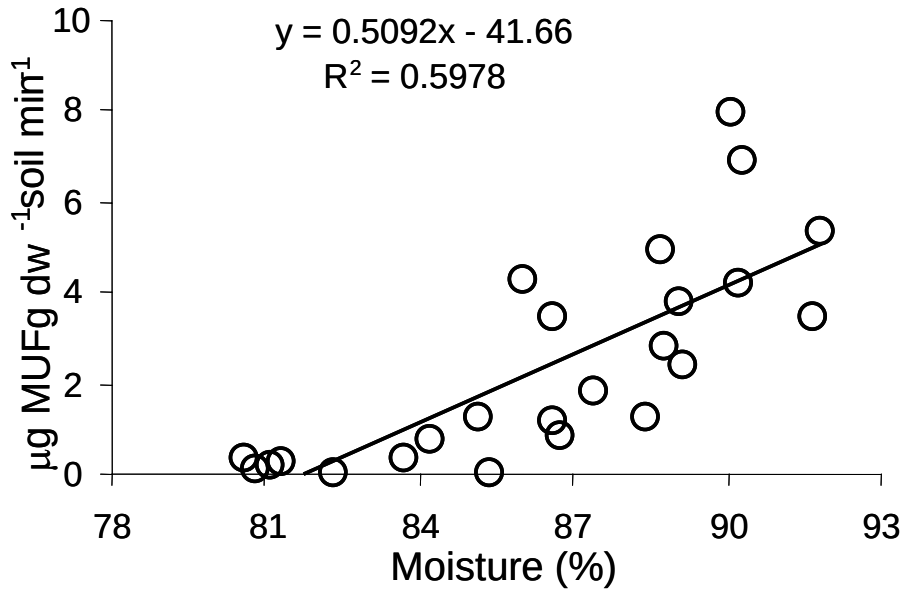

Fig 3.16 Correlation between β glucosidase activity and the moisture content in flooded and drained soil cores.

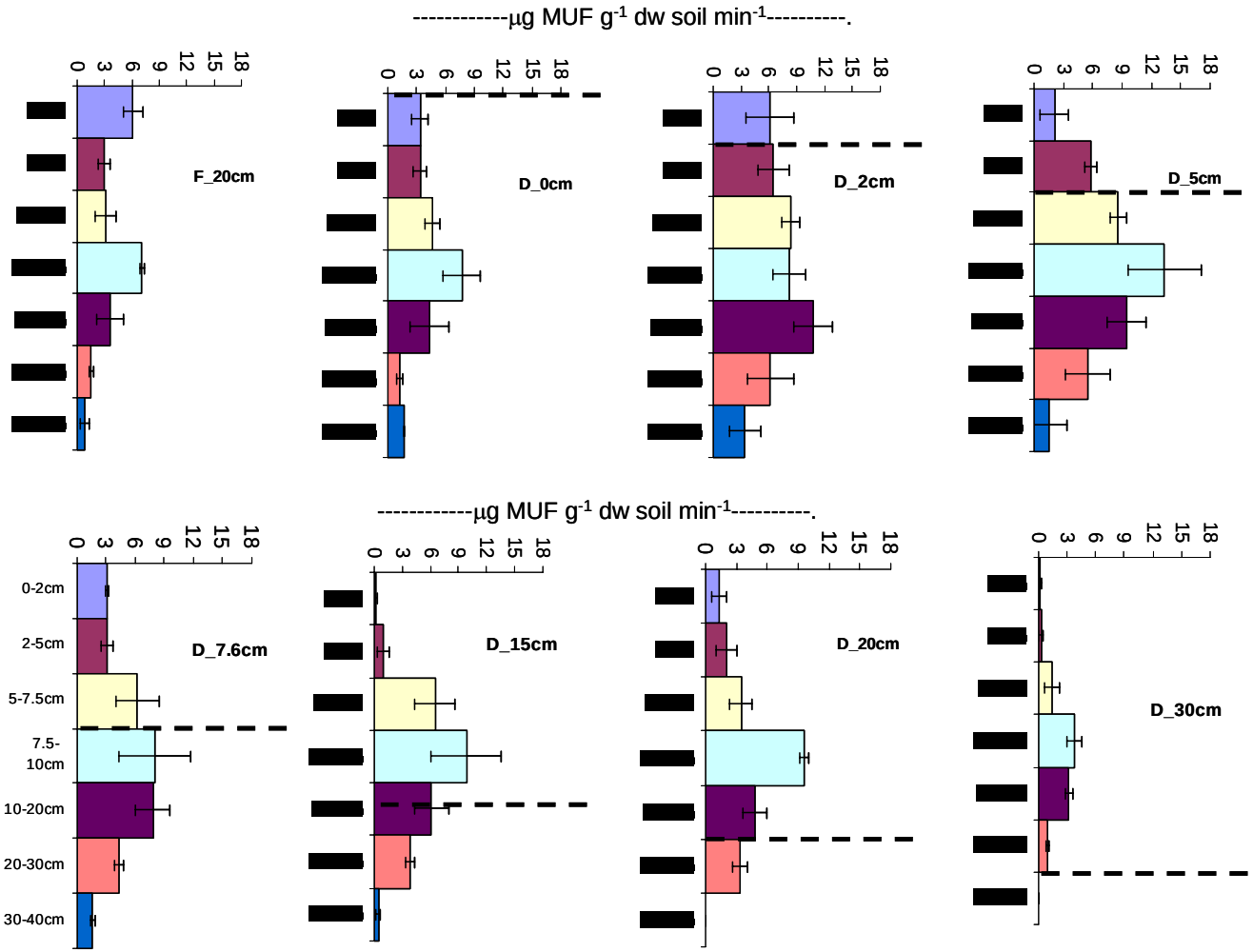

Fig 3.17 β glucosidase activity with soil depth within the flooded and drained soil cores.. Water table was adjusted relative to the soil surface. Flooded 20 cm (F_20), surface saturated (D_0), drained -2cm(D_2), drained 7.6 cm (D_7.6), drained -120 (D_20), and drained -30 cm (D_30). Dotted line represents the water level in soil cores.The y-axis represents the soil depth.

3.4 DISCUSSION

3.4.1 Carbon dioxide flux and water levels

The observed levels of CO_2 production under variable water table treatments were quite constant throughout the duration of the experiment suggesting that the microbial populations responsible for oxidation of soil organic matter rapidly reached a threshold population, dictated in part by substrate quality and availability, which was maintained throughout the experiment (figure 3.1). There were some declines observed from the first to second week in some treatments suggesting that upon draining, microbial populations increased above the threshold capacity of the available macro and micro nutrients needed to sustain the population and thus declined leading to a slight decline in CO_2 production. Throughout the experiment, lower water tables resulted in greater production of CO_2 as expected. Calculation of rates suggests that drained soils experience oxidation at a rate 5 times that of saturated or flooded organic soils in the BCMCA (Figure 3.3). Interestingly, the cumulative C production and the rates of production (Figures 3.2 and 3.3) suggest that there is a rapid increase in oxidation in the top 10 cm of soil upon drainage, but that below approximately 15 cm, production rates plateau, even when water levels are at -30 cm. There are two plausible explanations for this observation. First, the top 10 cm of the soil profile contains the freshest and most labile organic matter, which due to its quality, facilitates more oxidation when drained. Also, by matter of depth, the older, more refractory materials are lower in the profile, and even when drained, they are not preferred substrates until more labile materials are exhausted. Secondly, observations of redox potentials below the level of aerobic conditions were routinely observed above the water levels in most drained treatments (Figures 3.6 and 3.7) suggesting a capillary

fringe of near saturation conditions extending above the set treatment water level. This capillary fringe could reduce oxidation as well and be a significant factor affecting C production rates.

Surface redox probes captured the dewatering trend in treatments with subsurface water levels (Figure 3.3). These results suggest a slow rate of soil dewatering in many treatments after the initial rapid decline in water content and penetration of aerobic conditions into the surface soils. It is important to note that these experiments were conducted in temperature controlled environments in the laboratory in the dark. Field conditions, where sunlight, humidity, temperature, wind and plant associated evapotranspiration would likely accelerate the drying of soil cores and thus potentially increase oxidation in the short term after drawdown events occur.

3.4.2 Relationships between water table depth and the microbial enzymatic processes.

It is a general assumption that lowering of water table in wetlands and peatlands enhances the soil C efflux by stimulating the microbial activity. Increase in microbial activity is mainly due to increased oxygen availability and aerobic respiration that yields more energy. However, our results showed that even though the CO_2 efflux from soils increased with lowering of water table, it was not accompanied by increased microbial enzyme activities in soils. Although the enzyme activities were higher in the drained soil cores, there was no relationship between the water table depth and the microbial enzyme activities. Increase C mineralization in drier soils (Heathwaite, 1990) as a result of microbial activity is reflected in the subsidence of soils (Schothorst, 1977). In this study, increased rate of respiration in soil cores was accompanied by subsidence in soils

indicating increased microbial activity. The lack of similar correlation between enzyme activities and water table level suggests that the enzyme activity may have been inhibited. Soil particles and organic compounds in soils are known to immobilize enzymes by partially or completely inactivating them (Wetzel, 1993; Burns, 1983). In our study it can be speculated that the soils have a high enzyme immobilizing ability. This may be true for phenol oxidase enzyme but, did not appear to be true for b glucosidase which was correlated with moisture content in soils.

Phenolic compounds are potential enzyme activity inhibitor (Freeman et al, 1990, Wetzel, 1992). In this present study total phenolic content was high even in drained soil cores. With low phenol oxidase activity, phenolic substances may have acted as enzyme inhibitors. However, the b glucosidase enzyme did not appear to be affected by the presence of phenolics. Therefore, it could be speculated that most carbon that was mineralized in the soil cores may have been due to cellulosic substances that form the substrates for glucosidase enzyme.

For cores that were drained to 30 cm soil depth, phenolic content was much lower. This may have been due to the phenolic oxidase activity which appeared to be high in these drained cores, in contrast to the b glucosidase activity that was suppressed. Degradation of phenolic content in these drained cores may be contributing to the CO_2 efflux. At this moment it is not clear if the two enzymes are produced by the same microbial community that exists in these soils or if the microbial community structure changes with the change in the oxygenated conditions in soils.

3.5 Conclusions

Intact core study indicated higher C-loss by microbial activity (CO_2 production) in drained soils. The C loss increased with lowering the water column within one week. Constant CO_2 production in cores for the next 14 weeks suggested the degradability of C sources was not too different although Eh conditions continued to increase throughout the course of the experimental phase.

Calculated rates of soil loss due to secondary subsidence (subsidence due to soil organic matter oxidation) was found to increase with water level drawdown. Estimates of soil loss are as follows: for surface water treatment, subsidence rates were estimated to be 0.23 cm yr^{-1} which increased to 0.57cm yr^{-1} for the lowest water treatment (-30cm). These results suggest that any drawdown event will have some soil subsidence associated with it. The observed primary subsidence (subsidence due to shrinkage or compaction) would exacerbate this scenario as drawdown depth increases.

Although the phenolic content in soils did not vary much with lowering of water table, there was a significant decline in the phenolic content in cores drained to 30 cm soil depth. Increased phenol oxidase activity was also recorded in this soil core. β−glucosidase activity remained high in the drained cores was well correlated with the moisture content of soils. However, this enzyme activity appeared to be suppressed in the soil core that was drained to 30 cm soil depth. Most C loss in soil cores with higher water table occurred due to degradation of cellulosic and lignocellulosic compounds. Polymeric carbon compounds get hydrolyzed with soil depth contributing to the C loss and causing subsidence of soils. Difference in activity zones of the two enzymes with soil depth most likely is due to the enzyme –specific substrate accumulation.

CHAPTER 4

INFLUENCE OF INUNDATION FREQUENCY ON SOIL ORGANIC MATTER DECOMPOSITION

4.1 Introduction

Frequency and duration of inundation are defining characteristics of most wetland ecosystems. The time between draw down events of surface water and reflooding often determine plant community structure and recruitment, as well as, modulate biogeochemical cycles within the water column and associated organic soils common to wetland systems. The duration of flooding can affect critical changes to soil biosphere by driving oxygen depletion and thus the fluctuation between aerobic and anaerobic conditions. These fluctuations between dominantly oxidizing and reducing environments is the force behind much of the unique redox chemistry associated with wetland soils.

Of great importance to the management of wetland systems is the timing and delivery of water to natural systems such that loss of organic soils to oxidation processes is minimized or negated. This portion of research investigates the effects of different frequencies of inundation on the rates of organic soil oxidation in BCMCA.

4.2 Methods

Intact, flooded soil cores were taken from the same location as those taken in Chapter 3 (*Nymphea* dominated slough reference site B) and transported to the laboratory core study tank where they were kept at constant temperature and protected from light. Cores were subjected to variable inundation frequencies for a period of 150 days. The

inundation treatments consisted of flooding the cores to a depth of 15 cm with BCM site water for variable lengths of time and then slowly draining cores to a depth of -15cm for equal amounts of time. The durations of inundation were 10 days, 25 days, and 50 days with corresponding durations of drawdown. Two additional sets of cores acted as controls for permanent inundation (15cm) and permanent draw down (-15 cm). Soil CO_2 flux was measured as described in Chapter 3.

4.3 Results

Experimental treatments of drawdown and reflooding at different frequencies showed marked responses of soil microbe to flooding and drawdown. For each treatment, flooding resulted in significant decreases in flux of CO_2 from the cores while drawdown events elicited a greater response in oxidation of organic matter (Figure 4.1). at the scale of measurement (every 10 days), delays of microbial responses were not observed. In comparison to the continually flooded (CF)(Figure 4.2A) treatment, the 10, 25, and 50 day frequency treatments showed significantly greater total oxidation. In contrast, the continually drawndown (CD) control was greater in total CO_2 loss than any of the frequency treatments (Figure 4.2B). Calculation of mean rates of CO_2-C loss from the treatments and controls shows the loss to continual flooding to be the least and the continually drawndown to be the greatest (Figure 4.3). The treatments however, fell in between these two extremes with the 10 and 25 day treatment frequencies being somewhat similar and the 50 day treatment being greater in CO_2-C loss.

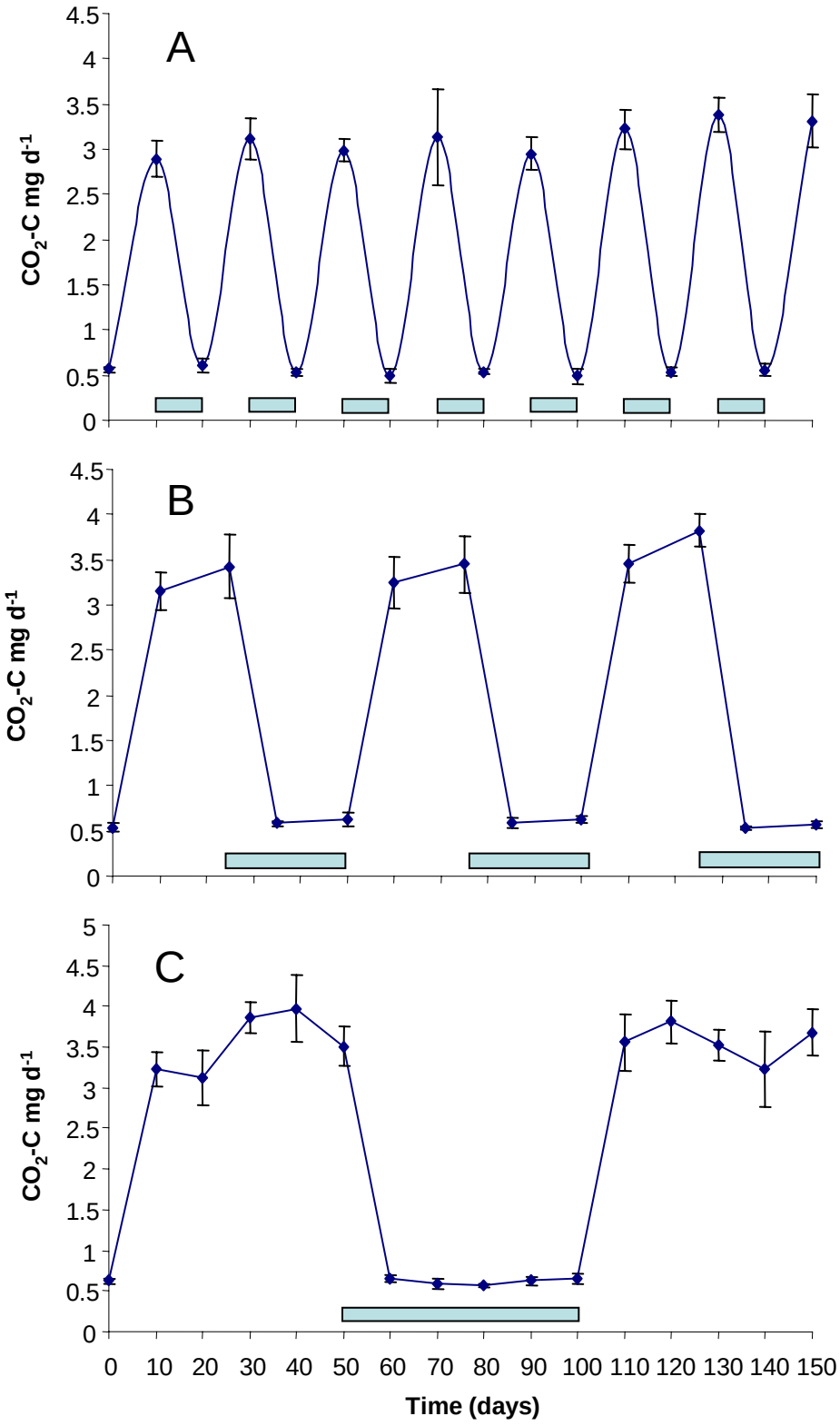

Figure 4.1. Results of frequency treatments. (A)represents 10 day flooding, (B) 25 days, and (C) 50 days. The bars on the x axis of each graph indicate the periods of flooding. Error bars indicate 1 S.D. from the mean.

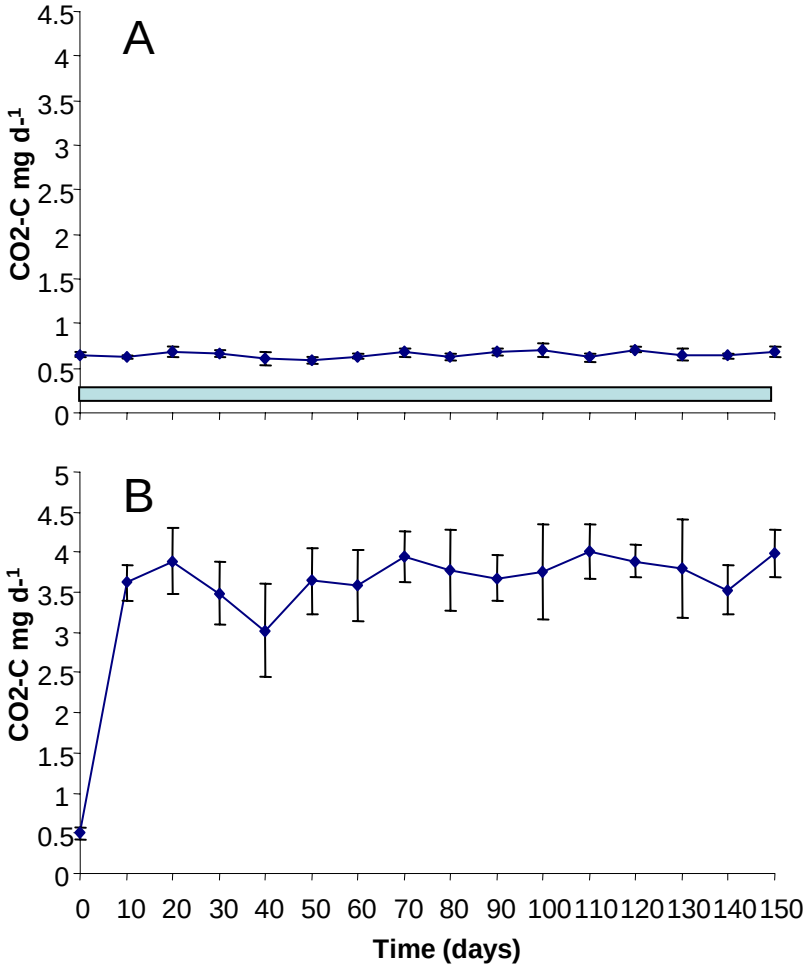

Figure 4.2. Results of frequency treatment controls. (A) indicates the continually flooded control (CF) and (B) indicates the continually drained (CD) treatment. Error bars represent 1 S.D. from the mean.

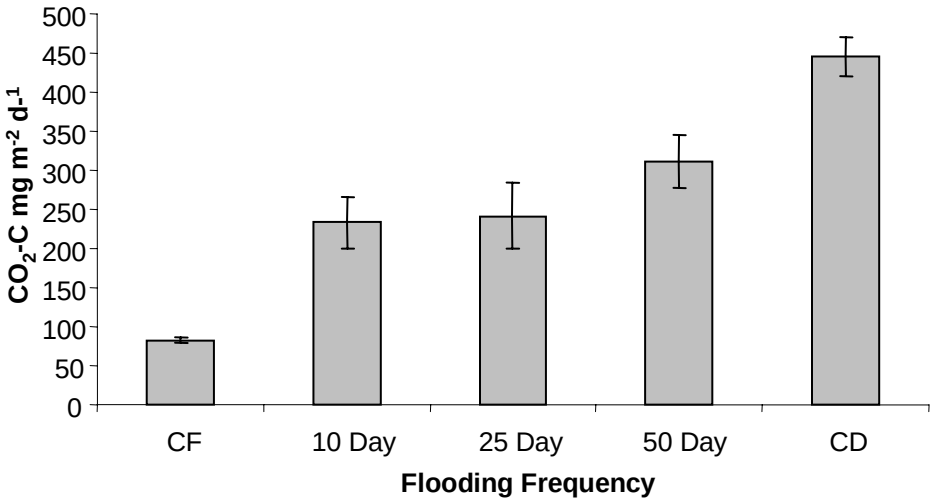

Figure 4.3 Mean flux rates (k) for each treatment frequency. Continuously flooded treatment is denoted by CF and continuously drained by CD. Error bars represent 1 S.D. from the mean.

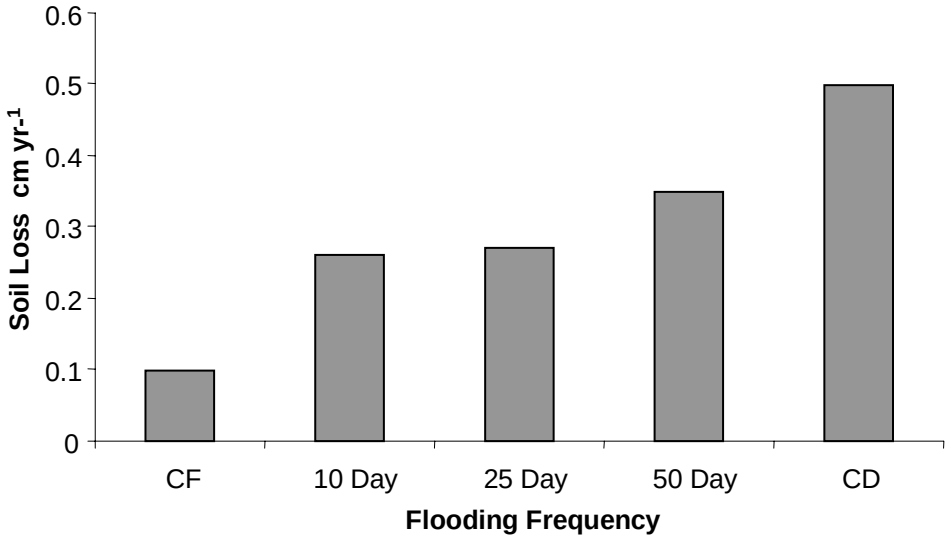

Figure 4.4 Calculated soil loss in cm per year for each of the frequency treatments.

4.4 Discussion

Investigation of potential soil subsidence with respect to frequency of flooding suggests that continual flooding minimizes the oxidation of organic matter. Short term flooding and drawdown events (10 days) seemed to be lower in carbon loss with respect to the other duration treatment, however, there was no statistically significant difference between the 10 day and 25 day treatments. The 50 day treatment cycle did produce greater amounts of CO_2 production, with only the continually drained controls exhibiting a greater loss for the study period.

It was expected that short term draining and flooding cycles would inhibit production of CO_2 through maintaining a higher moisture content through out the profile and thus creating a lag phase or period in which moisture content was high enough in the capillary fringe (above the water table) to inhibit aerobic mineralization of organic matter. Stephens (1956) suggested that keeping the moisture content above 0% or below 50% slowed microbial activity markedly. The repeated wetting and drying of the 10 day treatment was believed to be sufficient to maintain the high moisture content required to slow microbial activity. This was not observed and likely is effective only in the short term (1-5 days). Due to the sampling design of these experiments (every 10 days), this potential lag phase of oxidation was no observed.

Glaz (1995) reported that the range of inundation required to protect organic soils from significant secondary subsidence was 15-94% of the year. The experiments presented here fall within that range, but do not account for the extremes. If these treatments were to persist for a full year, all treatments would be inundated

approximately 50% of the time. Brooks and Lowe (1984) suggested that 60% inundation annually would significantly slow subsidence in Upper St. Johns River soils.

Calculation of potential losses of organic soil to secondary subsidence (Figure 4.4) suggest that optimal draining periods would be 25 days or less based upon the intervals studied here. Continual flooding would be the optimal treatment for conservation of organic soils, unfortunately, continual flooding is neither natural in many wetland systems nor feasible from a management perspective. The potential losses reported here indicate that even continual flooding has loss associated with it. This is due to the lack of a primary production consideration. Still these estimates of loss are lower than others calculated for the drawdown experiments in the previous chapter. This indicates the effect of reflooding events interspersed throughout the term of drawdown. Relative to other studies, the Upper St. Johns River basin soils found in BCMCA are still relatively recalcitrant in comparison. Dolman and Buol (1967) calculated soil subsidence rates for drained wetlands in North Carolina to be 1.2 to 3.5 cm yr^{-1}. Further, in close proximity to the BCMCA is the Everglades Agricultural Area where Stephens (1984) reported an estimated subsidence rate of 3 cm yr^{-1}. Both of these estimates do not distinguish between primary and secondary subsidence. This study only predicts secondary subsidence due to organic matter oxidation and therefore, if primary subsidence were accounted for, these rates may be comparable.

4.5 Conclusions

Frequency of flooding and draining appears to have a significant effect in slowing subsidence via oxidation when the soils are drained and flooded again in cycles versus continual draining. However, no frequency of draining and flooding measured in this study approximates the effect of continual flooding or that of natural hydrologic cycles in the BCMCA without the addition of primary production. Frequencies of 25 days or less, in this study, suggest that short term drain and flood cycles are better for sustaining organic soils than prolonged drainage as observed in the previous chapter. While shorter duration drawdown events were not investigated in this study (<10 days) it is expected that drawdown events less than 10 days would be optimal for protection of soils from subsidence in the BCMCA. Estimated rates of soil subsidence calculated from this study suggest that drawdown evens should be managed for the shortest duration possible.

CHAPTER 5

INFLUENCE OF VEGETATION TYPE, TEMPERATURE, AND NUTRIENT LEVELS ON DECOMPOSITION OF SOIL ORGANIC MATTER

5.1 Introduction

Subsidence of soils in a peatland has been defined as the lowering of the soil surface elevation by Wosten et al (1997). Rates of subsidence vary strongly and depend on factors as type of peat, rate of decomposition, density, thickness of peat layers, mineral content, drainage depth and, climate. Processes leading to subsidence include consolidation, shrinkage, and oxidation with the latter one being the major contributor. Oxidation refers to the introduction of oxygen in the peat layer resulting in physical loss of organic matter due to increased rates of microbial decomposition. Even though the quantitative contribution of microbial respiration towards subsidence is unclear, it has been reported to be increasingly important in drained systems (Eggelman, 1976). However, others have shown respiration to have only a minor contribution in subsidence of soils (Glenn et al, 1993, Dirks et al, 2000).

Organic matter decomposition is a function of 1) substrate quality 2) hydrological regime that controls the aerobic/anaerobic condition, and 3) environmental factors as pH and temperature. The source of organic matter in peatlands is plant detritus, animal residues and other microbially synthesized organic compounds, that is mainly composed of the lignin, cellulose, hemicellulose, and phenolic material. Majority of these complex C compounds and polymers cannot be directly utilized by the microbial populations until they have been processed enzymatically to simple C compounds and monomeric forms. Oxidation of organic matter occurs in both aerobic and anaerobic conditions as wetland

soils are rich in functional microbial communities that are capable of utilizing a large variety of electron acceptors such as O_2, NO_3, SO_4, and Fe(III). The rate of decomposition is significantly higher under aerobic conditions (McLatchey and Reddy, 1998). DeBusk and Reddy (2003)showed that anaerobic mineralization of wetland soils occur at 1/3 rate of aerobic soils.

Soil temperature greatly influences primary productivity and organic matter decomposition. It increases primary productivity which leads to high detrital matter production and influences the microbial activity and enzyme activity; it increases loss of organic C. Vegetation is mostly the major contributor of the detritus material and therefore influences the C quality in soils. Nutrient levels in the water and sediments can change the dominant vegetation type in a particular system thereby affecting the C quality of the organic matter. Many studies have shown the relationship between OM decomposition and temperature (Reich and Potter, 1995; Howard and Howard 1993). These studies have empirically derived Q_{10} values that are specific for their study sites. Since temperature changes may involve a shift in microbial community composition (Zogg et al. 1997), shift the accumulation of secondary microbial products (Dalias et al. (2001a, 2001 b)) and may affect the interaction between OM and soil minerals (Thornley and Cannell, 2001), the C dynamics may be different under different temperature conditions.

The accumulation of the characteristic organic soils of the Upper St. Johns River Basin (USJRB) has taken several thousand years. The stability of these soils is highly dependent on hydrology; their formation is due to the historically extended hydroperiod of the USJRB floodplain wetlands. In some regions of the USJRB, the hydrology of the

floodplain has been dramatically altered, resulting in substantial loss of organic soil. The agricultural discharge has altered the nutrient levels in BCMCA which have resulted in altered vegetation communities. The annual temperature ranges from 3ºC (min) to 35ºC (max) (Melbourne being the closest location to BCMA, approximate annual temperature was determined using website: http://www.srh.noaa.gov/mlb/mlbclimat.html) and seasonal effects would change the SOM decomposition rates. This study was designed to determine the influence of temperature on C loss (as respiration) in this semi tropical peatland.

The objectives of this chapter were to examine carbon mineralization in SOM of different vegetation communities, that are representative of different nutrient levels, and to determine the effect of hydrology on C mineralization. In this study we investigated, by comparing SOM from regions that had different dominant vegetation communities (Chapter 2), using laboratory experiments, how rate of soil respiration is affected by temperature. The effect of C substrate quality on C respiration under different temperatures was also investigated. Rates of aerobic CO_2 , anaerobic CO_2 and CH_4 flux was measured.

5.2 Material and Methods

5.2.1 Site description and soil sampling.

Samples were collected on June 21, 2005 via airboat in a combined effort of Wetland Biogeochemistry Laboratory and St. Johns River Water Management District staff from the Blue Cypress Marsh Conservation Area (BCMCA) located in Central Florida in the headwater region of the St. Johns River in Indian River County. Five soil

sampling locations were established within the BCMCA to evaluate representative organic soils differing in their nutrient impact status and dominant vegetation communities. These representative sites include nutrient impacted areas dominated by *Typha domingensis* (site A) and *Salix caroliniana* (site E) as well as, non-impacted sites with vegetation communities dominated by slough vegetation such as *Nymphea odorata* and *Eleocharis interstincta* (site B). Wet prairie ecotypes are represented with sites dominated by *Cladium jamaicense* (site C) and *Panicum hemitomon* (site D). Soil samples (0-10 cm depth) were obtained from the 5 sites. Soil physico-chemical parameters shall be determined including soil bulk density, total P (TP), total C (TC) and total N (TN).

5.2.2 Soil Physio-chemical characterization

Results of the physical and chemical characterization efforts are summarized in Table 5.1 Bulk density was highest in the *Salix* and *Panicum* sites, and lowest in *Typha* and *Nymphea* sites. Soil pH was similar at all sites except the *Nymphea* site, which was noticeably low in comparison. Soil TC was similar at all sites, however, TN was more variable (range = 27.9-37.2 g/kg) with the *Panicum* site being highest and the *Cladium* site the lowest. TP was found to be highest in the *Salix* and *Typha* sites and relatively lower in the interior sites. Ash contents were similar among sites ranging from 93.1% to 95.6%.

Table 5.1 Physical and chemical characterization of Blue Cypress Marsh soils and soil organic matter. Values represent means of five replicates (±1 standard deviation)

Vegetation community Site	BD $g\ cm^{-3}$	pH	TC $g\ kg^{-1}$	TN $g\ kg^{-1}$	TP $mg\ kg^{-1}$	LOI %	Water soluble %SOM	Hemicellulose %SOM	Cellulose %SOM	Lignin %SOM
Eleocharis	0.070	5.8±0.1	461±4	28.1±0	622±30	93.2±0.4	16±1	21±2	35±3	28±5
Typha	0.069	5.4±0.1	478±4	33.3±1	584±34	95.6±0.5	11±2	15±4	36±4	39±7
Salix	0.075	5.6	467±7	27.9±1	540±25	94.8±0.9	7±1	13±2	18±6	62±4
Panicum	0.084	5.8±0.1	468±2	37.2±1	538±73	93.1±0.3	8±2	9±1	25±2	58±5
Cladium	0.091	5.8±0.1	481±9	29.7±1	851±60	94.6±0.6	5±3	10±3	30±5	55±5

5.2.3 *Microcosm experiments*

We examined the effect of various temperatures on aerobic and anaerobic C mineralization in soils from five sites that included different dominant vegetation. For both aerobic and anaerobic incubations, three replicates were set up for each soil type under each temperature (10ºC, 20ºC, and 30ºC) treatment.

Aerobic microcosms were prepared by adding 1 gram (dry wt) of soil samples in 60 ml glass serum bottles. For aerobic incubations prior to weighing out, moisture content in soils was adjusted to 65- 75%. Bottles were left uncapped during incubation and were covered with a moist light absorbent cloth to prevent the samples from drying out. Bottles were capped with gray butyl stoppers and aluminum crimps and flushed with hydrocarbon free air (<1 ppm CO_2), 16 hours prior to gas measurement. CO_2 gas was sampled on 4, 6, 10, 14, 21, and 28 days.

Anaerobic microcosms were prepared by weighing out approximately 5 gram of wet weight soil in a 120 ml glass serum bottle and adding 10 ml of sterile double distilled water. Bottles were sealed with butyl stoppers and aluminum crimps before being purged with N_2 gas to create anoxic conditions in the head space. Unlike the aerobic microcosms, the anaerobic samples remained capped throughout the experiment and the samples were N_2-flushed after each gas measurement to remove any residual CO_2 and CH_4 in the bottle. CO_2 and CH_4 gas sampling were at 3, 6, 10, 14, 21, and 28 days.

Two soil-free controls were included with every set of microcosm to account for background CO_2 production, which were negligible compared to soil respiration. Moisture content of soil samples was determined prior to establishing the incubation

experiment by oven drying soil sub samples at 70ºC for 72 hours. Soil samples were weighed after each gas measurement to account for any water loss in the samples.

5.2.4 *Carbon dioxide and methane measurements*

Gas pressure in each bottle was measured during every analysis using a pressure meter (Kane-May, Great Britain). CO_2 gas samples were separated and analyzed using a Carboxen 1000 (Supelco, Bellefonte,PA) column fitted in a Shimadzu 8A gas chromatograph (GC) (Shimadzu Scientific Instruments Inc., Columbia, MD) equipped with a thermal conductivity detector (TCD). Temperature conditions were set as 110ºC (injection port and detector) and160ºC column. Calibration curves were prepared using Standard CO_2 gas (Scotty Specialty Gases, Plumsteadville, PA).

Methane production rates were measured simultaneously with CO_2 in anaerobic incubations. CH_4 was analyzed using a Shimadzu 8A GC fitted with a Poropak N column (Supelco, Bellefonte, PA) and equipped with a flame ionization detector. Temperature conditions were set as 110ºC (injection port and detector) and160ºC column. Calibration curves were prepared using standard CH_4 gas (Scotty Specialty Gases, Plumsteadville, PA).

5.2.5 *Calculations*

Gas measurements were calculated by using the equation PV=nRT where P is the pressure in the head space of the bottle, V is the volume, n is the number of moles of gas produced, R is the gas constant, T is the temperature in Kelvin. Cumulative curves were prepared by sampling the gas produced at certain time period. Rates of CO_2 production (k) were the slopes of the linear portion of the cumulative curves.

Responses of biological systems are often expressed as a Q_{10} function

$$Q_{10} = \left(\frac{k_2}{k_1}\right)^{\left(\frac{10}{T_2 - T_1}\right)}$$

where k_1 and k_2 are rate constants for C mineralization at two observed temperatures t_1 and t_2. For instance, if rate of decomposition is occurring twice as fast at 30ºC as at 20ºC, it would have a Q_{10} of 2.

5.2.6 *Statistical analyses*

Differences between vegetation types and temperature treatments were tested with two way ANOVA (JMP, SAS Inst). Net CO_2 and CH_4 production data was log transformed prior to data analysis. All difference reported were significant at alpha= 0.05.

5.3 Results

Effect of temperature on anaerobic and aerobic decomposition rates of soil organic matter from five different vegetation communities in BCMCA was studied using laboratory microcosms. Total C released as CO_2 and CH_4 during decomposition was determined for a short term period of 21 days where aerobic decomposition released 4-6 times higher C content than the anaerobic decomposition (table 5.5, 5.6, 5.7). CO_2 and CH_4 emission rates showed a significant increase with temperature (table 5.1, 5.2, 5.3). Organic matter from different vegetation communities decomposed at different rates and the rates of decomposition responded differently to the increased temperature.

5.3.1 *Effect of vegetation communities on carbon mineralization*

Fiber analysis conducted on soil subsamples from 5 sites with different dominant vegetation (Chapter 2), revealed different fractions of labile, moderately labile and recalcitrant pools with the sites. *Typha* and *Nymphaea* dominated sites had higher labile organic matter. The *Cladium* and *Panicum* dominated sites showed similar amounts of labile organic materials, less than *Typha* and *Nymphea* sites, but more than the *Salix* dominated site. *Cladium, Salix* and *Panicum* sites also contained the highest fraction of lignin. Short term decomposition studies (CO_2 evolution) in laboratory microcosms indicated that *Nyphaea* and *Eleocharis* sites had high labile fraction as evidenced by the high rate of CO_2 production. Decomposition rates were temperature dependent and rates of CO_2 production increased with temperature.

Cellulose and hemicellulose together represent a moderately labile pool. *Typha* and *Nymphea* and *Eleocharis* sites have the largest pools of this material. However, *Cladium* has the lowest amount of this material with *Salix* and *Panicum* being intermediate and similar in value. The refractory pools were found to be highest in the *Cladium* site followed by *Panicum* and *Salix* respectively. *Typha* and *Nymphea* had the lowest values of refractory organic matter.High lignin content in *Cladium, Panicum* and *Salix* sites indicated higher refractory C, followed by *Typha* and *Nymphea* sites that had low levels of lignin but had the highest content of cellulose. The most labile fraction, the soluble soil constituents, were highest in *Typha,* and *Nymphea,* the other species being significantly lower in this labile fraction, but similar to each other.

Two different rates of CO_2 production were observed in the aerobic decomposition of soils (Table 5.1, 5.2, Figure 5.1, 5.2, 5.3). Initial rate of CO_2 production (k_1) was observed in the first week of incubation and was three times higher than the final

rate of CO_2 production (k_2) than that observed in the second, third and fourth week of incubation.

Q_{10} values for the first rate (k_1) of CO_2 production ranged from 1.3 to 1.5 and 1.9 to 2.9 for lower range (10ºC-20ºC) and higher range (20ºC-30ºC) temperatures respectively. For the final stage of CO_2 evolution (k_2), Q_{10} values for higher temperature were lower than those calculated for lower range temperature in all plants, except *Typha*. Increasing temperature did not affect rate of OM decomposition of all the soils in the same manner. Highest response to increasing temperature in the form of increased rate of CO_2 production was observed in *Typha* soils followed by *Nymphaea* & *Eleocharis* and, *Salix*. In *Typha* soils, total CO_2 produced increased 1.4 and 3.4 times when incubation temperature was raised from 10ºC to 20 ºC and 30ºC respectively whereas other soils showed an increase from 1.4 to 2.0. At 10ºC, largest amount of CO_2 was evolved in 4 weeks from *Nymphaea* & *Eleocharis* soils and *Panicum* soils 4 weeks followed by *Cladium*, *Salix* and *Typha*. (Table 5.5). Similar trend was observed at 20ºC, however, at 30ºC, *Typha* soils released the maximum amount of CO_2 followed by *Panicum*, *Nymphaea* and *Eleocharis*, *Cladium* and *Salix*.

Decomposition of soils under anaerobic conditions produced CO_2 and CH_4 at a constant rate for 28 days (Table 5.3 5.4, Fig 5.4-5.9). Highest CO_2 produced by the *Nymphaea* and *Eleocharis* soils followed by *Typha*, *Panicum* and *Cladium*. Under anaerobic conditions, lowest amount of CO_2 was released by *Salix* (Table 5.6). With increase in temperature, there was an increase in the amount of CO_2 produced at the end of 28 days. Rate of anaerobic CO_2 production was 10 times lower than the aerobic conditions. The Q_{10} values remained the same for *Cladium* and *panicum* soils and

increased for *Typha*, *N&E* and *Salix* soils. CH_4 production was the highest in *Nymphaea* and *Eleocharis* soils followed by *Salix*, *Typha*, and *Panicum*. *Cladium* produced the lowest CH_4 at all the three incubation temperatures. Total amount of CH_4 produced at 10ºC increased 0.5 and 4 times when the temperature was increased to from 20ºC to 30ºC (Table 5.7). Rate of CH_4 production did not increase much when the temperature was increased from 10ºC to 20ºC, however, there was a 4-6 times increase with the increase in temperature. The Q_{10} values for CH_4 ranged from 3.8 to 6.0 with the highest one being or *Cladium*. *Nymphaea* and *Eleocharis* showed low Q_{10}. Initial low rate of CH_4 production indicated a lag period for the stimulation of the methanogenic populations.

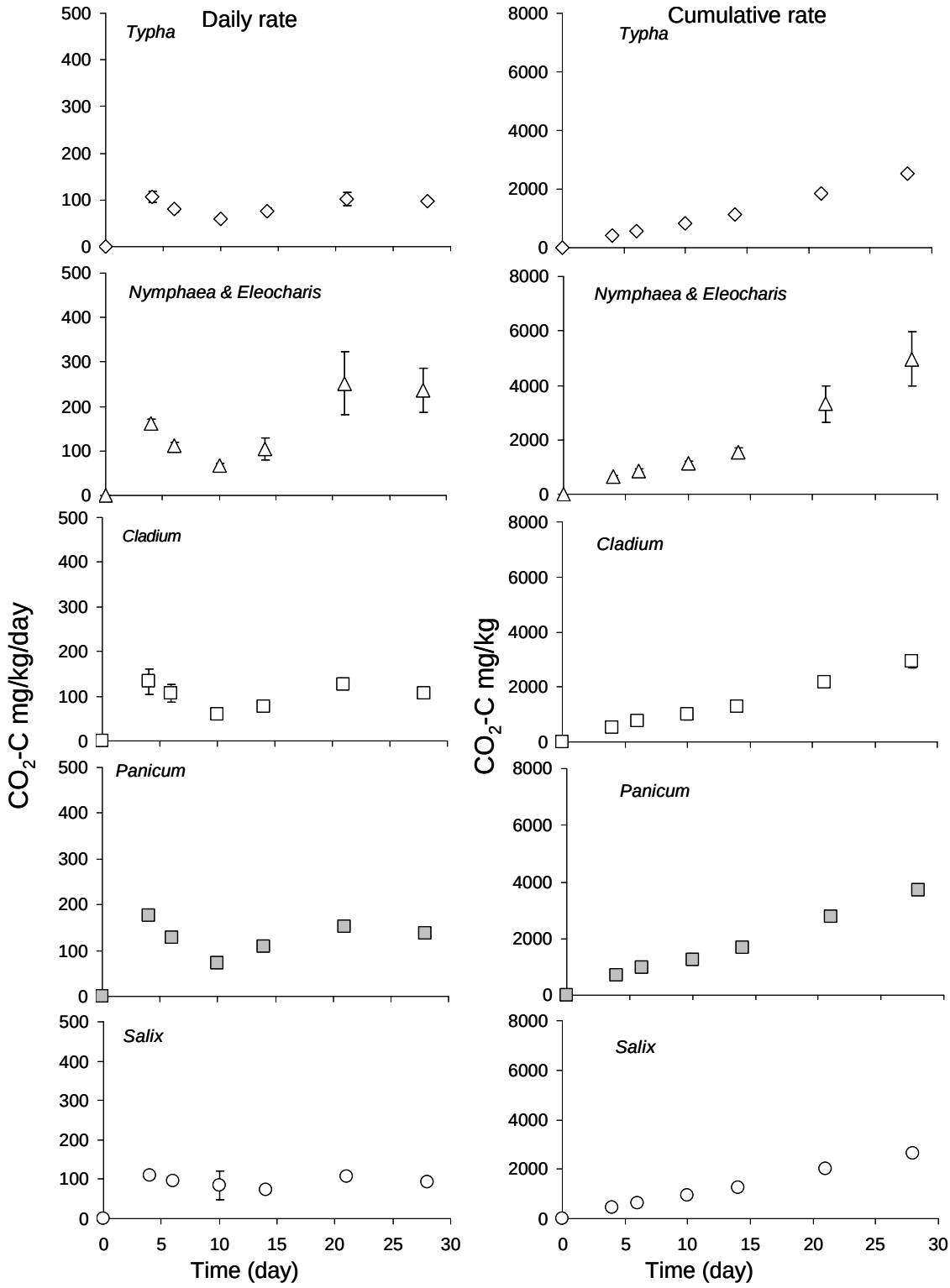

Fig 5.1 Rates of CO_2 production for soil samples during aerobic incubation at 10ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils..

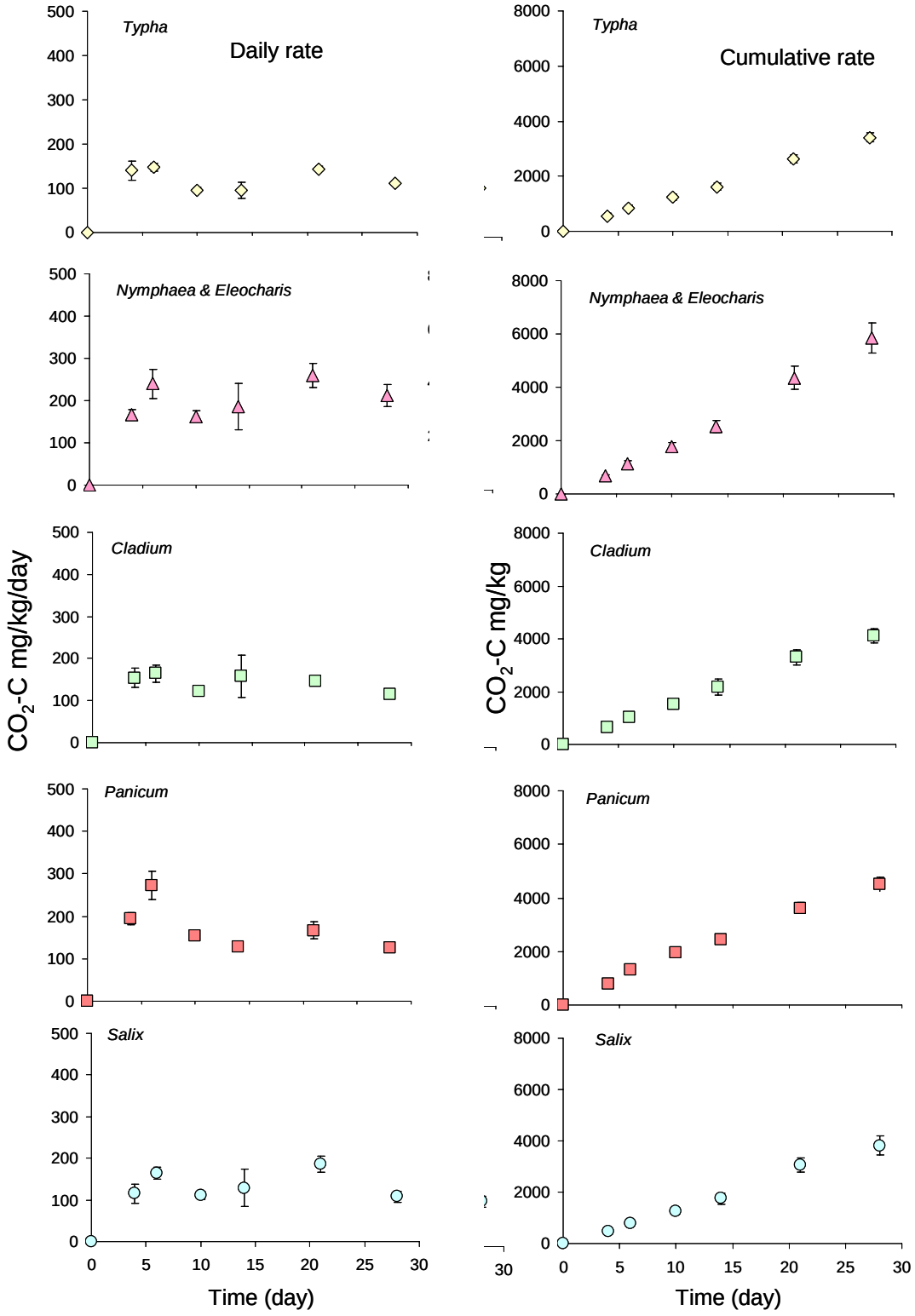

Fig 5.2 Rates of CO_2 production for soil samples during aerobic incubation at 20ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils..

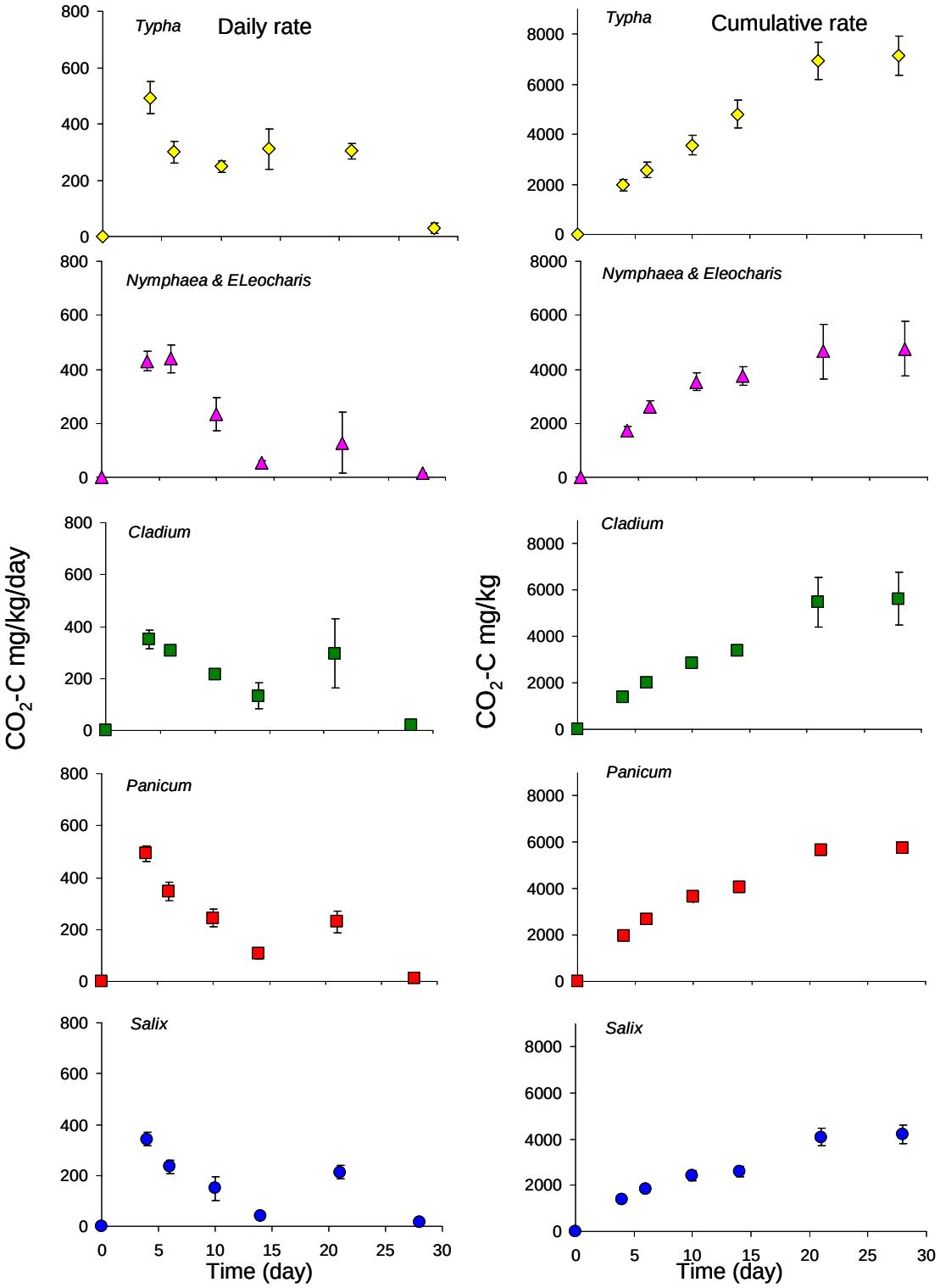

Fig 5.3 Rates of CO_2 production for soil samples during aerobic incubation at 30ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

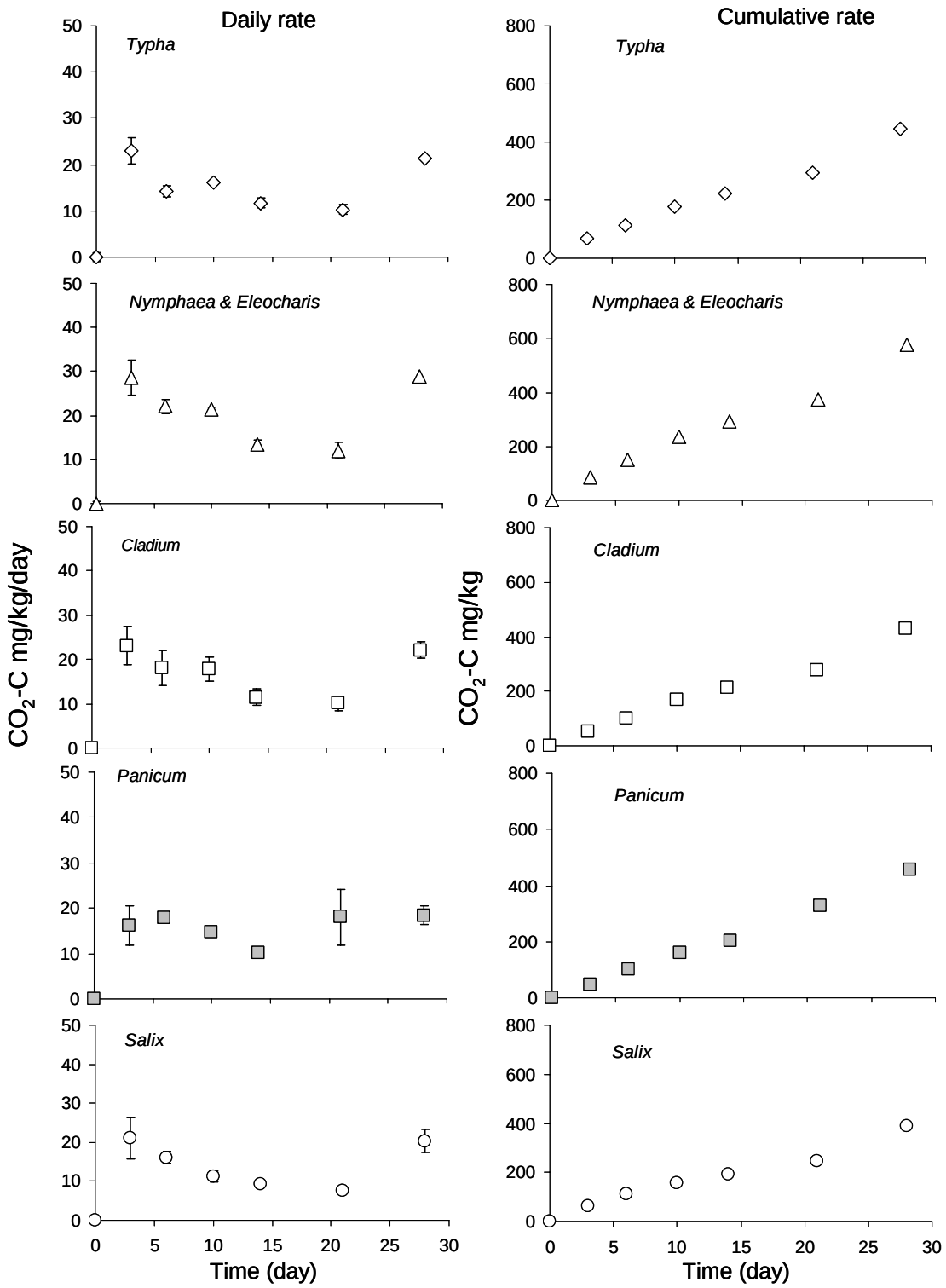

Fig 5.4 Rates of CO_2 production for soil samples during anaerobic incubation at 10ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

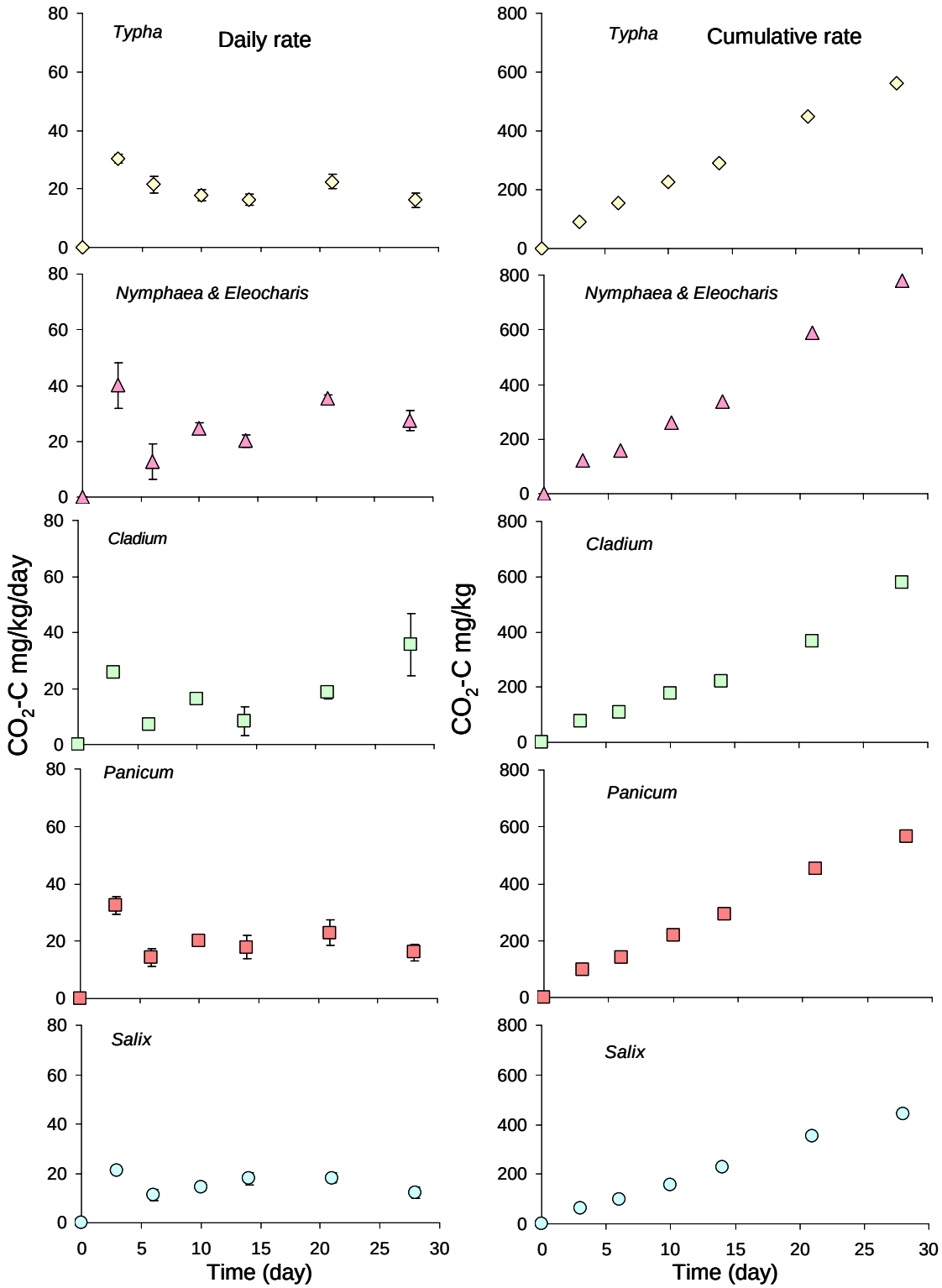

Fig 5.5 Rates of CO_2 production for soil samples during anaerobic incubation at 20ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

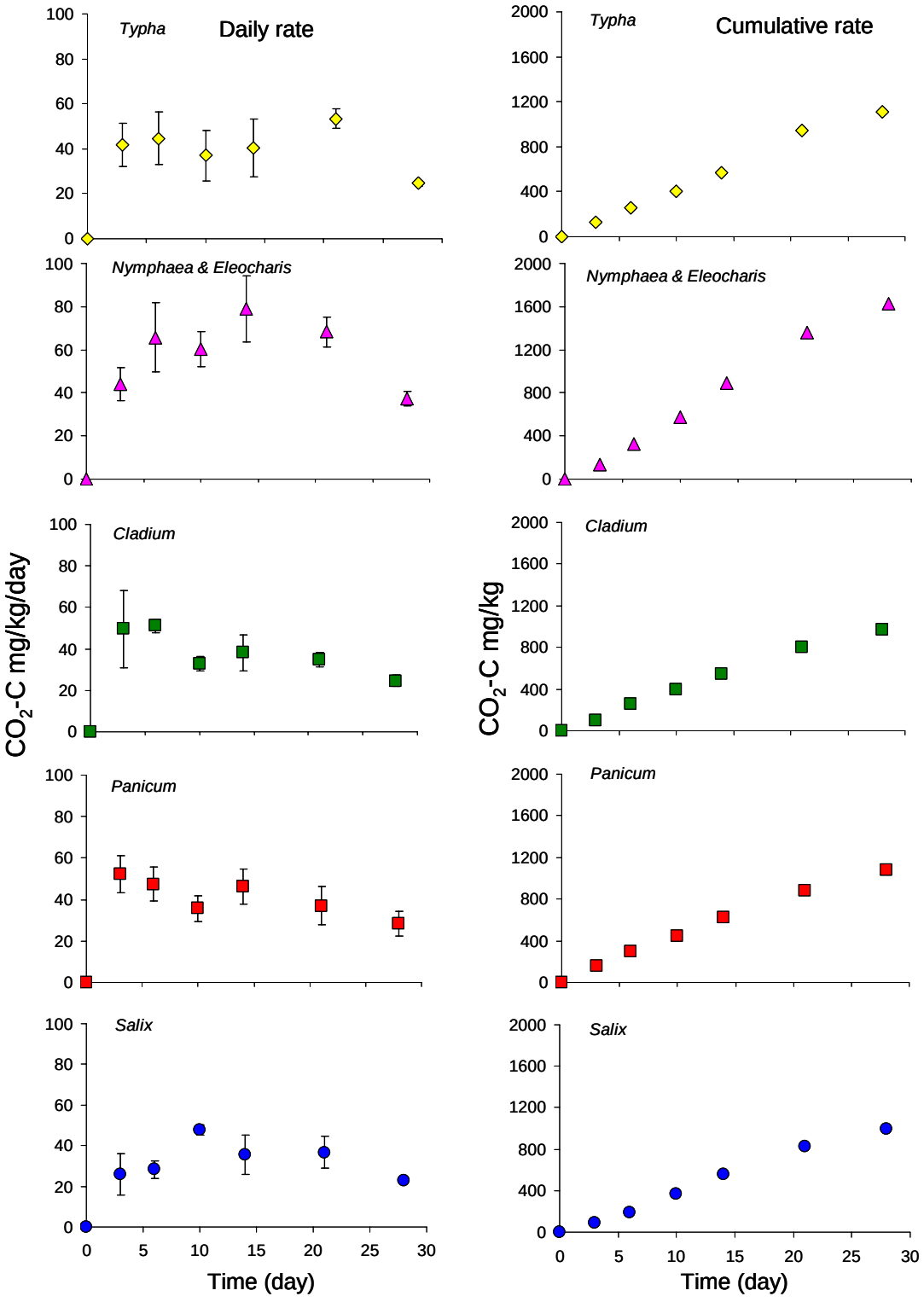

Fig 5.6 Rates of CO_2 production for soil samples during anaerobic incubation at 30ºC. Rates of CO_2 production at each sampling period (a,b,c,d,e) and cumulative CO_2 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

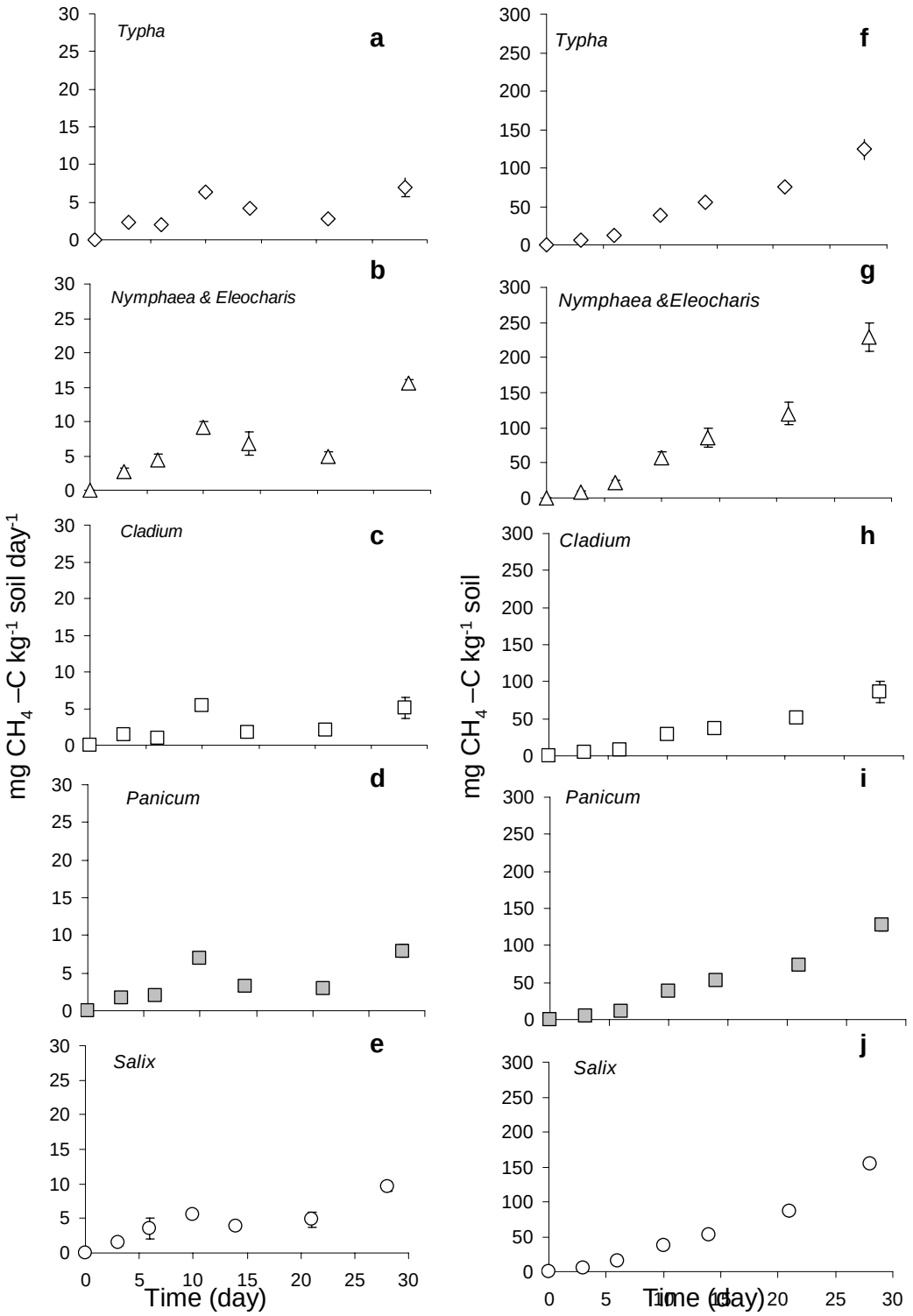

Fig 5.7 Rates of CH_4 production for soil samples during anaerobic incubation at 10ºC. Rates of CH_4 production at each sampling period (a,b,c,d,e) and cumulative CH_4 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

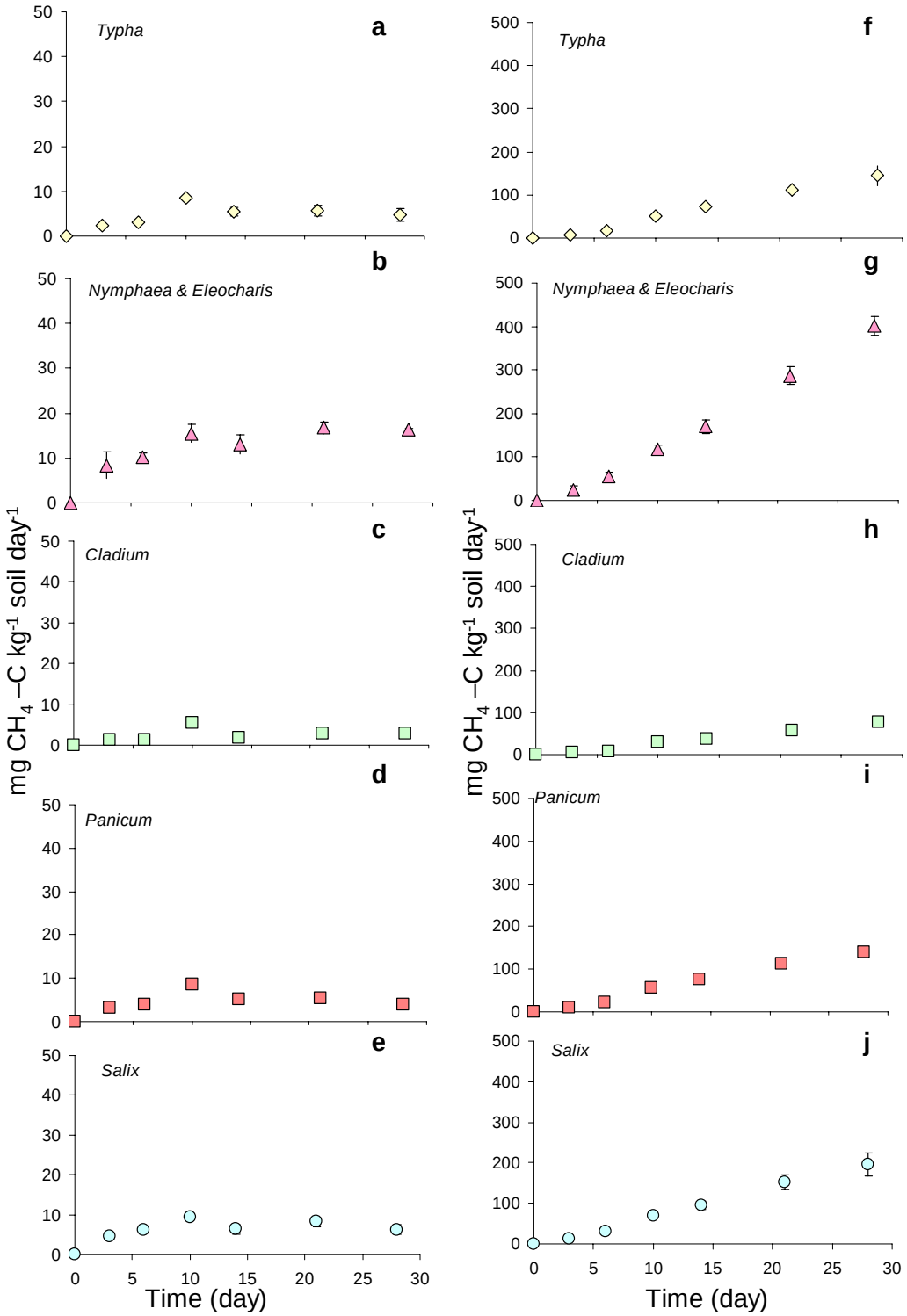

Fig 5.8 Rates of CH_4 production for soil samples during anaerobic incubation at 20ºC. Rates of CH_4 production at each sampling period (a,b,c,d,e) and cumulative CH_4 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.

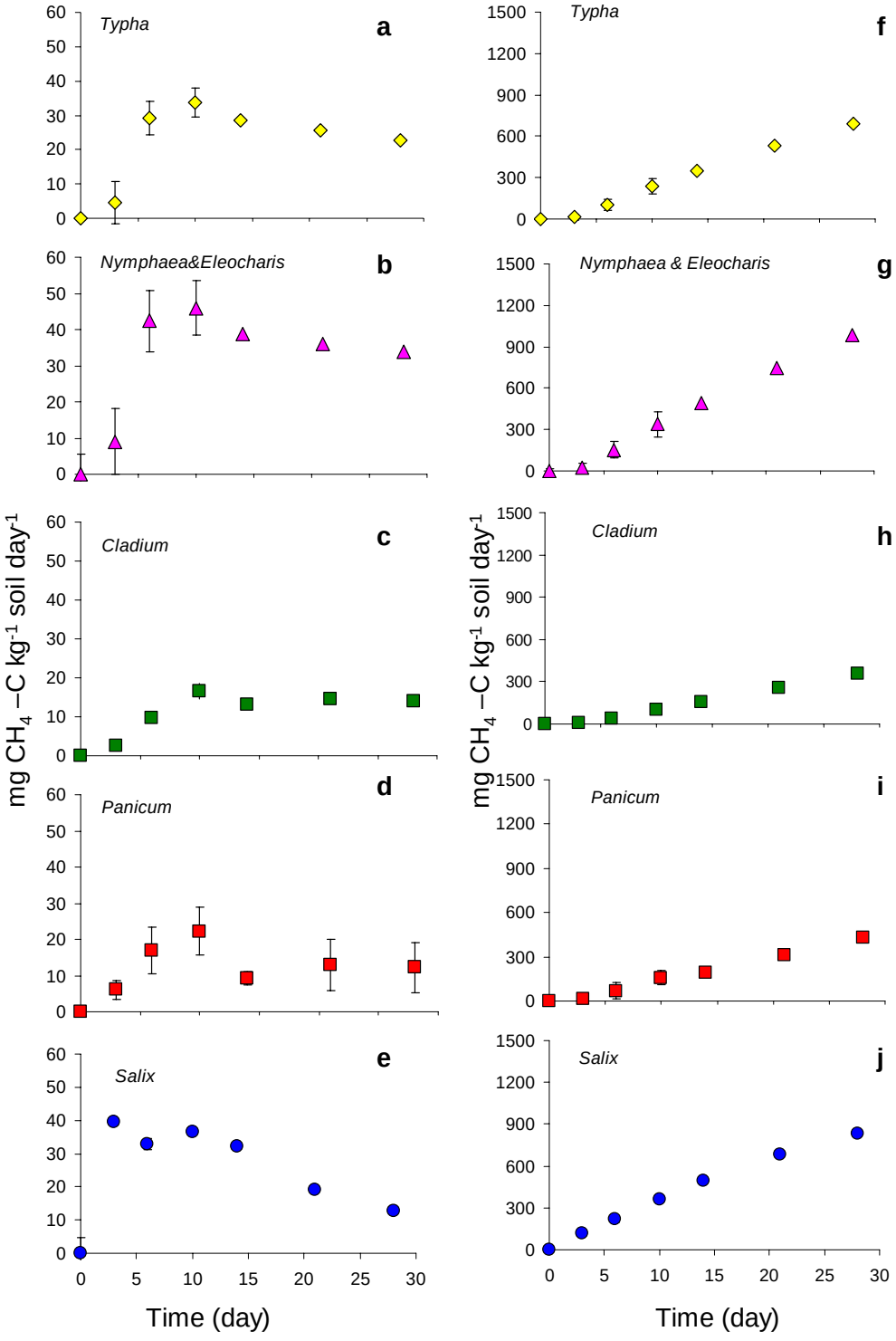

Fig 5.9 Rates of CH_4 production for soil samples during anaerobic incubation at 30ºC. Rates of CH_4 production at each sampling period (a,b,c,d,e) and cumulative CH_4 produced (f,g,h,i,j) in *Typha* soils, *Nymphaea* and *Eleocharis* soils, *Cladium* soils, *Panicum* soils, and *Salix* soils.Fig 6.9 Anaerobic Ch4 30ºC

Table 5.1 . Rate of respiration (k) and Q_{10} values during aerobic incubation of soil subsamples from following vegetation dominated sites in Blue Cypress Marsh. Rate of CO_2 production was calculated over a period of 21 days.

	CO_2-C mg/kg soil/ d			Q10	
	10°C	20°C	*30°C	10°C-20°C	20°C-30°C
Typha	84.0	121.0	353.9	1.4	2.9
Nymphaea and Eleocharis	146.1	204.3	356.8	1.4	1.7
Cladium	97.8	154.9	287.9	1.6	1.9
Panicum	128.2	168.5	363.5	1.3	2.2
Salix	123.0	190.8	324.5	1.6	1.7

*rates were measured over a period of 10 days.

Table 5. 2. Rate of respiration (k) and Q_{10} values during anaerobic incubation of soil subsamples from following vegetation dominated sites in Blue Cypress Marsh. Rate of CO_2 production was calculated over a period of 21 days.

	CO_2-C mg/kg soil/ d			Q10	
	10°C	20°C	30°C	10°C-20°C	20°C-30°C
Typha	14.8	38.6	43.9	2.6	1.1
Nymphaea and Eleocharis	19.0	27.2	66.1	1.4	2.4
Cladium	14.4	16.4	38.4	1.1	2.3
Panicum	15.9	19.9	41.7	1.3	2.1
Salix	16.7	21.1	54.1	1.3	2.6

Table 5. 3. Rate of CH_4 production (k) and Q_{10} values during anaerobic incubation of soil subsamples from following vegetation dominated sites in Blue Cypress Marsh. Rate of CH_4 production was calculated over a period of 21 days.

	CH_4-C mg/kg soil/ d			Q10	
	10°C	20°C	30°C	10°C-20°C	20°C-30°C
Typha	4.0	5.7	28.9	1.4	5.1
Nymphaea and Eleocharis	6.5	15.7	40.0	2.4	2.5
Cladium	2.7	2.9	17.6	1.1	6.1
Panicum	4.0	5.3	16.2	1.3	3.1
Salix	4.6	7.4	22.8	1.6	3.1

Table 5.5 Total CO_2 -C produced during aerobic incubations (21 days) at various temperatures from soils collected from Blue Cypress Marsh. Values represent the mean (n=3) with standard error.

	CO_2-C g /kg soil		
	10°C	20°C	30°C
Typha	1.8±0.1	2.6±0.1	6.9±0.7
Nymphaea and Eleocharis	3.3±0.7	4.4±0.4	4.7±1.0
Cladium	2.2±0.2	3.3±0.3	5.5±1.1
Panicum	2.8±0.1	3.6±0.2	5.7±0.2
Salix	2.7±0.2	4.1±0.3	5.5±0.4

Table 5.6 Total CO_2 -C produced during anaerobic incubations (21 days) at various temperatures from soils collected from Blue Cypress Marsh. Values represent the mean (n=3) with standard error.

	CO_2-C mg /kg soil		
	10°C	20°C	30°C
Typha	296±7	450±37	942±72
Nymphaea and Eleocharis	375±22	587±36	1364±191
Cladium	277±33	364±52	799±75
Panicum	328±32	453±59	885±67
Salix	247±13	355±31	827±28

Table 5.7 Total CH_4 -C produced during anaerobic incubations (21 days) at various temperatures from soils collected from Blue Cypress Marsh. Values represent the mean (n=3) with standard error.

	CH_4-C mg /kg soil		
	10°C	20°C	30°C
Typha	75±5	112±14	533±83
Nymphaea and Eleocharis	120±16	287±20	746±125
Cladium	51±6	57±3	339±60
Panicum	73±3	113±5	314±91
Salix	116±10	152±19	512±176

Table 5.8 Rates of C mineralization under aerobic and anaerobic conditions in different vegetation community organic matter. For comparison purposes, data represents values obtained by organic matter incubation at 20ºC.

Vegetation community	Aerobic	Anaerobic	
	CO_2-C mg/kg/d	CO_2-C mg/kg/d	CH_4-C mg/kg/d
Typha.	121.0	38.6	5.7
Nymphaea & Eleocharis	204.3	27.2	15.7
Cladium	154.9	16.4	2.9
Panicum	168.5	19.9	5.3
Salix	190.8	21.1	7.4

5.4 Discussion

Temperature effects on decomposition of organic matter.

Microorganisms are the key players in C-mineralization during SOM oxidation. Complex polymeric compounds, that are major constituents of SOM, have to be transformed into simple monomeric forms before they can be utilized by the microorganisms. Transformation to these bioavailable forms is carried out by the microbial enzymatic processes (eg. hydrolases, oxygenases etc.). Higher temperatures favor the enzyme processes and the microbial growth rates and therefore, increase the SOM decomposition rate is observed with increase in temperature. This study showed increased CO2 production in SOM of all vegetation communities. These results are consistent with observed in other studies (Lloyd and Taylor, 1994; Kirchbaum, 1995, 2000; Liski et al, 2003). However, our results showed that rate of increase in decomposition rates is not the same for all vegetation soil type. In aerobic respiration, with exception of *Typha*, there were two rates of decomposition observed in SOM from all vegetation communities at 30ºC. The initial rate, observed in the first week of SOM decomposition, was 3-4 times higher than the final rate that remained constant for rest of the incubation period of 2 weeks. For lower temperatures, the rates of Co2 production remained constant through the whole period of incubation. This indicated that at higher temperatures the first rate of CO2 production was from the labile C pool that was readily bioavailable for microorganisms to use. The labile C pool was rapidly oxidized and then the other C pool, perhaps a relatively less bioavailable pool was being oxidized. At lower temperatures, the rates of CO2 production were constant for 21 days. This effect of temperature may be explained by the higher enzymatic activities that are responsible for

the breakdown of the larger polymers and the cellulosic material. Total amount of CO_2 produced in 21 days at 10ºC was lower than that produced within a short period of 7 days at 30ºC. It is generally believed that the response of decomposition is more sensitive at lower temperature than at the higher temperatures (Kirchbaum, 1995) and some studies have shown that the temperature does not effect the carbon efflux processes(Giardina and Ryan, 2000). However, results from this study indicated that was not the case. The Q_{10} values showed an increase with temperature indicating that rate of decomposition increases with the change in temperature as the temperature is increased from 10ºC, to 20ºC and to 30ºC. Bol et al (2003) also reported similar results when they found the scale of difference between CO2 production between CO2 production at 35DC and that at 20ºC is significantly greater than difference in production at 20ºC and 10ºC. Higher range of temperature. Q_{10} values for all vegetation soil types ranged from 1.2-1.4 when temperature was increased from 10 to 20 indicating the temperature had the same effect on all soils OM decomposition. However, for *Typha*, Q_{10} value increased from 1.4 to 3.1at 10-20 and 20-30ºC temperature increase. This indicated much higher increase in decomposition in soils at higher range temperature. A likely explanation for this observation is the carbon quality of SOM from this vegetation type. Higher microbial activity at higher temperature would increase the CO2 production. However at this time it is not clear if the higher temperature affected the C quality of the SOM abiotically and SOM becomes less refractory and more bioavailable for microbial utilization. This alternative explanation cannot be ignored. For all other soil types, the second pool of carbon was perhaps not available or not as refractory.

Nymphaea and *Eleocharis* soil showed an increased Q_{10} value for the initial rate. However, the Q_{10} value for 20-30ºC was halved when compared with the Q_{10} value for 10-20ºC. This indicated that at higher temperature, most carbon was respired in the first few days and only a small portion of C pool that could be respired was respired in the second half.

Temperature effects on anaerobic decomposition of organic matter.

Anaerobic CO_2 and CH_4 production increased with the increase in temperature indicating that increase in temperature increases the C loss both aerobically and anaerobially when the temperature is raised. Unlike in the aerobic decomposition, the anaerobic decomposition rate, anaerobic CO2 production, remained the same all through the 28 day incubation period. Highest amount of CO_2 and CH_4 was produced in *Nymphaea* and *Eleocharis* soils indicating that the C substrates produced in the soils could be used by the anaerobic communities of soils. C lost in anaerobic decomposition as CO_2 versus CH_4 was high in *Panicum* and *Cladium* , followed by *Typha* and *Nymphaea* and *Eleocharis*. *Salix* soils produced 53% of total C as CO_2 and 46% as CH_4. These results suggested that the methanogenic communities associated with *Salix* soils are perhaps the highest or that this soils provides a better substrate to support the communities. In contrast, *Cladium* and *Panicum* substrates either create conditions that are not conducive for supporting these methanogenic communities like the C substrates or the anaerobic conditions. By comparing the total C evolved in aerobic decomposition, the C evolution as anaerobic CO_2 and CH_4 increased with the increase in temperature in all vegetation soils except in *Typha*. This indicated that the temperature did play a major

role in releasing the labile form of carbon either by increasing the enzyme efficiency or by abiotic means.

Observed Q_{10} values for anaerobic CO_2 production remained the same for both low and high range of temperatures for *Cladium* and *Panicum* soils. This indicated that increase in temperature did increase the decomposition of SOM and the increase may be due to the microbial enzyme activity. *Typha, Nymphaea* and *Eleocharis,* and *Salix* soils showed an increase from 1.4,1.3, and 1.3 to 2.0,2.1,2.2 respectively indicating that the increase in rate of production of anaerobic CO_2 was higher when increase in temperature was in the higher range. This can be explained by presence of material that requires enzymes that are highly efficient in the higher temperatures alternatively the temperature increase increases the labile forms of carbon that can easily be mineralized under anaerobic conditions.

Q_{10} values for methane production did not follow the same pattern as the CO_2. CH_4 production rates were higher for all vegetation soils for higher temperature range. *Cladium* showed Q_{10} values of 0.9 and 6.0 for 10-20ºC and 20-30ºC indicating that the C loss increased by a large factor when the temperature was raised from 20 to 30. This suggests that the methanogenic community was more active at the higher temperature either due to the enzyme activity or the presence of substrate. Low difference in Q_{10} values for *N*&*E* indicated that at 20ºC the labile C content was higher or the methanogenic community was more active than at 10ºC and then by increasing the temperature to 30ºC there was an increased rate in methane production but it was much lower than the rate increase seen in *Typha* soil, *Panicum* soil, and *Salix* soils. Total C evolved as methane was highest in *Nymphaea* and *Eleocharis* and lowest in *Cladium* at

all temperatures 10ºC, 20ºC, and 30ºC. Increased CO2 respiration and lower CH4 production indicated the presence of electron acceptors in soils.

5.5 Conclusions

Water table drawdown greatly affected C mineralization in organic matter from all five vegetation communities. Higher organic matter decomposition, as evidenced by higher rate of CO_2 evolution from soils, during aerobic/ non flooded conditions led to higher C loss when compared with decomposition during anaerobic/ flooded conditions. Total C mineralization under anoxic conditions was 4 to 6 times lower than that under oxic conditions.

Decomposition rates of soil organic matter from regions with different vegetation communities varied as a response to the water table drawdowns. At 20ºC, organic matter from slough vegetation communities like *Nympahea* and *Eleocharis* spp. were more susceptible to drawdowns as indicated by the highest rates of aerobic decomposition. C loss from soils with *Typha* as the dominant vegetation community was relatively lower than all other vegetation communities we studied. Rate of organic matter decomposition in *Nymphaea* and *Eleocharis* was higher than other vegetation communities indicating relatively higher labile organic matter. *Cladium* and *Salix* vegetation communities appeared to have more recalcitrant organic matter as evidenced by the low rates of decomposition (aerobic and anaerobic).

C loss from regions of different vegetation communities is also dependant on temperature. At lower temperature (10ºC) relative rates of aerobic decomposition in soils from all vegetation communities were similar to that observed at 20ºC however, the rates

of decomposition was 1.3 to 1.6 times lower. At 30ºC, rate of aerobic decomposition of soils with different vegetation communities declined after the initial period of 6-10 days. Rate of organic matter decomposition was highest in *Panicum*, and lowest in *Cladium*. Highest change in the rate of decomposition of soil organic matter, as a response to temperature, was observed in *Typha* vegetation community. Q_{10} values for all vegetation communities indicated that higher range temperature (20ºC-30ºC) affected rates of decomposition in all vegetation communities more than that observed in the lower range temperatures. (10ºC-20ºC).

Higher temperatures increased the rates of CH_4 production during anaerobic decomposition in all vegetation communities. However, increase in rates of CH_4 production was higher than that observed in CO_2 production in either aerobic or anaerobic incubation. Relative increase in CH_4 production to anaerobic CO_2 production was observed at higher temperature. This disproportionate increase in methane production may be explained by either substrate increase, change in microbial community structure, or increased efficiency of enzymatic reactions involved in methanogenesis.

Therefore, results from this study indicate that C loss from regions of Blue Cypress Marsh with different vegetation communities varies as a response to water table drawdowns and temperature changes. Higher temperatures, and drawdown conditions will greatly increase C loss from regions with *Typha* vegetation communities. At relatively moderate temperatures more carbon would be lost from soils that are in the *Nymphaea* and *Eleocharis* regions when drained.

CHAPTER 6

SUMMARY AND CONCLUSIONS

The accumulation of the characteristic organic soils of the Upper St. Johns River Basin (USJRB) has taken several thousand years. The stability of these soils is highly dependent on hydrology as their formation is due to the historically extended hydroperiod of the USJRB floodplain wetlands. In some regions of the USJRB, the hydrology of the floodplain has been dramatically altered, resulting in substantial loss of organic soil. Knowledge of the critical water depth at which accelerated soil loss occurs is needed to refine estimates of marsh water levels which are protective of the soils of the region. The Environmental Water Management Plan for The Upper St. Johns River Basin (1996) calls for a "...mean depth and inundation frequency of the central critical elevation such that there will be no net loss of organic soils through oxidation." This is one element of a set of hydrologic criteria that includes frequency of inundation, maximum depth, magnitude of annual fluctuation, timing of fluctuation, and water level recession rates. Environmental hydrologic criteria that numerically describe each of these characteristics are currently being developed and refined. These criteria will ultimately be used to direct the operation of project structures when water levels are below established flood control regulation schedules.

The central goal of this research was to determine the minimum water levels in wetlands of the Blue Cypress Marsh Conservation Area (BCMCA) needed to prevent net loss of organic soils from oxidation. This was achieved by investigating the effect of various water drawdown scenarios on soils with different vegetation type, temperature and nutrient levels. The effect was determined by measuring microbial activities such as

aerobic and anaerobic respiration, CH_4 production, phenol oxidase activity and b glucosidase enzyme activities.

The objectives of this study were to review historic studies on the subject of soil subsidence and summarize this information for application within the scope of this research; characterize the organic soils found in the BCMCA both physically and chemically by standard methods, as well as, identifying the lability of different pools of soil organic matter; determine the effect of water level variation on soil organic matter oxidation rates; determine the influence of inundation frequency on soil organic matter decomposition; and finally, determine the influence of dominant vegetation type, temperature, and soil nutrient levels on soil organic matter decomposition.

The first objective was to review the pertinent literature available on the subject of soil subsidence with special focus on organic soils in subtropical climates. A majority of the literature available for subtropical climates focuses on Everglades soils and agricultural organic soils of the Everglades Agricultural Area. Much research was conducted there as a result of massive subsidence observed over the last 60 years. In places, over 2.5 meters of soil have been lost to subsidence. Organic soil subsidence is the result of over drainage and involves three major processes; oxidation of organic matter, loss of soil buoyancy and shrinkage of organic soils when water is removed. Loss of soil due to microbially mediated oxidation of soil organic matter under aerobic conditions is the most damaging effect of over drainage as the loss of organic matter is terminal. The major factors influencing organic matter oxidation are organic matter lability (food quality from a microbial perspective), soil moisture, oxidation/reduction potential, nutrient availability, temperature, and microbial communities/activities.

The second objective was to characterize the BCMCA soil organic matter from five sites with various dominant vegetation and nutrient enrichment status by qualitative and quantitative means. Investigation of soil physical and biogeochemical parameters, as well as, organic matter lability were used in this characterization.

Sites dominated by the monotypic species *Typha* and *Salix* were indicative of areas of BCMCA that have been effected by nutrient enrichment. Sites in areas dominated by these vegetation types were the two highest sites observed with respect to soil phosphorus content. All sites were similar in value with respect to LOI, pH, and TC. TN was variable within sites and nutrient impacted sites were not the highest values observed, suggesting that vegetation may be a significant factor in rates of soil organic matter loss, due to species specific nutrient retention and deposition characteristics.

Bulk density was lowest in *Typha* and *Nymphea*, a finding that was expected due t the nature of the dominant plant detritus. Similarly, the highest bulk density was observed in the sites dominated by Salix, a woody species, suggesting possible influence of woody vegetation to soil density. Further, *Panicum* and *Salix* sites are found in ecotypes (wet prairie and swamp) known to drawdown more frequently than others, like sloughs for instance. These more frequent drawdowns may also help to explain the higher bulk densities observed in these site soils.

Fractionation of soil organic matter into labile, moderately labile, and recalcitrant/refractory pools indicted Typha and *Nymphea* sites had the largest pools of labile and moderately labile organic matter. *Cladium* and *Panicum* sites, were indicative of unimpacted BCM soils with high levels of recalcitrant organic material. Therefore, it

is expected that *Typha* and *Nymphea* dominated sites would be the most susceptible to oxidation in the event of drawdown.

The third objective of this research was to determine the effects of various water levels on oxidation of BCM organic soils. Microbial biomass carbon, enzyme activities, CO_2 production, and redox potential were used to evaluate oxidation processes under various water table scenarios in a controlled laboratory setting with near constant temperature and no natural sunlight (dark conditions).

Rates of CO_2 evolution and thus oxidation of soil organic matter increased as water table decreased. This relationship was constant throughout the experimental period, with highest rates being from the cores with lowest water tables and lowest being from the flooded cores. Evaluation of the data suggests that the top 10 to 15 cm is the most reactive area of oxidation as below that the CO_2 flux does not increase as dramatically with each increment of water table drawdown. This is likely due to increased lability of surface soils and the larger amount of active microbial communities in this zone of organic soil. However, physical impediments to oxidation could also be at functioning in these deeper regions to slow oxidation processes by microbial communities.

Redox potential (Eh) indicated that capillary fringe existed above the water table sufficient to maintain anaerobic conditions above the water table in most cores with deep water tables, however, because the experiments were conducted in dark, temperature controlled environments, it is expected that soils exposed to sunlight, variable temperature and humidity, and possibly vegetation, could dry more rapidly than the cores

in the laboratory experiments. This would translate to faster drying under natural; field conditions and likely increased oxidation rates for similar soils in the field.

Calculation of potential subsidence due to oxidation at the various treatment levels of water table suggests that subsidence is minimal at flooded conditions with a rate of 1.2mm per year. The following are the projected rates of secondary subsidence with associated water table depths: 2.4mm yr^{-1} at saturated surface conditions, 2.9mm yr^{-1} at -2cm of water table, 3.6mm yr^{-1} at -5cm waer table, 4.8mm yr^{-1} at -7.6cm water table, 5.4mm yr^{-1} at -15cm, 5.4mm yr^{-1} at -20 cm and 5.7mm yr^{-1} at -30cm.' These estimates are thought to be conservative as actual field conditions may vary significantly. In a natural condition in the field, temperatures well above those maintained in the laboratory during these experiments are possible (22º C). Temperatures exceeding 35º C at the soil surface are not uncommon in the warmer months with high incident sunlight. If a $Q_2$10 factor (biological activity quotient that indicates biological activity doubles every 10 degree increase in temperature) is applied, these rates could double in the warmer seasons in BCMCA.

Phenol content was high along the depth of the flooded and the saturated cores. Drained sections of the soil cores showed a decline in the total phenolics content. However there was no correlation found between the moisture content in soils and their total phenolics content

Microbial enzyme phenol oxidase and β glucosidase activities did not show any increasing or decreasing trend with the water table drawdown. However, flooded and surface saturated cores showed low phenol oxidase activity than the drained cores. Increased β-glucosidase activity along the 5-10 cm sections of the soil cores irrespective

of the water table depth, suggested that it may be strongly correlated to the substrate quality present along the depth of the soil core.

The fourth objective of this research was to determine the effect of frequency of inundation on soil organic matter oxidation by comparing CO_2 flux measurements from three different flooding regimes over the course of many months. Results indicate that flooding and raining cycles protect organic soils from oxidation to a greater extent than does continual drainage, but are significantly less effective than continual flooding. The rate of oxidation from the 10 day and 25 day frequency appear similar, but are more effective at arresting oxidation than the 50 day frequency. The calculated loss of soil to secondary subsidence based upon the treatments is as follows: continually flloded soils to 15 cm are expected to lose 1mm yr^{-1}, 10 day flooding and draining frequency allows f 2.6mm yr^{-1}, 25 day frequency allows 2.7mm yr^{-1}, 50 day frequency allows 3.5mm yr^{-1}, and continually drained soils at -15cm allow 5mm yr^{-1} soil loss to oxidation. As mentioned previously, these rates have the potential to increase with increased soil temperature.

The fifth and final objective of this study was to evaluate the effects of vegetation type (OM source), temperature, and soil nutrient status on aerobic and anaerobic decomposition of soil organic matter. Change in OM decomposition rates in all soils increased with temperature. Temperature coefficient values (Q10) were calculated for each soil type. Different soil types showed different decomposition rates and different amount of total CO_2 evolved. *Nymphea* & *Eleocharis* soils, representative of slough ecotypes, and *Panicum* soils, representative of wet prairie ecotypes, evolved more CO_2

than other vegetation types under aerobic conditions. Chemical characterization studies completed prior to this experiment suggest that elevated nitrogen and labile carbon fractions may be the drivers of these observations. There were two different rates of decomposition observed when soils were incubated under aerobic conditions. Under anaerobic conditions, even though the overall production of C loss is decreased by 40-50% there is continuous loss as anaerobic CO_2 production and CH_4 production. Anaerobic CO_2 evolution was high in *Nymphea* & *Eleocharis* soils as well. Production of methane was high in *Salix* soils and *Nymphea* & *Eleocharis* soils, again suggesting the importance of nutrients and labile organic matter in decomposition under aerobic and anaerobic conditions.

These combined studies indicate that organic soil subsidence in the Blue Cypress Marsh Conservation Area are subject to various levels of impact with respect to water level drawdown. Carbon dioxide flux studies suggest that the surface 10 cm of soil is the most reactive and requires protection from subsidence due to lowered water levels. Soils in the BCMCA are also variable and depending upon dominant vegetation type contributing to soil formation and site nutrient status, their responses to subsidence inducing low water events may likewise be variable. Microbial mediated oxidation is the primary driver of organic soil subsidence, and while shrinkage and compaction due to dewatering can have an effect, the long term losses of organic carbon due to oxidation are the most critical threat. This study provides evidence that suggests any drawdown even resulting in water levels below the soil surface can result in increased soil organic matter

losses to oxidation and that these losses will be variable across the landscape given variation in soil nutrient availability and organic matter quality.

REFERENCES

Andriesse, J.P., 1998. Nature and Management of Tropical Peat Soil. FAO Soils Bulletin 59, FAO, Rome

Armentano, R.V., 1980. Drainage of organic soil as a factor in the world carbon cycle. BioScience 30:825-830

Armentano, T.V. and Verhoeven, J.T.A. 1998. The contribution of freshwater wetlands to the global biogeochemical cycles of carbon, nitrogen and sulfur. Wetlands and Shallow Water Bodies 1 SPB Academic Publishing, The Hague.

Baldwin A.H., M.S. Egnotovich and E. Clarke 2001. Hydrologic change and vegetation of tidal freshwater marshes: field, greenhouse, and seed-bank experiments. Wetlands, 21: 519–531

Baldwin, D.S. 1996. Effects of exposure to air and subsequent drying on the phosphate sorption characteristics of sediments from a eutrophic reservoir. Limnol. Oceanogr. 41:1725-1732.

Ball, D.M., M. Collins, G. Lacefield, N. Martin, D. Mertens, K. Olsen, D. Putnam, D. Undersander, and M. Wolf. 2001. Understanding Forage Quality. American Farm Bureua Federal Publication 1-01, Park Ridge, IL.

Berglund, K. 1995. Optimal drainage depth of five cultivated organic soils. Swedish J. Agric. Res. 25:185-196

Braekke, F.H. 1987. Nutrient relationships in forest stands: effects of drainage and fertilization on surface peat layers. Forest Ecol and Mgt 21:269-284

Bolger, R. T., R. Cully and D. Little. 2003. Recalcitrant soil organic materials mineralize more efficiently at higher temperaures. J. Plant Nutr. Soil. Sci. 166, 300-307

Casanova, M. T. and M. A. Brock. 2000. How do depth, duration and frequency of flooding influence the establishment of wetland plant communities? Plant Ecology 147:237–250.

Clymo, R.S., 1987. The ecology of peatlands. Sci. Prog. Oxford, 71:593-614.

D'Angelo, E.M., and K.R. Reddy 1994. Diagenesis of organic matter in a wetland receiving hypereutrophic lake water. I. Distribution of dissolved nutrients in the soil and water column. J. Environ. Qual. 23:937-943.

Dalias P. Anderson J.M. Bottner, P. Couteaux M-M 2001a. Temperature responses of carbon mineralization in conifer forest soils from different regional climates incubated under standard laboratory conditions. Global change Biolo 7:181-192

Dalias P. Anderson J.M. Bottner, P. Couteaux M-M 2001b. Long term effects of temperature on carbon mineralization processes Soil biolo. Biochem 33:1049-1057.

Davis, J. H., Jr. 1940. The ecology and geologic role of mangroves in Florida. Papers from the Tortugas Laboratory 32. Carnegie Institution of Washington Publication 517:307–412.

Davis, S. 1991. Growth, decomposition, and nutrient retention of *Cladium jamaicense* Crantz and *Typha domingensis* Pers. In the Florida Everglades. Aquatic Botany 40: 203-224

De Groot, J.C., and C. Van Wijck 1993. The impact of desiccation of a freshwater marsh (garines Nord, camargue, France) on sediment-watervegetation interactions, Part 1 Sediment chemistry. Hydrobiol. 252:83-94.

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

DeBusk, W., S. Newman, and K.R. Reddy. 2001. Spatio-temporal patterns of soil phosphorous enrichment in everglades water conservation area 2A. J. Environ. Qual. 30(4): 1438-1446

DeBusk, W. and K.R. Reddy. 1998. Turnover of detrital organic carbon in a nutrient impacted Everglades marsh. Soil Sci. Soc. Amer. J. 62(5): 1460-1468

Doels, D. 1995. The Peat OXidation and Permanent Shrinkage (POXAPS) model. DLO Winand Staring Centre for Integrated Land , Soil and Water Research, Wageningen.

Dolman, J.D. and S.W. Buols. 1967. A study of organic soils in the tidewater region Of North Carolina Agricultural Experiment Station, Raleigh, NC, USA. Technical Bulletin No. 181

Eivazi, F. and M. A. Tabatabai, 1988. Glucosidases and galactosidases in soils. Soil Biol. Biochem. 20: 601-606.

Farnham, R.S and H.R. Finney. 1965. Classification and properties of organic soils. Adv. Agron. 17:115-162

Fenchel, T.M., and B.B Jorgenson. 1977. Detritus food chains of aquatic ecosystems: The role of bacteria. Adv. Microbial Ecol. 1:1-58.

Fisher, M.M., and K.R. Reddy . 2001. Phosphorus flux from wetland soils affected by long-term nutrient loading. J. Environ. Qual. 30:261-271.

Freeman, C., G. Liska, N.J. Ostle, M.A. Lock, B. Reynolds, and J. Hudson 1996. Microbial activity and enzymatic decomposition processes following peatland water table drawdown. Pl. Soil. 180:121-127.

Freeman, C., R. Gresswell, H. Guasch, J. Hudson, S. Hughes, and B. Reynolds. 1995. Climate change: Man's indirect impact on wetland microbial activity and the biofilms of a wetland system. In Man's influence on Freshwater Ecosystems and Water use. International Association of Hydrological Sciences Publication No. 230. IAHS Press, Wallingford, UK.

Giardina, C.P. and M.G. Ryan, 2000, Evidence that decomposition rates of organic carbon in mineral soil do not vary with temperature. Nature, 404, 858-861

Glaz, B. 1995. Research seeking agricultural and ecological benefits in the Everglades. Journal of Soil and Water Conservation 50: 609-613

Godshalk, G.L., and R.G. Wetzel . 1978. Decomposition of aquatic angiosperms: I. Dissolved compounds. Aquat. Bot. 5:281-300.

Gorham, E. 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecological applications. 1:182-195.

Haakh, F. 2002. Water resources management within the waterprotection zone of the "Donauried": solving conflicts between water supply, ecology and agriculture. Water Sci and Tech., 46:63-70

Holdren Jr., G.C., and D.E. Armstrong. 1980. Factors affecting phosphorus release from intact lake sediment cores. Environ. Sci. Technol. 14:79-87.

Howard D.M. Howard P.J.A . 1993. Relationships between Co2 evolution, moisture content and temperature for a range of soil types . Soil Biolo. And Biochem 25:1537-1546

Ilniki, P. 1977. Sackung in wiederholt entwässertne Hochmooren des nordwestdeutschen Flachlandes. 3. Berich: Vorausberechnung der Moorsackung. Z. Kulturtech. Flurber. 18:153-165

Johnson, S. A., W. J. Mitsch, and N. L. Christensen. 2000. Unpublished data. p. 394–395. *In* W. J. Mitsch and J. G. Gosselink, Wetlands, Third Edition. John Wiley and Sons, New York, NY, USA.

Jongedyk, H.A., R.B. Hickok, I.D. Mager and N.K. Ellis 1950. Subsidence of muck soil in Northern Indiana. Bull. S.C. no. 366 Purdue Univ. Agric. Exp. Stn., Lafayette, Ind.

Kirchbaum, M.U.F. 1995. The temperature dependence of soil organic matter decomposition and effect of global warming on soil carbon storage. Soil Biology and Biochemistry, 27. 753-760

Kirchbaum, M.U.F. 2000. Will changes in soil organic matter act as positive or negative feedback on global warming? Biogeochemistry. 48. 21-51.

Lloyd, J. and J.A. Taylor, 1994, On the temperature dependence of soil respiration. Functional ecology, 8. 315-323.

Liski, J. A. Nissinen, and M. Erhard. 2003. Climatic changes on litter decomposition from arctic tundra to tropical rainforest. Global change Biology. 9. 575-584

Lorenzen, B., H. Brix, I.A. Mendelssohn, K.L. McKee, and S.L. Miao 2001. Growth, biomass allocation and nutrient use efficiency in Cladium jamaicense and Typha domingensis as affected by phosphorus and oxygen availability. Aquatic Botany 70:117-133.

Martin, H.W., D.B. Ivanoff, D.A. Graetz and K.R. Reddy 1997. Water table effects on Histosol drainage water carbon, nitrogen and phophorus. J. of Environ. Qual. 26:1062-1071

Mathur, S.P. and M.P. Lévesque 1977. Letter to the editor: subsidence of an organic soil in southwestern Quebec. Can J. Soil Sci. 57:503-504

McAfee, M. 1989. Aeration studies on peat soil. 1. The effects of watertable depth and crop cover. Int. Peat. J. 3:157-174

McLatchey, G.P., and K.R. Reddy 1998. Regulation of organic matter decomposition and nutrient release in a wetland soil. J. Environ. Qual. 27:1268-1274.

Mitsch, W. J. and J. G. Gosselink 2000. Wetlands, third edition. John Wiley and Sons, New York, NY, USA.

Moore Jr., P.A., K.R. Reddy, and M.M. Fisher 1998. Phosphorus flux between sediment and overlying water in Lake Okeechobee, Florida: spatial and temporal variations. J. Environ. Qual. 27:1428-1439

Morris, D.R. B. Glaz and S.H. Daroub 2004. Organic soil oxidation potential due to periodic flood and drainage depth under sugarcane. Soil Sci. 169:600-608

Nesterenko, I.M. 1979. Subsidence and wearing out of peat soils as a result of reclamation and agricultural utilization of marsh lands. Proc. V Int. Peat Congr., Poznan, 1:218-232

Newman, S., and K.R. Reddy 1993. Alkaline phosphatase activity in the sediment-water column of a hypereutrophic lake. J. Environ. Qual. 22:832-838.

Newman, S., J. Grace, and J. Koebel. 1996. Effects of nutrients and hydroperiod on *Typha*, *Cladium*, and *Eleocharis*: Implications for Everglades restoration. Ecol. Appl. 6(3): 774-783

Newman, S., J. Schuette, J. Grace, K. Rutchey, T. Fontaine, K. Reddy, and M. Pietrucha. 1998. Factors influencing cattail abundance in the northern Everglades. Aquat. Bot. 60(3): 265-280

Odum, H.T. 1971. Energy, power and society. McGraw Hill, New York, N.Y. 1000 pp.

Okruszko, H. 1989. Critical and optimal groundwater tables in peat soils determined by hydrophysical properties. Agrokemia es Talajtan 38:596-600

Olila, O.G., K.R. Reddy, and D.L. Stites. 1997. Influence of draining on phosphorus forms and distribution in a constructed wetland. Ecol. Eng. 9:157-159.

Osborne, T.Z. 2005. Characterization, mobility, and fate of dissolved organic matter in a wetland ecosystem. Doctoral Dissertation, University of Florida.

Pant, H.K., and K.R. Reddy 2001. Hydrologic influence on stability of organic phosphorus in wetland detritus. J. Environ. Qual. 30:668-674.

Raich J.W. Potter C.S 1995. Global patterns of CO2 emissions of carbon dioxide from soils. Global biochemical Cycles, 9, 23-36

Reddy, K. R., and P. S. C. Rao 1983. Nitrogen and phosphorus fluxes from a flooded organic soil. Soil Sci. 136:300-307.

Roberts, J., and A. Rowland. 1998. Cellulose fractionation in decomposition studies using detergent fiber pretreatment methods. Commun. Soil Sci. Plant Anal. 25: 269-277

Rojstaczer S., and Deverel, S.J. 1995. Land subsidence in drained histosols and highly organic mineral soils of California. Soil Sci. Soc. Am. J. 59:1162-1167.

Rojstaczer, S., and S.J. Devel. 1993. Time dependence in atmospheric carbon inputs from drainage of organic soils. Geophys. Res. Lett. 20:1383-1386

Salmah, Z., G. Spoor, A.B. Zahari and D.N. Welch. 1994. Importance of water management in peat soil at farm level. In: B.Y. Aminuddin (Editor), Tropical Peat; Proc. Int. Symp. on Tropical Peatland, 6-10 May 1991, Kuching, Sawarak, Malaysia.

Schipper, L.A. and M. McLeod. 2002. Subsidence rates and carbon loss in peat soils following conversion to pasture in the Waikato Region, New Zealand. Soil Use and Mgt 18:91-93

Schothorst, C.J., 1977. Subsidence of low moor peat soils in the Western Netherlands. Geoderma, 17:265-291

Shih, S., B. Glaz and R.E. Barnes, Jr., 1997. Subsidence lines revisited in the Everglades Agricultural Area, 1997. Tech. Bulletin 902, Ag. Exp. Station. University of Florida.

Shih, S.F., B.Glaz and R.E. Barnes, Jr. 1998. Subsidence of organic soils in the Everglades Agricultural Area during the past 19 years. Soil Crop Sci. Soc. Fla. Proc. 57:20-29

Shih, S.F., E.H. Stewart, L.H., Allen, Jr., and J.W. Hilliard. 1978. Variability of depth to bedrock in Everglades organic soil. Soil and Crop Sci. Soc. Fla. Proc. 38:66-71

Shih, S.F., J.W. Mishoe, J.W. Jones, and D.L. Myhre. 1979. Subsidence related to land use in the Everglades agricultural area. Trans. ASEA 22:560-563.

Silins, U. and R.L. Rothwell. 1998. Forest peatland drainage and subsidence affect soil water retention and transport properties in an Alberta peatland. Soil Sci. Soc. Am. J. 62:1048-1056

Simpson, R. L., R. E. Good, M. A. Leck, and D. F. Whigham. 1983. The ecology of freshwater tidal wetlands. BioScience 33:255–259.

Stephens, J.C. 1956. Subsidence of organic soils in the Florida Everglades. Soil Science Society of America Proceedings 20: 77-80

Stephens, J.C., C.H. Allen, Jr. and E. Chen. 1984. Organic soil subsidence. Rev. Eng. Geol. 16:107-122

Stewart, J.M. 1994. Subsidence in cultivated peatlands. In: B.Y. Aminuddin (Editor), Tropical Peat; Proc. Int. Symp. on Tropical Peatland, 6-10 May 1991, Kuching, Malaysia.

Tate, R.L. 1979. Effect of flooding on microbial activities in organic soils: carbon metabolism. Soil Sci. 128:267-273

Tate, R.L. 1980a. Variation in microbial activity in Histosols and its relationship to soil moisture. Appl. Environ. Microbiol. 40:313-317

Tate, R.L. 1980b. Effect of several environmental parameters on carbon metabolism in Histosols. Microb. Ecol. 5:329-336

Templet, P.H., and K.J. Meyer-Arendt 1988. Loiusiana Wetland Loss: A regional Water Management Approach to the Problem. Environ. Mgt. 12:181-192

Thornley J.H.M Cannell M.G.R 2001. Soil carbon storage response to temperature: an hypothesis. Ann Bot. 87:591-598

Van Soest, J.P. 1970. Forage and Fiber Analysis Handbook No. 379. USDA. Washington, D.C.

Whigham, D. F., R. L. Simpson, and M. A. Leck 1979. The distribution of seeds, seedlings, and established plants of arrow arum (*Peltandra virginica* (L.) Kunth) in a freshwater tidal wetland. Bulletin of the Torrey Botanical Club 106:193–199.

White, J.R., and K.R. Reddy 2001. Influence of selected inorganic electron acceptors on organic nitrogen mineralization in Everglades soils. Soil Sci. Soc. Am. J. 65: 941-948.

Williams, C.J., Shingara, E.A., and Yavitt, J.B. 2000. Phenol Oxidase activity in peatlands in New York state: response to summer drought and peat type. Wetlands 20:416-421

Wösten, J.H.M., A. B. Ismail and A.L.M. van Wijk 1997. Peat subsidence and its practical implications: a case study in Malaysia. Geoderma 78, 25-36

Wright, A.L. and K.R. Reddy 2001. Phosphorus loading effects on extracellular enzyme activity in Everglades wetland soils. Soil Sci. Soc. Am. J.: 65:588-595.

Zogg G. P. Zak, DR., Ringelberg D.B. MacDonaldN.W, Pregitzer K.S. White, D.C. 1997. Compositional and functional shifts in microbial communities due to soil warming. Soil Sci. Soc. Am. J. 61:475-481