Soil carbon dynamics and aquatic metabolism of a wet-dry tropics wetland system

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Declaration

I hereby declare that this 9-month Masters by Research thesis has not been previously submitted to any other institution or university for a higher degree. This thesis is comprised entirely of my own work, except where otherwise acknowledged. There were no ethics considerations which required approval for this thesis.

Danelle Agnew
9 October 2018

Cover photo: Kings Lake permanent wetland looking south. D Agnew

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Abstract

Freshwater wetlands are a key component of global carbon dynamics. Globally, wet-dry tropics wetlands function as wet season carbon sinks or dry season carbon sources with low aquatic metabolism controlled by predictably seasonal, yet magnitude-variable flow regimes and inundation patterns. However, these dynamics have not been adequately quantified in Australia's relatively unmodified wet-dry tropics freshwater wetlands. A baseline understanding is required before analysis of land-use or climate change impacts on these systems can occur.

This study characterises geomorphology and sedimentology across a seasonally connected ephemeral and permanent wet-dry tropics freshwater wetland system at Kings Plains, Queensland, and quantifies soil carbon stocks and wet- and dry-season aquatic metabolism. Sediment from 33 cores was assessed for carbon content using loss-on-ignition. Soil carbon stocks to 1 m depth are 51.5 ± 7.8 kg C m^{-2}, with potential for long-term retention at greater depths. This is higher than other wet-dry tropics wetlands found globally. Aquatic metabolism was measured using biological oxygen demand method on sediment collected during both seasons and inundated under laboratory conditions. Results show overall low productivity, with respiration dominant in both seasons. Quantification of sediment and aquatic metabolism carbon in this nutrient-limited wetland system serves as a baseline for future research and environmental management.

1 Introduction, literature review and thesis aims

1.1 Wetlands in the wet-dry tropics

Wetlands are areas of either permanent or recurrent saturated land, ranging from freshwater to saline, where the presence of water influences their biogeochemical composition, and creates unique ecosystems (Bernal and Mitsch, 2013). Wetlands perform key ecological functions supporting biodiversity, mitigating flood and coastal erosion impacts, enhancing water quality, and storing carbon (Finlayson et al., 2013, Junk et al., 2013, Villa and Bernal, 2018). Globally, wetlands are geographically and functionally diverse and are found from high to low latitudes, cold to hot climates, and in upland and lowland settings. An individual wetland's characteristics, including inundation patterns, vary according to their underlying lithology, geomorphology, climate, vegetation structure and hydrological regimes (Kayranli et al., 2010). Many wetlands, particularly in tropical zones, are highly modified and at ongoing risk from drainage, development and population pressures (IPCC, 2014). These pressures lead to a loss in ecosystem services and environmental degradation, including sediment loss, changes to hydrology, soil oxidation, and carbon loss (Junk et al., 2013).

Tropical environments are characterised by temperatures >18 °C, and constant or seasonal periods of significant rainfall forming several climate sub-zones (Peel et al., 2007). Within the Köppen-Geiger climate classification, Monsoon and Humid tropics climate zones experience year round elevated rainfall (Peel et al., 2007). The Savannah, or wet-dry, tropics climate zone is globally the most spatially extensive (Peel et al., 2007). It is characterised by large differences in wet season rainfall duration and intensity, and minimal dry season rainfall (Mitsch et al., 2010, Peel et al., 2007). The dry season is mostly cloud free (Ward et al., 2013). The dominant tropical climate zone in Australia is the wet-dry tropics, ranging across the northern one-third of the continent (Cook et al., 2010).

The hydrological flux of Australia's wet-dry tropics wetlands is characterised by a short intense wet season and a long dry season with minimal rainfall (Ward et al., 2013). Year-to-year magnitude differences, with predictably seasonal, yet magnitude-variable, flow regimes occur (Page and Dalal, 2011). These wetlands span a continuum from permanent inundation to seasonally ephemeral. Wet-dry tropics wetlands become increasingly disconnected as the dry season progresses (Pettit et al., 2017a). Northern Australia palustrine wetland waterbody area shrinks by an estimated >90 % from March (wet season extent peak) to end October (Ward et al., 2013). The river systems of Australia's wet-dry tropics, and their associated wetland complexes, are relatively undisturbed and unmodified, due to low population density and minimal economic development (Warfe et al., 2011).

1.2 Geomorphological template of wetlands

Landscape evolution is a consequence of an environment's geological setting, and its shaping by fluvial processes and climate (Brierley and Fryirs, 2005, Fryirs and Brierley, 2013). In low relief landscape settings, variation in valley width can develop from alternating sequences of local resistant bedrock and carved out alluvium filled valleys. This can control the balance between fine and coarse sediment deposition in floodplain wetland systems (Tooth et al., 2002). In these settings, wide valleys can be characterised by low slope, channel meanders and fine-grained vertically accreted sediment deposition. Confined stretches typically have higher slope, incised channels, and deposition of comparatively coarser sediments.

Geomorphology has a dynamic influence on wetlands and their ecosystems, with modification by erosion, sedimentation and hydrology, impacting on functions such as carbon cycling and water filtering (Renschler et al., 2007). In wet-dry tropics wetlands, wet season flood pulsing is the driver for hillslope erosion and sediment discharge downstream, as evidenced by changes in the sedimentology of alluvial basins over long time scales (Best and Dallwitz, 1963).

Australian wet-dry tropics soils are predominantly infertile sandy to sandy clay loams (Isbell, 2016, Woinarski et al., 2005). However, within this is a mosaic of sediment environments dominated by dark-coloured 'cracking clays', or vertosols, occurring in alluvium filled valleys (Hutley et al., 2011). These cracking clay soils occur globally on large land masses in wet-dry tropics climate zones, such as Australia, USA and Africa. Their profile is characterised by shrinking in the dry season, and swelling and becoming plastic in the wet season (Cook et al., 2010). Vertosols facilitate carbon cycling down through the soil profile during the dry season as surface organic matter falls into the cracks (DES, 2013). They are dominant in the smectite group clay minerals, especially montmorillonite, with resultant high water holding capacity (Ahmad, 1983). The strong physicochemical bonds of clay enable strong adsorption of organics to clay particles compared to silty or sandy particles. The strength of these physicochemical bonds results in slow decomposition rates of organic matter in clay soils compared to sandy soils. This slow decomposition contributes to carbon stability within the soil in tropical regions (Hassink, 1997). Within soils, indicators of wetland environments include mottling, segregations, such as nodules, and decreasing soil matrix chroma (Bryant et al., 2008). Within a wetland system, variation in hydrogeomorphic characteristics and physical landscape setting produces an assemblage of different sediment types, with different water and carbon storage and release capacities.

1.3 Carbon in wetlands

Carbon storage or carbon sequestration (CS) is defined as carbon uptake into terrestrial soils, biomass or marine and freshwater environments. CS plays a key role in mitigating global temperature increases, believed to be caused by heat trapped in the atmosphere by anthropogenic greenhouse gas (GHG) production, mainly carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). This ultimately leads to a changing global climate (IPCC, 2014). Terrestrial environments, which include freshwater rivers and wetlands, play a key role in global carbon dynamics (Sutfin et al., 2016), through the storage of carbon in biomass, soils and sediments, carbon transformation through aquatic metabolism (Kayranli et al., 2010, Mitsch et al., 2013) and gaseous end products (Mitsch and Gosselink, 2000), often with long residence times (Sutfin et al., 2016). Wetlands, one of the biosphere's most productive ecosystems, are important carbon sinks, due to high aquatic metabolism and slow decomposition (Morris et al., 2012).

Coastal and marine wetlands generally have a higher carbon density store than freshwater wetlands, due to inhibited methane production (Howe et al., 2009). Globally, wetland research has focused on the 'blue carbon' of tidal wetlands, and boreal and temperate freshwater wetlands, rather than tropical freshwater wetlands (Howe et al., 2009, Pearse et al., 2018). However, globally, the more extensive areal extent of freshwater wetlands results in substantially higher total stores of carbon, and warrants increased focus on tropical freshwater wetland research (Nahlik and Fennessy, 2016, Pearse et al., 2018, Pettit et al., 2017a). Within tropical freshwater wetlands, carbon research has focused on Rainforest and Monsoon tropics climate zones (Peel et al., 2007), and less on Savannah, or wet-dry, tropics climate zones. This is also inadequately quantified in the Australian context. The relatively undisturbed and unmodified wet-dry tropics river and freshwater wetland systems of northern Australia provide an ideal opportunity to establish a baseline understanding of soil carbon and aquatic ecosystem dynamics in this zone (Warfe et al., 2011).

Within Australia, research on the carbon dynamics of ephemeral freshwater wetlands has focused on semi-arid wetlands (Kobayashi et al., 2013). Whilst research on aquatic metabolism in the wet-dry tropics has been undertaken (Jardine et al., 2013, Pettit et al., 2017b, Ward et al., 2013), less is known of carbon stored in sediments in wet-dry tropics ephemeral wetlands (Finlayson et al., 2013, Kelly, 1997). As Australian wet-dry tropics extends across the northern third of Australia, more detailed knowledge of the carbon dynamics in the wetlands of this region is vital for conservation management and climate change modelling.

Soil carbon (SC) is made up of organic matter (soil organic carbon; SOC) and inorganic matter (soil inorganic carbon; SIC) (Scharlemann et al., 2014). SOC is derived from allochthonous inputs, such as riparian vegetation (Sutfin et al., 2016), and SIC from carbonates and bicarbonates. SC has varying turnover periods in soils, ranging from hours to days for dissolved organic carbon (DOC), to >1000 years for SIC. SOC is slowly stored in wetlands, often in saturated anaerobic conditions with low decomposition rates, resulting in lower CO_2 production and higher SOC levels. This is compared to faster aerobic decomposition in terrestrial soils with lower SOC levels (Page and Dalal, 2011). The level of carbon storage is in part determined by a wetland's decomposition characteristics, that is, the balance between anaerobic and aerobic conditions (Sutfin et al., 2016). Soil organic matter (SOM) is all organic matter found in soils, that is all plant and animal residues at varying decomposition stages, and the metabolic products of soil organisms. SOC is the <2 mm diameter component of SOM, whereas >2 mm diameter component is detritus and plant matter (Rayment and Lyons, 2011). Soils rich in organic matter, such as peat soils, have discernable plant matter which must be excluded from SOC.

The relationship between the concentration of SOC in a given mass, and total stocks of SOC by weight for a given area to a predetermined depth, is dependent on a soil's bulk density. Dry bulk density (DBD) is the dry weight of soil in a given volume, thereby taking into account soil pores containing air and water (Brown and Wherrett, 2018). The high SOC stocks of tropical peat soils are due to high SOC concentrations, which are rich in organic matter yet moderated by their low bulk densities (Köchy et al., 2015). In contrast, soils with high bulk densities require lower levels of SOC concentration to achieve similar levels of SOC stock. Baseline SOC concentration and stock, and subsequent CS measurement over time, will aid in understanding the source and stability of carbon in wetland environments. CS can be measured through soil accretion rates, or comparison of old and new cores (Adame and Fry, 2016, Bernal and Mitsch, 2013). Whilst this study is not addressing CS, measuring SOC concentration and stock serves as a baseline for future CS analysis.

In the water column, carbon is derived from autochthonous inputs from aquatic primary production, and allochthonous inputs from the littoral zone and aquatic macrophytes (Sutfin et al., 2016). Inorganic carbon is converted to organic carbon through photosynthesis (Kayranli et al., 2010) with the balance between gross primary productivity of phytoplankton (GPP) and planktonic respiration (PR) determining carbon turnover in wetlands.

Whilst wetland ecosystems are generally considered important carbon sinks (Howe et al., 2009, Sutfin et al., 2016), wetland diversity results in a broad range of carbon storage levels and sequestering capacity (Bernal and Mitsch, 2012). Carbon stocks and CS rates vary according to wetland type, for example, carbon stocks of boreal wetlands are highest due to their areal extent and depth, however have the slowest CS rates of all wetland types (Mitsch et al., 2013). Wetlands can also function as carbon sources, releasing carbon to the atmosphere as GHGs, CO_2 through decomposition and respiration, and as CH_4 through methanogenesis (Leifeld and Menichetti, 2018). Wetland biogeochemistry, environmental condition and the biological response are inextricably linked. Nutrient availability and the metabolic pathway, aerobic or anaerobic, determine wetland carbon accumulation and metabolism (Kayranli et al., 2010). These ecosystem processes are influenced by biomass structure, soil type, geomorphology, climate, and topography (Allen et al., 2010, Kayranli et al., 2010), especially soil temperature and moisture (Ontl and Schulte, 2012). Atmospheric carbon emissions from anthropogenically modified terrestrial soils are estimated as the second largest after fossil fuel combustion, however, quantity and spatial distribution estimates vary significantly (Scharlemann et al., 2014). The wetland carbon cycle is shown in Figure 1.1.

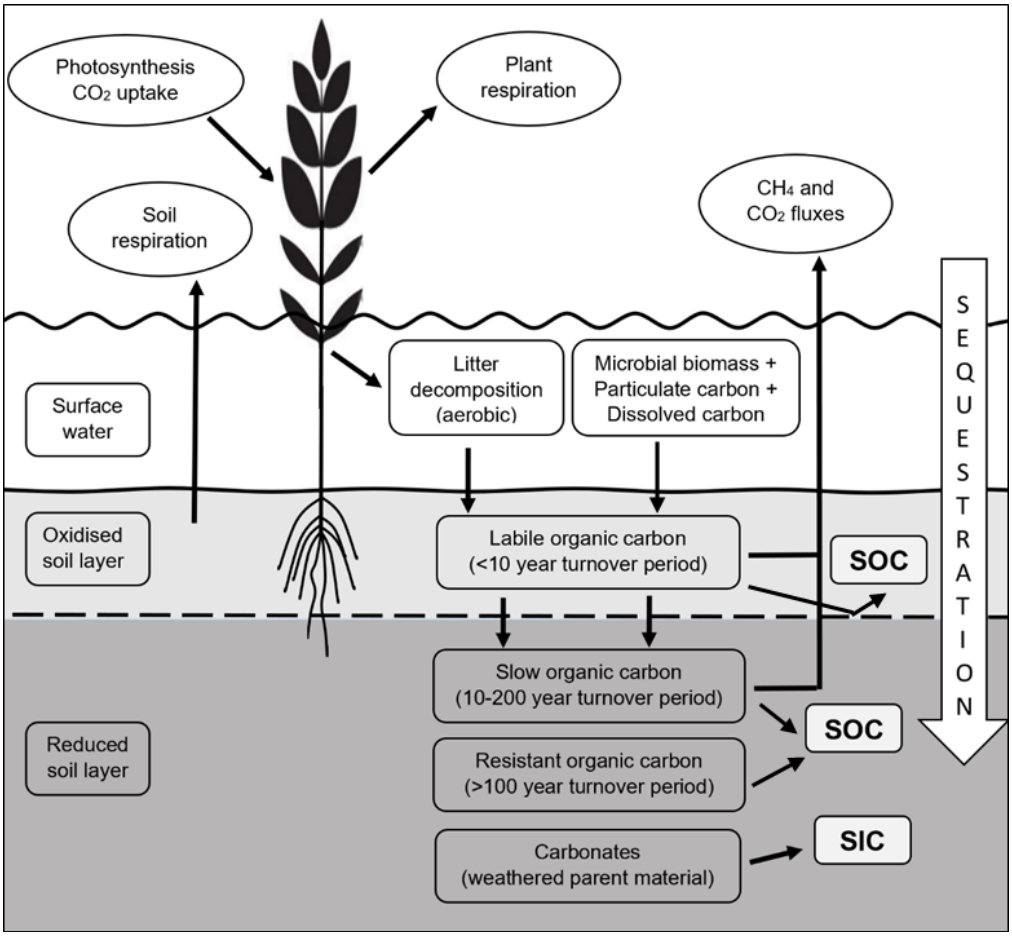


Figure 1.1 Conceptual diagram of wetland carbon cycle, including various carbon pools. Adapted from Kayranli et al. (2010) .

1.3.1 Soil carbon in wetlands

Global SOC stores in terrestrial environments are significant, greater than carbon stored in plant biomass and the atmosphere (Scharlemann et al., 2014), and exceeded only by marine storage. However, considerable uncertainty surrounds global estimates of wetland carbon storage. For example, differing definitions of 'wetland' and assumptions of bulk density lead to broad ranges of total carbon pools in global estimations (Köchy et al., 2015). Wetlands only account for 5-8 % of global terrestrial extent, yet are believed to store proportionally higher (20-30 %) SOC reservoirs compared to terrestrial soils and have increased residence times (Kayranli et al., 2010, Mitsch et al., 2013, Zedler and Kercher, 2005). CS rates in temperate and tropical wetlands are estimated to be 4-5 times higher than in boreal wetlands (Bernal and Mitsch, 2013). Nevertheless, tropical wetlands can be either carbon sinks or sources dependent largely on anthropogenic drainage or seasonal hydrological regimes (Bernal and Mitsch, 2013). Within the wetland biome, global SC pools of tropical forests, northern peatlands and coastal and marine wetlands have been estimated. However, less is known of global SC pools of temperate and tropical freshwater wetlands. Combined temperate and tropical freshwater wetlands are estimated to form a global SC pool of ~750 Mg C ha^{-1} (Villa and Bernal, 2018). To enable comparison to other wetland classes, separate accurate global SC pool estimates of temperate and tropical wetland classes is required.

Globally, tropical terrestrial soils, including wetlands, have moderate SOC density. This is due to the balance between elevated primary productivity and decomposition rates from the regions' high temperatures and rainfall (Ontl and Schulte, 2012). Deep tropical peatlands, both soils and wetlands, are critical environments as their SOC reservoirs are highest (Junk et al., 2013, Page et al., 2011). SOC is usually measured to only 100 cm depth as it tends to rapidly decline at greater depths. However, deeper tropical peat soils have higher SOC stores at depth. Estimates vary but could potentially be 1.5-2 times greater to 3 m depth (Page et al., 2011, Scharlemann et al., 2014). Carbon stores in deeper soils (below 1 m) can be stable for centuries if left undisturbed, contingent on local environmental and biological controls (Schmidt et al., 2011). However, the residence time of this carbon in deeper soils is vulnerable to environmental and anthropogenic disturbance (Bernal et al., 2016). Economic exploitation leads to wetland drainage, soil oxidation, and SOC loss; a critical issue in developing countries with large tropical wetland areas (Junk, 2013).

Estimates of SOC concentrations, DBD and SOC stocks in global wetland classes are highly variable (Köchy et al., 2015, Villa and Bernal, 2018) as shown in Table 1.1.

Table 1.1 Soil organic carbon (SOC) concentration and stock, and dry bulk density (DBD for a) global wetland classes and b) selected freshwater wetlands. Wet-dry tropics wetlands highlighted.

Climate zone and wetland type	**Wetland system**	**SOC (%)**	**SOC concentration ($g\ C\ kg^{-1}$)**	**DBD ($g\ cm^{-3}$)**	**SOC stock ($kg\ C\ m^{-2}$)**	**Depth (m)**	**Reference**
a) Global wetland classes							
Wet-dry tropics, intermittent	Global average e.g. northern Australia, southern Africa	..	..	..	4.1	1.0	Köchy et al. (2015)
Wet-dry tropics, permanently inundated	Global average e.g. northern Australia, southern Africa	..	..	..	9.2	1.0	Köchy et al. (2015)
Tropical, swamp forest	Global average e.g. Brazil	..	..	..	11.0	1.0	Köchy et al. (2015)
Tropical, floodplain	Global average e.g. central Africa, Brazil	..	..	..	9.5	1.0	Köchy et al. (2015)
All temperate and tropical	Global average	..	..	..	75	1.0	Villa and Bernal (2018)
b) Selected freshwater wetlands							
Wet-dry tropics, seasonal	Palo Verde NP, Costa Rica	..	25.4-65.1	0.66	Avg 8.2 (1.36 per 5 cm)	0.3	Bernal and Mitsch (2013)
Wet-dry tropics, seasonal	Okavango Delta, Botswana	..	4.1-152.7	0.96-1.02	4.8 – 9.0 (0.8-1.5 per 5 cm)	0.3	Bernal and Mitsch (2013)
Wet-dry tropics, swamp	Veracruz, Mexico	20.0 ± 2.91*	50-225 87.59 ± 6.88*	0.31 ± 0.08*	34.96 ± 1.3*	0.8	Marín-Muñiz et al. (2014)
Wet-dry tropics, marsh	Veracruz, Mexico	13.73 ± 3.38*	50-150 57.59 ± 3.23*	0.28 ± 0.05*	25.85 ± 1.19*	0.8	Marín-Muñiz et al. (2014)
Sub-tropical, forested floodplain	Georgia, USA	5.2 ± 1.3*	..	1.02 ± 0.11*	5.3	0.30	Craft and Casey (2000)
Temperate, seasonally inundated	Cobboboonee NP, Victoria, Australia	29 -62	196.3 ± 13.9* (top 75 cm)	..	68.3 ±18.0* (high watermark)	1.0	Pearse et al. (2018)
Temperate, upland peat swamp	Blue Mountains, NSW, Australia	..	31-170	0.49-1.74	55.4-70	2.0	Cowley et al. (2018) , Cowley et al. (2016)
Temperate, wetland #	Ohio, USA	..	146 ± 4.2*	0.55 ± 0.01*	4.2	0.35	Bernal and Mitsch (2012)
Temperate, wetland ##	Ohio, USA	1.0-10.4	50.1 ± 6.9*	0.74 ± 0.06*	1.5 ± 7.8*	0.35	Bernal and Mitsch (2012)

Where available range plus mean ± std error (*) are provided. NP National Park, # depressional, ## flow-through.

1.3.1.2 Soil carbon in wet-dry tropics wetlands

The faster aerobic decomposition in wet-dry tropics wetlands results in lower SOC levels and CS rates than in humid tropical wetlands, causing these wetlands to function variously as carbon sources or sinks (Adame and Fry, 2016, Bernal and Mitsch, 2013, Page and Dalal, 2011). Australian wet-dry tropics wetlands can be dry season carbon sources or wet season sinks under predictable seasonal flows (Page and Dalal, 2011). The extent of ephemerality is believed to determine their overall carbon dynamics. Despite the lower SOC storage rates of Australian wet-dry tropics wetlands, their overall contribution to SOC stores is high due to their extensive land area (Cook et al., 2010). Understanding carbon dynamics in wet-dry tropics wetlands is important in efforts to mitigate climate change and manage economic and population pressures. Potentially significant carbon releases may occur through disturbance, temperature-induced evapotranspiration increases, or wetland desiccation under changed hydrology patterns (Bernal and Mitsch, 2013). For example, drainage of Australian wet-dry tropics wetlands would convert them from carbon sink to source, with SOC losses of ~25 % (Page and Dalal, 2011). The impacts of land clearing, overgrazing, conversion to cropping, and burning, on soil structure breakdown, erosion and altered vegetation community, all contribute to SOC loss (Cook et al., 2010, Page and Dalal, 2011). Pressure for agricultural development in northern Australia, for example in the Flinders and Gilbert catchments, would potentially release carbon through irrigation-induced changes to hydrological regimes (Finlayson et al., 2013). Both increased fire frequency and intensity, and total fire elimination, are thought to reduce SOC storage through reduced charcoal and lower grass carbon contributions to soil (Scharlemann et al., 2014). Comparison of isotopic analysis of century-old sediment cores with recent cores from tropical wetlands demonstrated long-term temporal changes in soil carbon, due to soil decomposition and changes in vegetation structure and carbon sources (Adame and Fry, 2016).

1.3.2 Aquatic metabolism in wetlands

Wetland inundation patterns vary according to their climate, hydrological regimes and geomorphology, which impacts their aquatic metabolism (Bortolotti et al., 2016, Hoellein et al., 2013). Semi-arid inland wetlands have highly intermittent inundation regimes and explosions of productivity (Kobayashi et al., 2013). In contrast, the Australian wet-dry tropics wetland inundation pattern drives a highly seasonal aquatic metabolism response (Pettit et al., 2017b, Ward et al., 2013). This response is a significant component of wetland carbon cycling and transformation, and aids in understanding whether a wetland system is autotrophic or heterotrophic (Bortolotti et al., 2016). Importantly, GPP and PR are fueled by carbon, nitrogen, phosphorous and other nutrients, so the balance of GPP/PR can have some impact on SOC over time(Kobayashi et al., 2013). Australian wet-dry tropics food webs are predominantly algal based (Pettit et al., 2017a). Autotrophic phytoplankton can store energy through photosynthesis alone, whereas heterotrophic zooplankton consume organic matter produced by other organisms (Kirschbaum et al., 2001). Wetland aquatic metabolism is complex. It is determined by wetland structure, vegetation type, underlying geology, and hydrological regimes (Adame and Fry, 2016, Kobayashi et al., 2013). Inundation patterns of Australian wet-dry ephemeral wetlands can vary considerably, both temporally and spatially. Some wetlands can remain inundated for more than 6 months, contain deep water, and support elevated levels of aquatic primary production. Other wetlands are briefly inundated, shallow and turbid, with short-lived primary production (Pettit et al., 2017a). Critical to regulation of carbon storage is the seasonal hydrological regime, and whether the wetland is permanent or ephemeral (Beringer et al., 2013, Page and Dalal, 2011). Permanent wetlands tend to remain carbon sinks under seasonal hydrology, though not in all cases. For ephemeral wetlands, the predictable wet season flooding may change carbon stores, moving wetlands from sink to source (Beringer et al., 2013, Page and Dalal, 2011). In Morris et al. (2013) Mediterranean ephemeral wetland carbon flux shifted from source during dry season low flow to sink during wet season high flow.

Analysing and modelling planktonic primary production assists in understanding their composition and regulating processes in hydrologically dynamic wetlands with unpredictable flood pulses. During wet phases, Kobayashi et al. (2013) concluded semi-arid wetlands are net carbon sinks during their wet phase, due to the net autotrophy of planktonic metabolism. Water column net primary productivity, and thus aquatic carbon sink or source, is dependent on nutrient (nitrogen and phosphorous), and dissolved organic carbon loadings, which varies across distinct wetland components (Kobayashi et al., 2013). The Australian wet-dry tropics is considered a low nutrient environment, partly due to its ancient and weathered geology (Pettit et al., 2017a). Stream nitrogen and phosphorous concentrations are low, with the highest concentrations found in end-of-dry season disconnected waterholes (Townsend et al., 2012).

Aquatic metabolism in the water column, as part of carbon cycling and transformation in wetlands, can be analysed by measuring the difference between phytoplanktonic photosynthetic oxygen production and oxygen consumption over a specified length of time (Hoellein et al., 2013). Planktonic metabolism enables carbon transformation in wetland aquatic ecosystems, with the balance between GPP and PR determining its contribution to wetland carbon storage (Kobayashi et al., 2013). Due to the potential seasonal carbon storage variability, an integrated spatial and temporal research approach is required. Aquatic metabolism rates vary widely in freshwater wetland systems (Table 1.2). Whilst some values for Australian wet-dry tropics river systems are available, values for wetlands are rare in the literature.

Table 1.2 Aquatic planktonic metabolism as represented by gross primary productivity of phytoplankton (GPP) and planktonic respiration (PR), in selected freshwater wetland systems in mg C m^{-3} d^{-1}. Wet-dry tropics wetlands highlighted.

Wetland system	Wetland type	GPP (mg C m^{-3} h^{-1})	PR (mg C m^{-3} h^{-1})	GPP/PR	Reference
Daly River, NT Australia	**Wet-dry tropics**, low flow river	0.00-0.03	0.03-0.07	0.0-0.5	Webster et al. (2005)
Pantanal, Brazil	**Wet-dry tropics**, seasonal wetland	..	..	0.5	Hamilton et al. (1995)
Okavango River, Botswana	**Wet-dry tropics**, seasonal floodplain	0.0-33.0 (mean 5.2)	..	..	Lindholm et al. (2007)
Macquarie Marshes, NSW Australia	Semi-arid, channel and non-channeled floodplain	3.7-405.5 89.4 ± 9.2*	1.5-251 43.2 ± 5.6*	0.2-15.6 3.0 ± 0.3*	Kobayashi et al. (2013)
Macquarie Marshes, NSW Australia	Semi-arid, in-channel floodplain	8.0-120.6	13.3-30.4	..	Kobayashi et al. (2011)
Ryan's Billabong, Murray River, NSW Australia	Semi-arid, floodplain wetland	0.3-3.1	..	..	Boon and Sorrell (1991)
Everglades, Florida USA	Sub-tropical, seasonal wetland	25.1-166.5 (mean 95.81)	131.2-278.1 (mean 204.7)	0.5	Hagerthey et al. (2010)

Where available range plus mean ± std error (*) are provided.

1.4 Wetland conservation and management

Tropical terrestrial and wetland systems provide valuable ecosystem services including carbon storage (Junk, 2013, Mitsch et al., 2013, Scharlemann et al., 2014). However, many tropical terrestrial natural environments, including wetlands, are undervalued and considered as 'free' resources or wastelands. They are vulnerable to exploitation via economically valuable projects including forestry, agriculture, aquaculture, and urbanisation. These activities place wetlands at risk of disturbance-induced carbon release (Junk et al., 2013, Scharlemann et al., 2014), with tropical wetlands at particular risk (Junk, 2013). Often, wetlands are not effectively included in emissions reduction policies or climate change modelling (Bernal and Mitsch, 2013, Junk, 2013, Scharlemann et al., 2014). Global policies and practices for carbon in terrestrial soils and wetlands focus on carbon storage as a means of GHG emissions reduction and carbon offset programs (Vázquez-González et al., 2017), and are often part of overall climate change and biodiversity policy development. For example, international organisations, such as the Intergovernmental Panel on Climate Change, provide scientific synthesis of current global climate conditions, including potential impacts of land use and climate change on carbon dynamics (IPCC, 2014). The Ramsar Convention for globally significant wetlands recognises and promotes the importance of wetland environmental and ecosystem services, including carbon storage (Bernal and Mitsch, 2013).

Within Australia, wetland management practices are generally focused on semi-arid and temperate regions, and include erosion and degradation controls, restoration, and environmental flows to improve resilience and support wetlands as carbon sinks (DES, 2018). More recently, wetland policies now include recognition of the importance of SOC storage. Australian policies for wetland CS in tropical environments incorporate development of SOC monitoring frameworks. For Australian tropical terrestrial soils, the two most crucial factors in SOC losses are land clearing for economic exploitation and savannah burning (Cook et al., 2010). Therefore, policies focus on mitigating these factors and increasing CS.

1.5 Thesis aims and hypotheses

This study aims to understand the carbon dynamics of a permanently inundated and ephemeral wetland system in the wet-dry tropics of far north Queensland, Australia, called Kings Plains. The study will provide a quantified baseline understanding of carbon dynamics across this wetland environment. The focus of this study is restricted to the quantification of soil carbon stored in sediments, as a measure of long-term carbon storage, and carbon transformation through aquatic metabolism, as a measure of short-term carbon cycling. Analysis of atmospheric flux of carbon from sediments and biological sources of aquatic metabolism is outside the scope of this study.

Specifically, this thesis aims to:

a) quantify soil carbon concentrations and stock levels of the permanent and ephemeral wetlands, across different geomorphic units and at various sediment depths;

b) quantify the level of carbon transformation as measured by wetland aquatic metabolism during the dry and wet seasons; and

c) place this case study in a broader global context by comparing these levels to other wet-dry tropics wetland systems.

2 Regional Setting and methods

2.1 Regional setting

The wetland study area is situated in the Kings Plains Nature Refuge, ~50 km southwest of Cooktown, Queensland, at an elevation of ~130 m above mean sea level (amsl), in the upper catchment of the Normanby River system (Figure 2.1). The study area is a series of palustrine (DES, 2016) wetlands totaling ~31 km^2, located in an alluvium-filled deeply incised valley situated in the headwaters of the East Normanby River (Best and Dallwitz, 1963).

2.2 Geology, climate and vegetation

The regional geology is Devonian/Lower Carboniferous Hodgkinson Formation slates and greywacke, forming prominent hills on both sides of the study valley (Lucas, 1962). Resistant chert strike-ridges project into the valley, confining the valley to several partly closed basins, and forming three connected westward-draining wetlands, known as Kings Lake (KL), Top Plain (TP) and Bottom Plain (BP) (Figure 2.1). The valley follows the ancestral bed of the Annan River when it formed part of the Normanby headwaters (Heidecker, 1973). However, the Annan River no longer flows into the Normanby catchment due to stream capture from a Pleistocene era easterly regional geological tilt, and hydrological input to the valley has reduced (Best and Dallwitz, 1963, Bik and Lucas, 1963). As a consequence, the ~50 m deep valley has aggraded with fine clay and sand alluvium from low energy flows, and wetlands have formed (Best and Dallwitz, 1963). In the contemporary Annan River, there is evidence of carbon concentrated at >50 cm depth in vertically accreted dark silty sediments, reflecting slow sediment deposition (Kelly, 1997). There may be parallels with the post-stream capture low energy sediment deposition of the study area. Changed hydrological and geomorphic characteristics post-stream capture warrants investigation of possible carbon storage at greater depth than usually measured. Sampling depth for soil carbon ranges from 30 cm for cropped soils, to 2-5 m in deep tropical soils (IPCC, 2000), however, Page and Dalal (2011) suggest measurement to 100 cm depth is appropriate as SOC generally rapidly declines at greater depths.

KL is a ~10 km^2 permanent wetland, bedrock-confined on its northern perimeter, and which contracts during the dry season. TP is a ~6 km^2 shallow ephemeral wetland, inundated during the wet season, which progressively dries out in the dry season except for its central channel, which retains water but becomes disconnected (Hughes, 2017). BP is a ~15 km^2 shallow ephemeral wetland, inundated in the wet season, which progressively dries out in the dry season except for the western section of the central channel (Hughes, 2017). The Normanby catchment provides a major north-south biodiversity

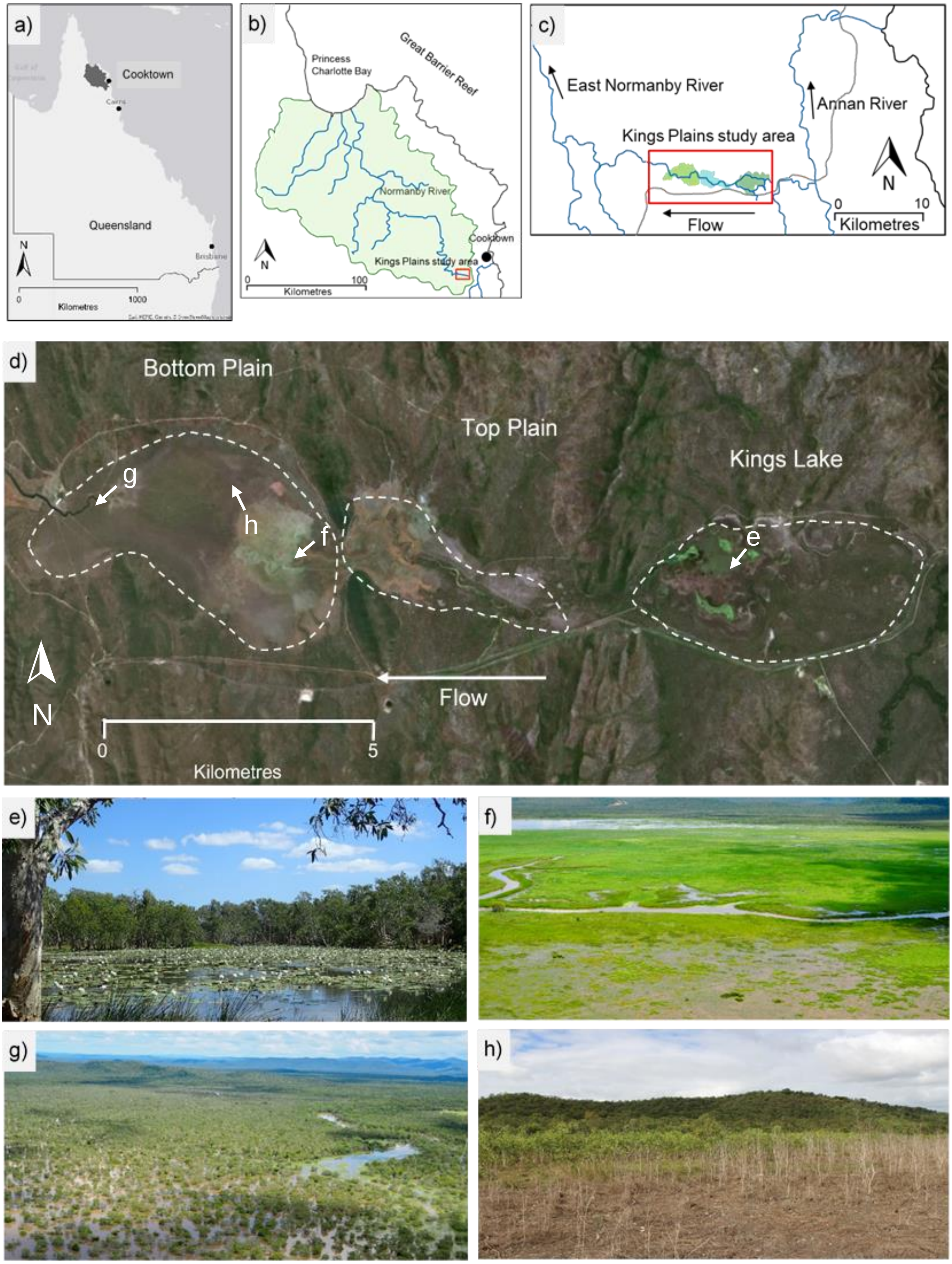


Figure 2.1 Study area location showing a) state of Queensland, b) Normanby catchment, c) Kings Plains study area location in catchment and relative to East Normanby and Annan Rivers, d) satellite imagery of the study site's three interconnected wetlands, Kings Lake, Top Plain and Bottom Plain including photo locations (ESRI, 2018), e) Kings Lake in dry season, view to SW, f) Bottom Plain in wet season, view to SW, g) Bottom Plain in wet season, view to SW, h) Bottom Plain in dry season, view to NW. Photos: e), h) D. Agnew; f), g) T. Hughes.

corridor between the Cape York and Wet Tropics bioregions (DEE, 2017). Within the study area, plant assemblage varies across the three wetlands. KL is dominated by a lower stratum of persistent emergent species including lotus lily (*Nymphaea nucifera*), giant water lily (*N. gigantea*), and widespread reed beds, with the wetland fringed by an upper stratum of broadleaved paperbark (*Melaleuca viridiflora*). Both TP and BP have dominant upper stratum species including *M. viridiflora*, wet season lower stratum small aquatic species (*Nymphoides spp., Caldesia oligococca*) and dry season native and exotic grasses (Queensland Herbarium, 2018). With a wet-dry tropical climate, mean monthly maximum temperatures at Cooktown (closest weather station) range from 26.2 °C (July) to 32.3 °C (December). Mean minimum monthly temperature range is 18.1 °C (July) to 24.5 °C (December). Average annual rainfall is 1485 mm, with monthly wet season average of 258 mm and dry season monthly average of 31 mm (BOM, 2018). During the wet season, the Mulligan Highway, which runs adjacent to the study area, can be periodically inundated at the eastern margin of the KL wetland, resulting in complete inundation. Here the Mulligan Highway is at its lowest elevation of 135 m amsl, ~3 m above the lake outlet.

2.3 Previous investigations

Previous investigations in the study area can provide valuable sedimentological information. Mineral exploration studies were undertaken during the 1960s, obtaining sediments from drill holes drilled to ~50 m depth at the KL wetland (Best and Dallwitz, 1963). In 2016, five bank exposures were excavated at selected locations to a depth of ~2 m using a backhoe excavator to identify sedimentology and stratigraphic changes. Samples from the bank exposures were analysed for qualitative mineralogy by CSIRO laboratories (Raven, 2016).

2.4 Topographic surveys and sampling plan

A geomorphology survey was undertaken using light detection and ranging (LiDAR) digital elevation model data, digital surface model data (JAXA, 2018), Google Earth satellite imagery, aerial photographs, and previous mineral exploration studies reports to determine the landscape components and geomorphic units in the study area. The output from this process was combined in ArcMap 10.6 (ESRI, 2018) to assist in detailed geomorphic mapping of the study area and used to determine cross-sectional and longitudinal profiles. Valley setting and margins, channel bed longitudinal profile, and the type and arrangement of channels, paleochannels, wetlands and floodouts in the system were established. Geomorphic units were identified according to Brierley and Fryirs (2005) .

Sampling locations within KL, TP and BP (Figure 2.2) were identified according to their different hydrological and geomorphological characteristics, and a twofold sediment sampling plan devised:

a) sediment cores were required for sediment characterisation and sedimentology, and for soil carbon analysis. Geomorphological diversity may result in considerable spatial and at-depth soil carbon variation; therefore, analysis of numerous samples was required; and
b) surface sediment samples were required for an aquatic metabolism experiment to determine a snapshot of dissolved organic carbon concentration through ecosystem metabolism. Sediment samples were collected twice, in the dry and wet season, to assess seasonal variability.

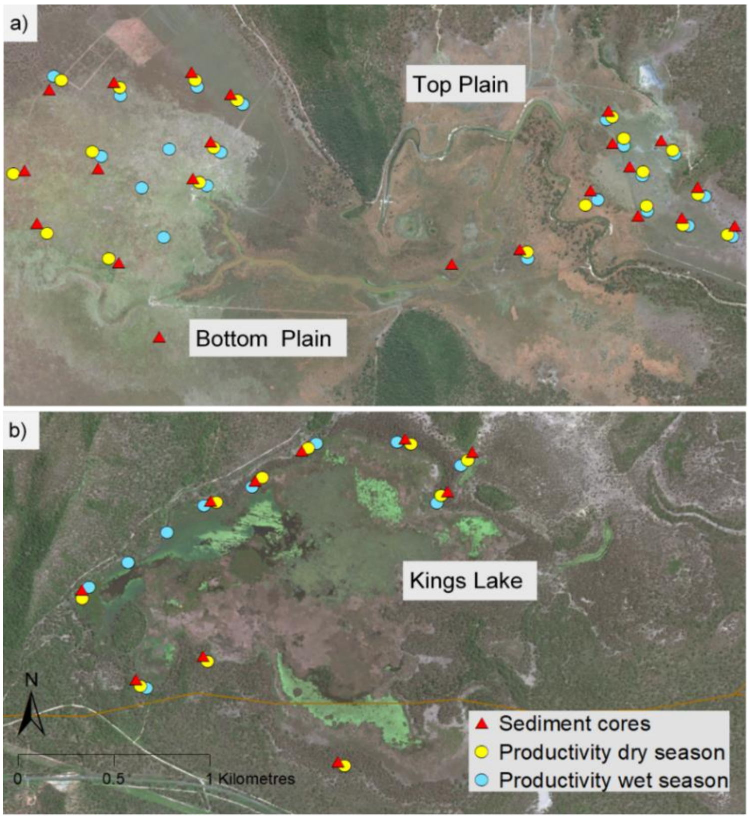


Figure 2.2. Sampling locations for sediment cores and wet and dry season aquatic metabolism surface sediment samples at a) Top Plain and Bottom Plain and b) Kings Lake (ESRI, 2018).

Sampling locations were chosen as representative of each wetland, however, at KL we were constrained by inaccessibility to the wetland's eastern and south-eastern perimeter. Where possible, cores and metabolism samples were taken within ~5 m proximity of each other, however, this was not possible for all metabolism samples due to wet season non-accessibility. The study area is inundated during and for several months post-wet season. Vehicular access was required for sediment core sampling. TP and BP usually dry out sufficiently for vehicular access by ~July (Hughes, 2017), therefore accessibility for sediment core sampling was restricted to August to November. Foot access, possible from May to November, was required for metabolism sampling.

2.5 Field sampling methods

2.5.1 Sediment cores for sedimentology and soil carbon

To describe and map sediment characteristics across the three wetlands, 20 x 100 cm sediment cores were taken at 10 different sampling locations at each wetland, totaling 60 x 100 cm sediment cores across the study area. On BP and TP, samples were taken on the floodplain, not in the channels. At KL, samples were taken as close to the water's edge as possible. At each sampling location, replicates were taken, one for sedimentology and soil carbon analysis, and one for bulk density analysis. Core length of 100 cm was chosen to enable comparison with soil carbon peer-reviewed literature. Additionally, 3 x 300 cm sediment cores were taken, one at each wetland. Sediment cores were taken using a Christie Soil Sampler, a 4WD utility truck-mounted hydraulic hammer-powered 50 mm push tube, providing a 38 mm diameter sample. The tube is high tensile chrome moly tubing with forged cutting tips. For the 100 cm cores, each undisturbed sample was extracted from the push tube, placed in a semicircular PVC channel, measured and divided at 10cm intervals and sealed in labelled double ziplock bags to prevent moisture loss. Where discrete sedimentary facies were discerned in the core, these were bagged separately. There was minimal compression of the cores. The 300 cm cores were divided into 3 x 100 cm sections, sealed intact as cores to prevent moisture loss. No visible charcoal samples were identified in the cores for radiocarbon dating. In total, 33 cores were collected, all 33 were required for soil carbon analysis, and representative samples were required for sedimentology. Samples were described and characterised in the laboratory. Using ArcGIS, six transects (Transects A – F) were established to examine the structure of the sedimentary alluvial valley fill, incorporating permanent and ephemeral channels, floodplains and floodouts. Along these transects, bank exposures and sediment cores were used to analyse sedimentology. Cores used for both soil carbon and transect sedimentology analysis include one of the following suffixes (A, B, C, D, E, F) on their identifying code.

2.5.2 Surface samples for aquatic metabolism

The study area's remoteness presented logistical issues for aquatic metabolism analysis. Instead of *in situ* water column analysis, an alternative method was utilised whereby sediment samples collected from the field were inundated under laboratory conditions (Knowles et al., 2012, Kobayashi et al., 2009). This is effectively a standing water biological oxygen demand (BOD) experiment for lakes applied in a mesocosm environment. At each wetland, 10 sampling locations were identified, and 10 x 150 g surface sediment samples were taken at random within a 10 x 10 m grid cell at each location. The 10 replicate samples were collated, resulting in 10 x ~1.5 kg samples from each of the three wetlands. Samples were refrigerated to avoid microcosms that could affect major biological change and were transported to the laboratory within two days of sampling. Inundation of sediment samples was commenced within 7 days of collection.

Dry season samples were collected in September 2017. Wet season sampling took place in May 2018, immediately site access was possible. Air temperature at sediment collection time was a consideration; average daily September Cooktown temperature range was 22.5-28.7 °C, and average daily May temperature range was 22.5-28.7 °C (BOM, 2018), both of which fell within acceptable parameters. Wet season sampling was undertaken as close as possible to dry season locations, however, increased water levels meant some dry season sample locations could not be accessed, so sampling occurred in nearby locations.

2.6 Laboratory methods

2.6.1 Sedimentology

Characterisation and stratigraphy of the core samples was analysed in the laboratory using standard sediment analysis (Appendix 1). Grain size, sorting and mineralogy were identified using a hand lens, colour determined according to soil colour charts by Munsell Color (2000) and facies described using Miall (1999) The samples and their stratigraphy were compared to the drill holes (Best and Dallwitz, 1963) and bank exposures from previous investigations. Two Bottom Plain bank exposure samples were sent to CSIRO laboratories for qualitative x-ray diffraction (XRD) analysis to determine mineralogy. Approximately 10 g of each sample was oven dried at 60 °C for several hours. 1.5 g of each oven dried sample was ground for 10 minutes in a McCrone micronizing mill under ethanol. The resulting slurries were oven dried at 60 °C then thoroughly mixed in an agate mortar and pestle before being lightly back pressed into stainless steel sample holders for XRD analysis. XRD patterns were recorded with a PANalytical X'Pert Pro Multi-purpose Diffractometer. Qualitative analysis was performed on the XRD data using CSIRO XPLOT and PANalytical HighScore Plus search/match software (Raven, 2016).

2.6.2 Soil carbon and moisture

For SOC analysis, SOM content (%) was determined for each 10 cm soil carbon sediment core sample by loss on ignition (LOI) method and using formulae described in Appendix 2. Sampling from the centre of each of sediment core was undertaken to avoid obtaining contaminated sediment from the outside of the cores. Each 10 cm sample was split lengthwise into quarters, and an ~10g sample obtained by slicing along the full length of the internal portion of the one of the quartered 10 cm core samples to ensure uncontaminated sediment (Appendix 1). This enabled measurement of SOC linearly throughout the entire core (and averaged for each 10 cm section) rather than at discrete intervals between samples. Visible vegetation residues in the samples were discarded. LOI was performed on 360 measured samples using a Lindberg Blue furnace. Each ~10 g sample was homogenized using mortar and pestle and a sample weighing between 4 and 7 g obtained. These were dried at 105 °C for 24 hours, weighed, then heated at 550 °C for 5 hours, to oxidize organic carbon, and weighed immediately after removal from the furnace after cooling to 105 °C to determine LOI_{550} and thereby estimating SOM content. Samples then underwent further ignition at 950 °C for 1 hour, to oxidize inorganic carbon, and weighed immediately after removal from the furnace after cooling to 105 °C. This estimated LOI_{950} and thereby carbonate (inorganic) content (Rayment and Lyons, 2011). Use of a desiccator was trialed and rejected, as there was no significant difference between weighing immediately after cooling to 105°C or after an additional one hour in a desiccator.

Given the substantial number of samples to be analysed, LOI was considered the most appropriate method of determining SOC content (%), with appropriate laboratory equipment readily available. In general, the conventional SOM to SOC conversion factor of 0.58, i.e. SOM = 58 % carbon, is not supported by empirical evidence and considered too high (Pribyl, 2010). The more conservative conversion factor of 0.50, as recommended by Pribyl (2010) was utilised. Variation in the conversion factor occurs from both natural soil composition variability, and from differing calculation methods and procedures to estimate carbon content. In order to avoid overestimating SOC, use of the 0.50 conversion factor was considered appropriate given potential problematic accuracy issues of LOI due to ignition temperature and duration, and structural water loss in clay rich soils (>35 % clay) (McKenzie et al., 2004, p. 364). Structural water loss in clay-rich soils heated to high temperatures can lead to overestimation of SOM. Hoogsteen et al. (2015) recommends a clay correction factor of 0.09 for samples dependent on furnace temperature and duration, and was considered relevant for samples ignited from 550 °C to 950 °C, so accordingly was used in this study. SOC density (g C cm^{-3}), SOC concentration (g C kg^{-1}) and SOC stock (kg C cm^{-2}) were calculated using formulae described in Appendix 2. Total SOC stock in each core was calculated by summation of incremental samples.

Soil moisture and DBD were determined from replicate cores taken adjacent to the sediment cores for soil carbon analysis. 360 samples were analysed following procedures according to Blake and Hartge (1986) and using formulae described in Appendix 2. Where possible, complete 10 cm samples, rather than subsampling, were used to enable DBD measurement linearly rather than incrementally. Subsampling of a 10 cm core section was done where there was soil fragmentation or cracks in the sediment cores. Samples were dried at 105 °C for two hours and weighed (Rayment and Lyons, 2011) to determine soil moisture. 105 °C is considered a standard temperature commonly used for drying wetland soils, enabling comparison with other studies (Bernal and Mitsch, 2008). A batch of 30 replicate samples was dried for a further two hours at 105 °C; there was no further weight loss, therefore constant weight had been reached after the initial two hours drying.

2.6.3 Aquatic metabolism

For aquatic metabolism analysis, the BOD method (Wetzel and Likens, 2000) was undertaken in the laboratory to measure GPP and PR. This was performed twice, firstly using 30 dry season surface sediment samples collected in the field, and again with 30 wet season samples. Each 1000 cm^3 sample was placed in a clean plastic 5.2 L bucket, forming a 4 cm sediment depth, and gently inundated with 4 L of de-ionised (DI) water, forming an ~14 cm water depth (Appendix 3). The DI water was added carefully to the bucket, to minimise disturbance to the sediment, and reduce potential turbidity. Use of DI water standardised the water quality and minimised potential contamination by microbial organisms. The sediments were incubated for five days to activate metabolism, with an imposed diurnal light cycle (12 hours light/dark) with light intensity of ~4000 LUX, and constant 25 °C ambient temperature. Three control samples of 4 L of DI water were also included. At the end of five days, the 24-hour BOD procedure was undertaken. For each sample, clear and dark 300 ml bottles were carefully filled with sample water, without air bubbles, sealed, and incubated for 24 hours, under the same light and temperature conditions as the five-day inundation. Salinity (electrical conductivity) and pH were measured at completion of the 24-hour BOD incubation using standard instruments. Dissolved oxygen (DO) concentration of each sample was measured at the beginning and end of incubation using a DO meter (YSI Model 5100 DO/Temperature Meter, YSI Inc., Ohio). From these measurements, GPP and PR for each sample was calculated, using formulae defined in Appendix 2.

2.6.4 Statistical analysis

Comparison of SOC stock, SOC concentration, soil moisture, GPP, RP and GPP/PR were made among the wetlands using one-way analysis of variance (ANOVA), with the significance level of p=0.05. When one-way ANOVA showed significant results, post-hoc Tukey's pairwise comparisons were used to compare each pair of means. Prior to analysis, data values for SOC concentration, GPP and PR were $\log_{10}$-transformed to stabilize the variance and meet the assumption of normality (Zar, 2013). Comparisons of SOC concentration were made among wetlands and at varying depths using one-way repeated analysis of variance (repeated measures ANOVA), with the significance level of p=0.05. This method was used as depth groups were related, and not independent of each other. All statistical analyses were performed using Minitab 18 (Minitab Inc, 2017), except for repeated measures ANOVA, which was performed using SAS PROC GLM statistical software (SAS Insitute Inc, 2017).

3 Results

3.1 Geomorphology

Kings Plains is a series of wetlands in a valley setting which alternates between partly-confined and laterally-unconfined. The wetlands are situated in broad alluvial filled basins up to ~4100 m in width, separated by narrow confined sections to ~450 m width at steep protruding chert ridges, resulting in three westward-draining connected wetlands (Figures 3.1a, 3.1c). Typically, the alluvium in the valley is ~50 m deep (Best and Dallwitz, 1963). The upstream wetland, KL, contains a permanent lake covering ~1/3 of the 2700 m wide valley basin, and several backswamps, all fed by ephemeral tributaries from surrounding hillslopes. The two downstream wetlands, TP and BP, contain a seasonally disconnected main channel. During the wet season, the KL valley basin is inundated. Downstream of the lake outlet, the valley constricts to 450 m in width. A channel zone ~2200 m in length continues downstream, connecting KL and TP, and comprising a main channel and floodchannels and variable valley width to 1500 m. The main channel, with an overall low sinuosity of 1.09, flows during the wet season, completely drying out in the dry season. This channel flows into TP, a 2000 m wide relatively flat basin, containing a meandering (average sinuosity 1.55) channel, floodplains, backswamps, and ephemeral tributaries fed from surrounding hillslopes. During the wet season, this channel contains flow. During the dry season, the channel becomes disconnected, but still retains some standing water. Sinuosity increases from 1.13 to 2.60 as the channel approaches TP's downstream end, where valley confinement to ~600 m width occurs once more, again from steep protruding ridges. The channel flows through this confined zone into BP, a broad flat valley basin up to 4100 m wide. Here the main channel divides and narrows, dispersing into distributary channels and a large floodout area across the floodplain. A palaeochannel is evident downstream of the left-hand branch of the main channel. At the downstream end of BP return channels flow into a re-formed main channel.

The longitudinal profile of the study area, prepared using LiDAR, produced a very low valley slope of 0.0004 (measured over 22.5 km) (Figure 3.1b). Within each wetland there was minimal change in slope. The steepest section along the longitudinal profile was in the channel zone, between KL and TP, directly downstream of the first confined section. Elevation changed from 132 m amsl at the lake outlet to 126 m amsl at the beginning of TP producing a valley slope of 0.0013 (measured over 4.7 km). Elevation change across the extent of TP and BP was ~2 m, mostly occurring in the downstream section of BP, and producing a very low valley slope of 0.0001 (measured over 11 km).

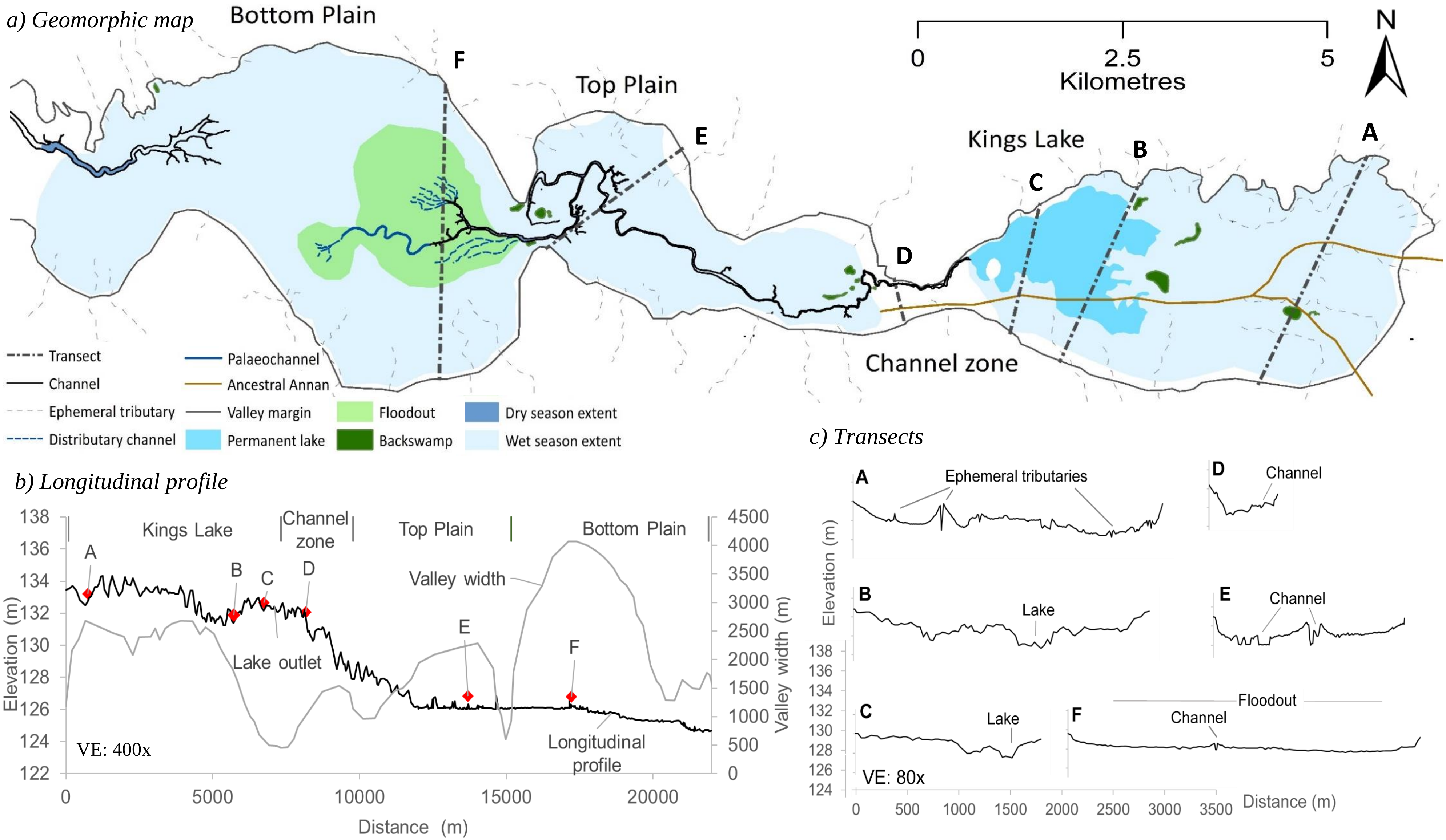


Figure 3.1. Valley morphology of Kings Plains study area, a) geomorphic map showing transects and geomorphic units: alluvial valley fill, channels, ephemeral tributaries, distributary channels, palaeochannels, ancestral Annan River, permanent lake, floodout, backswamps, and dry and wet season wetland extent, b) longitudinal profile, vertical exaggeration (VE) 400x, showing transect locations, wetland basin and channel zone locations, overlaid with valley width, and c) transects showing cross-sectional change downstream showing broad wetland basins, narrow channel zone, major geomorphic units (VE 80x).

3.2 Sedimentology

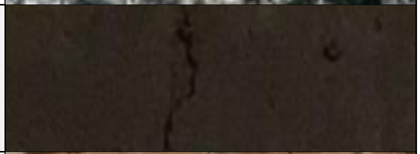
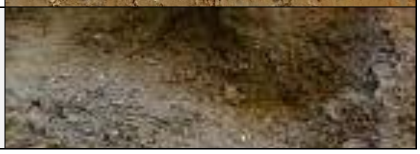
3.2.1 Major sedimentary units

The sedimentary structure within each wetland was relatively uniform, however, there was variation between the three wetlands. An overall fining of sediment occurred downstream through the three wetlands from silty/sandy clays found at KL, light medium/silty clays at TP, to very fine heavy clays at BP (Appendices 4, 5). The exception to this was in the steeper channel zone immediately downstream of KL, where medium and coarse sands were also deposited. Examples of the most common sediment classes identified are shown in Table 3.1. Excluding the BMR drill holes, there was also a reduction in stratigraphic variation moving downstream, from an average of four different sediment classes per sediment core at KL, to organic rich clays overlaying uniform massive heavy clays at BP. Across the study area, surface sediments generally comprised dark brown or yellowish brown coloured clays or clay loams with decaying organics, with highest organic content around KL. These overlaid massive clays, except in the channel zone which comprised interbedded clays and sands. Around KL, the dark organic rich clays transitioned into orange or black mottled grey clays, indicative of their saturated environment. On TP and BP, massive clays remained darker in colour with less orange mottling and increased black mottling (Appendices 4, 5). Generally, minimal concentrations of very fine roots were found below 30 cm depth, and in the case of TP and BP largely disappeared by 70 cm depth. Ferruginous-manganese nodules, indicative of periodically saturated soils, were present in most sediment samples of TP and BP. The clays of TP and BP also exhibited the characteristics of vertosols i.e.:

- clay content >35 %,
- shrink-swell properties: when the soils are dry, open cracks occur, at least 5 mm wide and extending upward to the surface, and
- slickensides occurring at depth (Isbell, 2016).

Bedrock was reached at a depth of 45 m in one of the drill holes of Transect A, and there was evidence of saprolite in proximity to the northern and western margins of KL. Bedrock was not reached in the channel zone, or on TP or BP.

Table 3.1 Characteristics of sedimentary classes found at Kings Plains using Miall (1999) , Munsell Color (2000) is representative of sedimentary class; colours in **bold** represent colour in images.

Sediment class	Grain size range	Facies classification incl. transect	Indicative Munsell colours	Images of representative sediment classes (scale = centimetres)
Silty-sandy clay loams	vfU-fU	**Fm** (massive, desiccation cracks) (E) **Fsm** (massive) (B, C, D)	**7.5YR 3/2 dark brown (E)** 2.5Y 4/4 olive brown (B, C, D)	
Organic rich clays	vfL-vfU	**Fm** (massive, desiccation cracks) (F)	**2.5Y 4/3 reddish brown**	
Heavy clays	vfL	**Fm** (massive, desiccation cracks) (F)	**2.5Y 3/3 dark reddish brown** 10YR 4/2 dark greyish brown	
Light medium clays	vfL-vfU	**Fm** (massive, desiccation cracks) (E) **Fsm** (massive) (B, C)	**7.5YR 5/4 brown (B, C)** 7.5YR 3/1 very dark grey (E) 10YR 2/1 black (B, C)	
Silty clays	vfL-vfU	**Fsm** (massive) (B, C, E) **Fl (**fine lamination) (D)	**Matrix: Gley1 6/N greyish brown Mottling: 10YR 6/8 brownish yellow (C)** 7.5YR 3/2 dark brown (B, C, E)	
Sandy clays	fL-cU	**Fsm** (massive) (B, C)	**10YR 5/8 yellowish brown (B)** Matrix: Gley1 5/N greyish brown Mottling: 5YR 5/8 yellowish red (C)	
Sands	mL-cU	**Fl** (fine lamination) (D) **Ss** (scour fill) (D)	**10YR 4/4 dark yellowish brown**	

3.2.2 Sedimentology by transect

3.2.2.1 Transect A

Transect A was located upstream of the permanent lake in the KL wetland in a relatively flat area (Figures 3.1a, 3.1c, 3.2, 3.3a). This transect contained cores BMR1_A and BMR2_A, two BMR drill holes (MRK6 and MRK7) drilled for alluvial tin exploration in the ancestral Annan River valley (Best and Dallwitz, 1963). Analysis of the drill hole results indicated 10-20 m of red and white mottled light medium clays, overlaying lenticular interbedded sandy clays and fine and coarse sand units to 40 m deep, of which the bottom 6-10 m contained interspersed gravel beds containing cobbles 3-5 cm in size (Figure 3.3a, Appendix 4). In BMR2_A a massive coarse sand unit from 23-42 m was interspersed with some clays, overlaying green/blue/grey clays containing up to 8 cm b-axis gravels from 50 m. Bedrock in this transect was reached at 46 m in drill hole BMR1_A when grey slate fragments were encountered. The sand beds were commonly saturated with water, with 'running sand' encountered (Best and Dallwitz, 1963). Some clay interbeds were dark grey to black, with a fetid odour, and contained carbonized plant remains (Best and Dallwitz, 1963).

3.2.2.2 Transect B

Transect B was located midway along the KL wetland intersecting the eastern end of the permanent lake from south to north (Figures 3.1a, 3.1c, 3.2, 3.3b). Cores were taken from both southern and northern lake edges. Midway through the transect there was an elevated area, splitting the permanent wetland in two. Surface sediments were organic-rich dark brown/black silty clay loams and/or black silty clays with fine roots ranging from 0-30 cm (Figure 3.3b, Appendix 4). Beneath this was a light medium clay layer extending to 70-100 cm deep, which graduated in color from dark brown/black to yellowish-brown with orange mottling from 60 cm. This overlaid orange mottled sandy clays in two cores, on opposite sides of the lake. The long core, KL7_3m_B contained orange mottled greyish-brown to light grey sandy clays from 120-280 cm. KL3_B and KL10_B were slumpy from 20-40 cm. Facies for this transect were uniformly Fsm, massive, backswamp deposits. Some angular gravels to 5 mm b-axis were present at the base of KL10_B, possibly deposition from flows sourced from the proximal valley margin hillslope. Bedrock was not reached in this transect.

3.2.2.3 Transect C

Transect C was located towards the southern end of the KL wetland, intersecting the western end of the permanent lake from south to north (Figures 3.1a, 3.1c, 3.2, 3.3c), and again, cores were taken from both sides of the lake. This section of the wetland also contained elevated terrestrial areas ~300 m west of the transect, indicating underlying bedrock and a transition from wetland to ridge. Cores in this transect were generally uniform, with a thin 0-20 cm surface layer of organic-rich black silty clay loam overlaying orange mottled yellowish-brown (southern side) or black (northern side) silty clays to 30-70 cm (Figure 3.3c, Appendix 4). This was followed by a layer of light medium clay 20-40 cm thick. There was significant color variation in this layer, with orange mottled grey/brownish yellow clays at KL4_C and KL8_C, and black slumpy light medium clay at KL9_C. These layers were moistest in the study area, with up to 45 % moisture content. Underneath this layer was a 40-60 cm layer of sandy clay, again with similar color variation. KL8_C and KL9_C sandy clays remained orange mottled grey/brownish yellow in colour whereas KL9_C transitioned to orange mottled reddish/yellowish browns. Facies for this transect were uniformly Fsm, massive, backswamp deposits. KL4_C contained a large tree root at 90 cm and transitioned to clast-dominated with 1 cm b-axis angular gravels from 110-130 cm, indicative of saprolite.

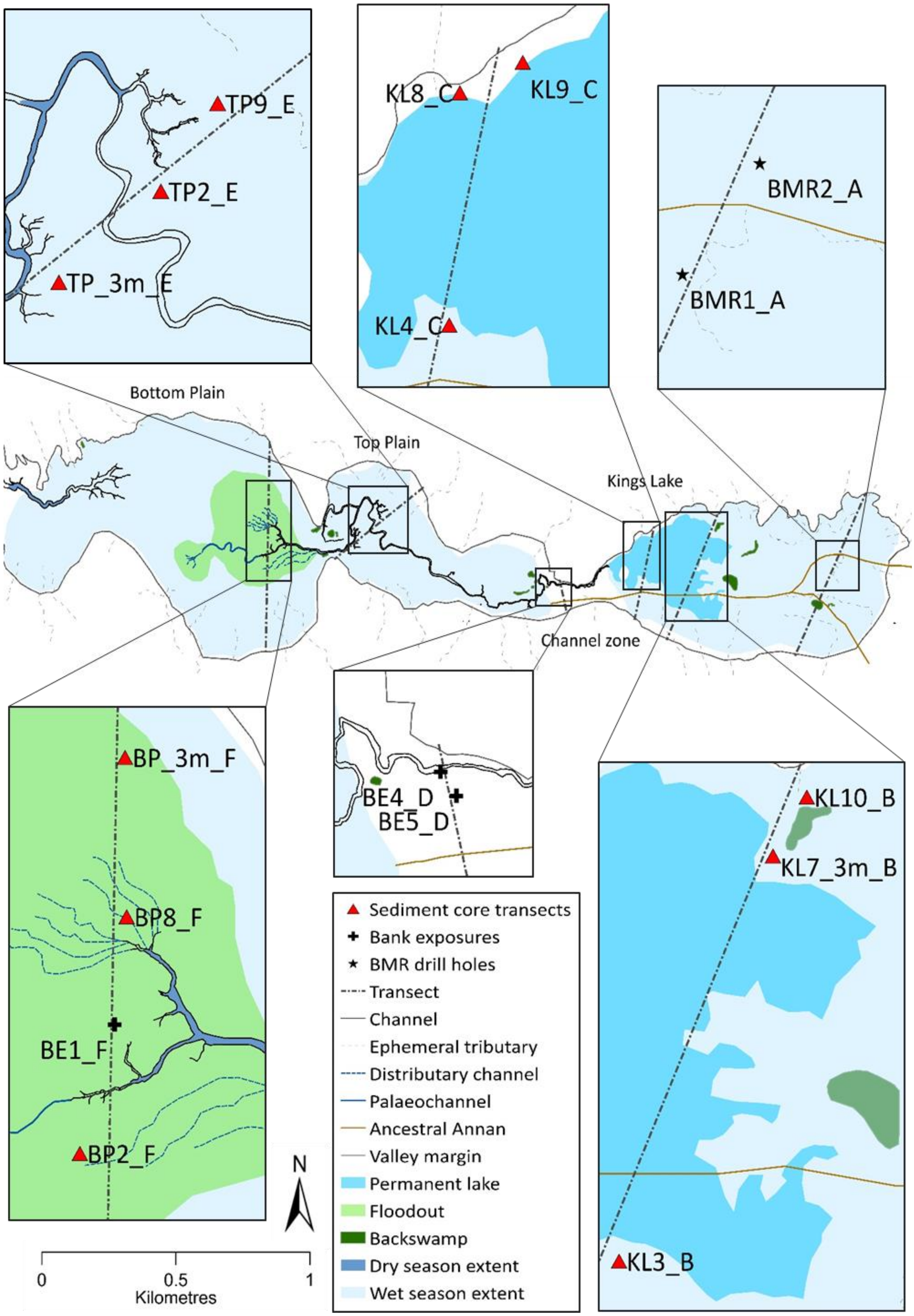


Figure 3.2 Location of transects, sediment cores, bank exposures and drill holes (ESRI, 2018).

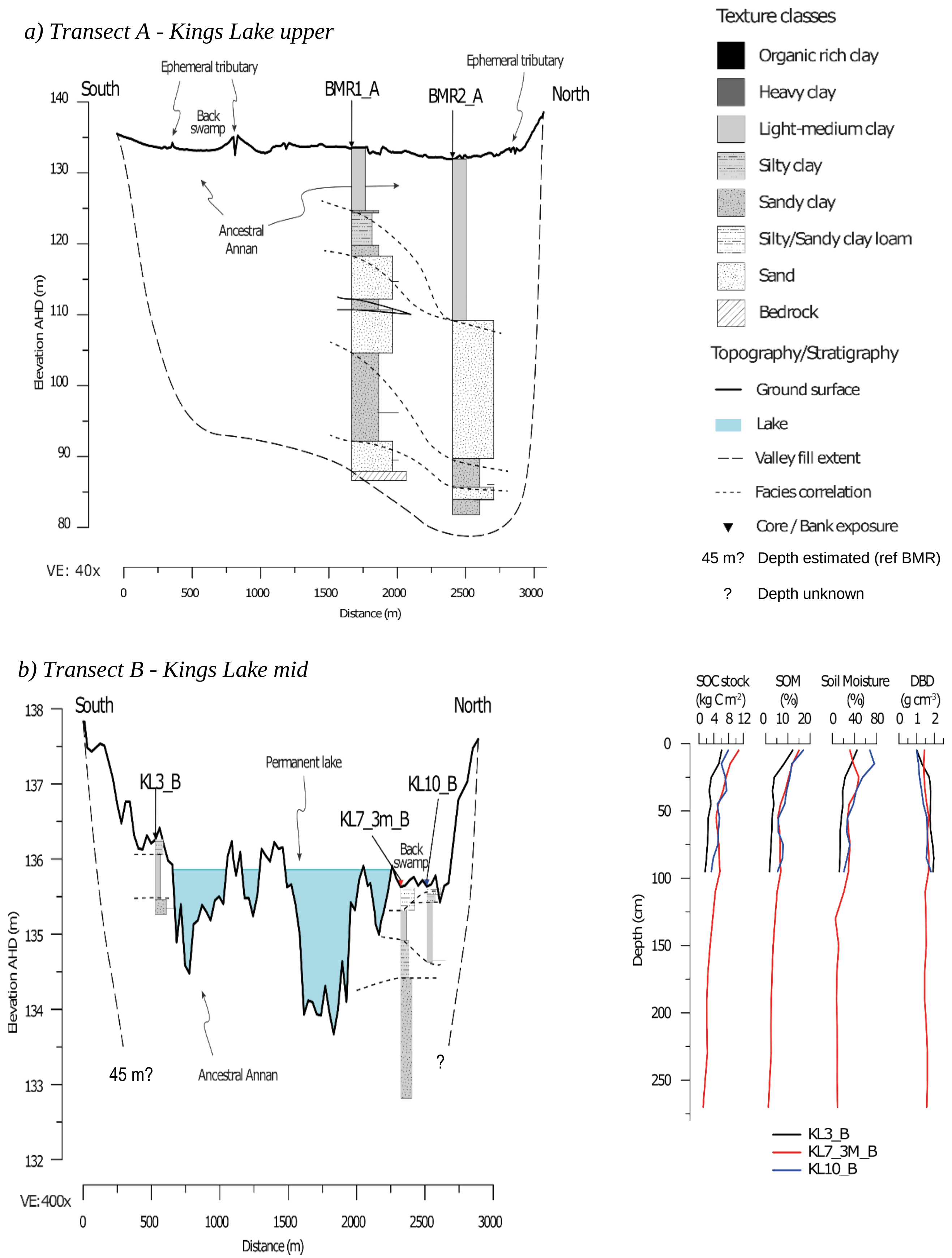
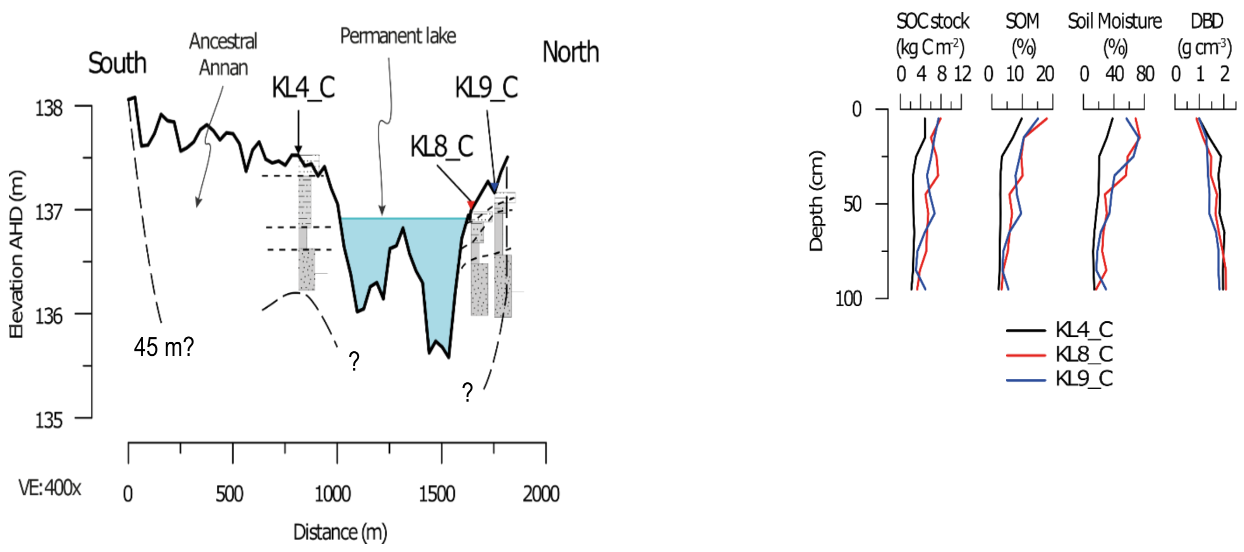


Figure 3.3 Correlated cross-sections (Transects A-F) with sediment columns (where available), and associated soil organic carbon (SOC) stock, soil organic matter (SOM), soil moisture and dry bulk density (DBD) plots of cores. Sediment columns describe texture classes. Transect A represents valley fill to ~50 m depth with a vertical exaggeration (VE) of 40x. Transects B-F show surface elevation to 6 m depth, to clearly display texture classes in sediment columns, and therefore the surface topography looks highly exaggerated (VE 400x). In reality, topography of transects B-F resemble Transect A. Refer to Appendix 4 for full description of sediment columns included in transects.
This page: a) Transect A, and b) Transect B.

c) Transect C - Kings Lake lower



d) Transect D - Channel zone

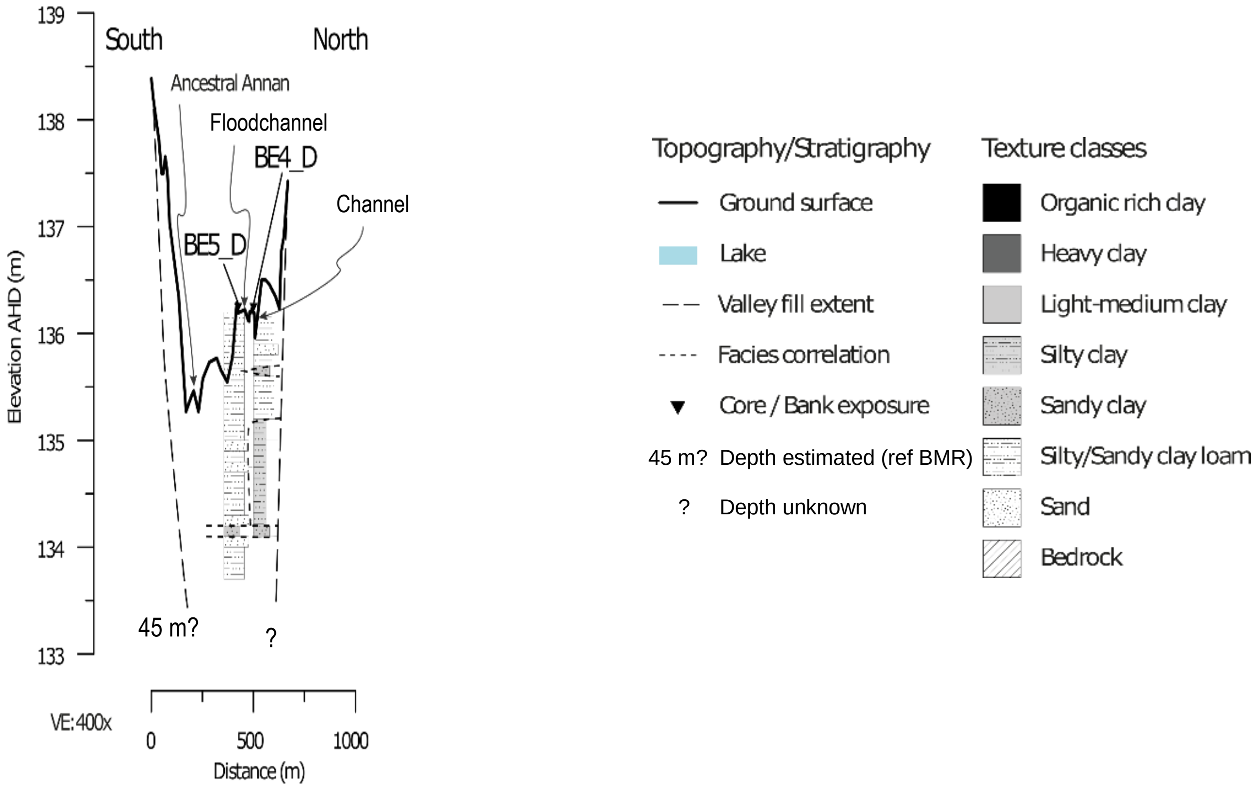
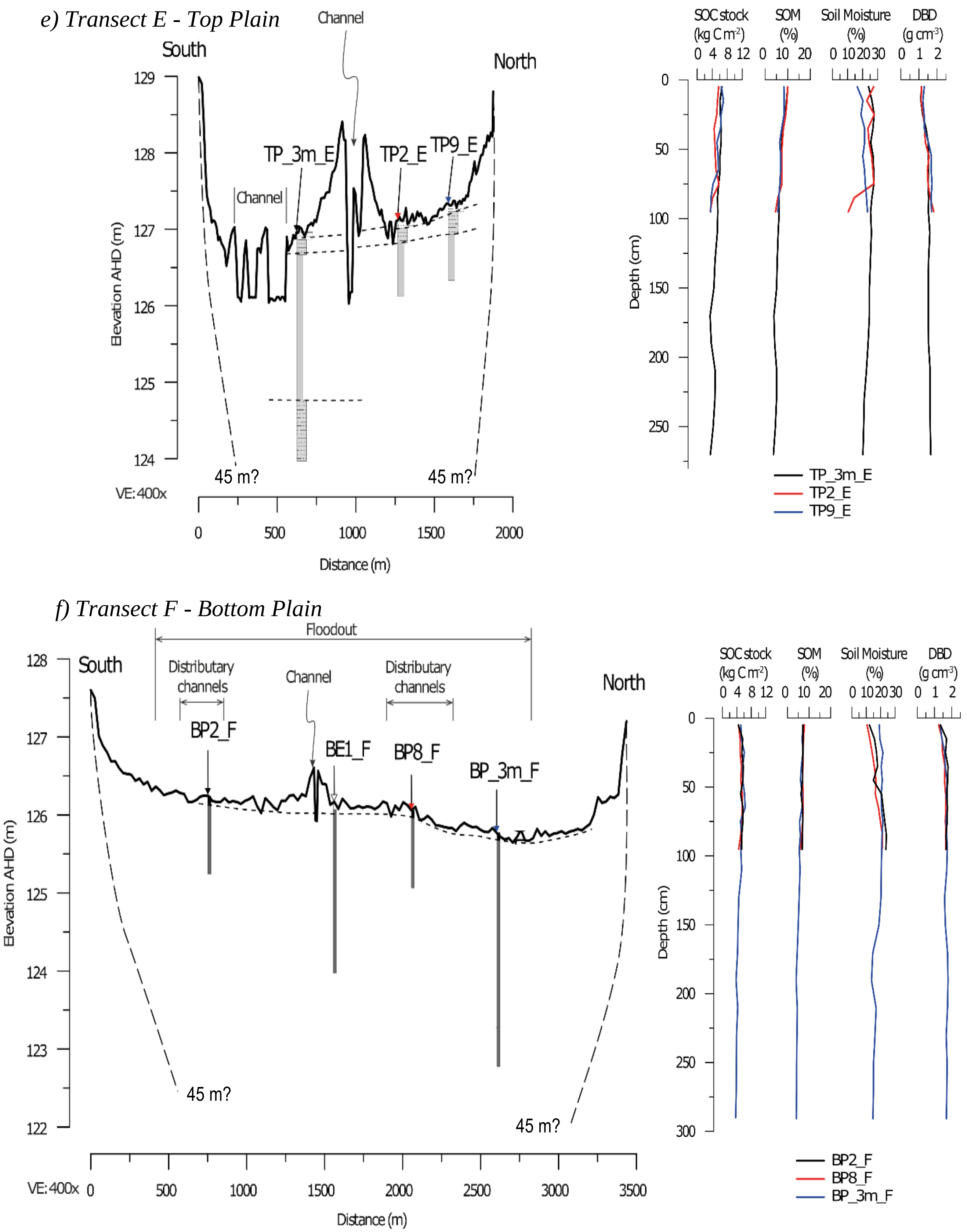


Figure 3.3 Correlated cross-sections with sediment columns (continued). This page: c) Transect C and d) Transect D



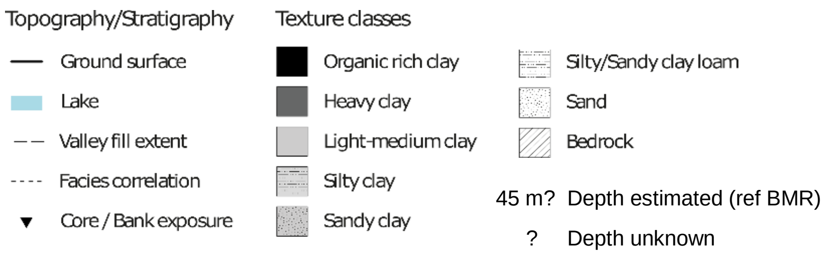
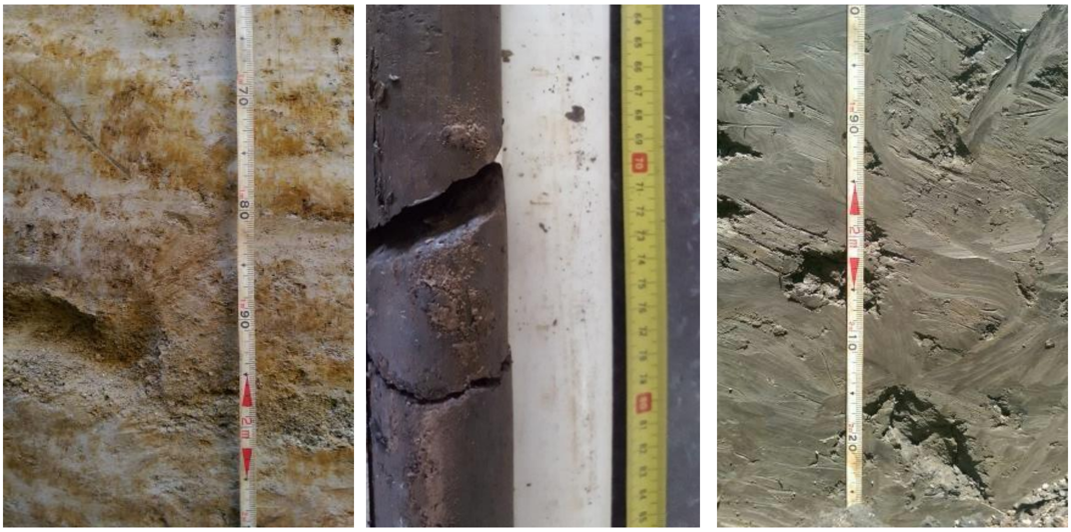


Figure 3.3 Correlated cross-sections with sediment columns (continued).
This page: e) Transect E and f) Transect F.



a) BE5_D sand layer b) TP2_E slickenside c) BE1_F massive heavy clay

Figure 3.4 Features observed in the sediment cores and bank exposures a) BE5_D bank exposure showing splayed coarse sand layer with 5 mm angular gravels at 200 cm depth, and b) TP2_E slickenside at 70-80 cm depth, and c) BE1_F massive heavy clays at 180-220 cm depth.

3.2.2.4 Transect D

Transect D was located in the channel zone downstream of the lake outlet, where the valley transitioned from KL permanent wetland to TP ephemeral wetland (Figures 3.1a, 3.1c, 3.2, 3.3d). Elevation at the southern end of the transect, where the ancestral Annan River flowed (Best and Dallwitz, 1963), was lower by ~2 m than the floodchannel and main channel area abutting the northern valley margin. Two bank exposures, BE5_D (floodchannel), and BE4_D (bench on main channel) comprised an array of coarser sediments than those found in the 1-3 m deep sediment cores elsewhere through the study area (Figure 3.3d, Appendix 4). In general, both bank exposures contained silty clay loams interbedded with thin layers of sand and silty/sandy clays, however, BE4_D contained a more varied stratigraphy than BE5_D. BE5_D was comprised of a thick 120 cm layer of dark brown silty clay loam with orange mottling from 30 cm, and nodules at 70-80 cm. This was followed by 120 cm of alternating orange mottled yellowish and olive brown sand/silty clay loam/sandy clay layers, most on average 10 cm thick. The distinct light olive and yellowish-brown sand bands contained fine to coarse sands with 5 mm angular gravels, were not level, and appeared to be splayed sides of the channel (Figure 3.4). Major tree roots finished at 70 cm, yet fine roots continued to 130 cm. BE4_D exhibited more stratification in the first 100 cm than BE5_D. It comprised dark brown silty clay loams, interbedded with distinct 5 cm layers of dark yellowish fine sand at 35 and 55 cm, and more diffuse 5 cm thick dark yellowish-brown silty/sandy clay layers at 20 and 45 cm. Underneath this was a 100 cm thick olive brown silty clay layer, with angular quartz gravels up to 8 cm b-axis from 100-120 cm, and a large tree root at 150 cm. Below this was a further layer of dry crumbly yellowish-grey sandy clay to 210 cm. Stratigraphy of these two bank exposures indicated an alternating sequence of low flow regimes, and channel infill of waning flood deposits.

Facies of the silty clay loams, and silty/sandy clays was Fl (waning flood deposits). Facies of the sands was Sm (massive, sediment gravity flow deposits). The sediments in this transect were drier than all other transects. Bedrock was not reached in this transect.

3.2.2.5 Transect E

Transect E was located at the western end of TP, where the channel meander is at its most sinuous. The southern end of the transect profile intersected the channel several times, and overbank deposition from the main channel was evident towards the centre of the transect (Figures 3.1a, 3.1c, 3.2, 3.3e). The sediment profile of TP was consistent across the cores. The floodplain was vertically accreted by overbank deposition of fine sediments during the wet season. Surface sediments of organic dark brown/black silty clay loams to 10 cm depth overlay a thin 20 cm layer of dark brown silty clays (Figure 3.3e, Appendix 4). Below this were massive reworked light medium clays which transitioned in colour from black to greys, with orange mottling from 40 cm, and black mottling from 60-80 cm. TP_3m_E, the 3 m long core, was comprised of black mottled light medium clays from 100 to 220 cm, and underneath this was orange and black mottled very dark grey silty clay to 300 cm. Facies for this transect were uniformly Fm (massive, overbank or waning flood deposits). Reworking of the sediment occurred from the shrink swell process. Cracking in the soil profile occurred in the two top layers to 30 cm, and a slickenside was evident at 70 cm depth in TP2_E (Figure 3.4). Sediment cores from the margins of TP, had slightly coarser sediments, indicating flows from the adjacent hillslopes. TP sediments exhibited the characteristics of vertosols. No visible charcoal was evident in the cores. Bedrock was not reached in this transect.

3.2.2.6 Transect F

Transect F was located on BP, across the end of the channel and through the floodout zone (Figures 3.1a, 3.1c, 3.2, 3.3f). Sedimentary units were consistent along this transect. Surface sediments to 10 cm were a dark coloured organic layer, overlaying uniformly massive dark grey reworked heavy clays from 10 cm to 100-300 cm (Figures 3.3f, 3.4, Appendix 4). Cracking in the soil profile occurred to 50 cm depth. Facies for this transect were uniformly Fm (massive, overbank or waning flood deposits). The floodplain was vertically accreted by deposition of fine sediments during the wet season via the floodout and distributary channels. For BE1_F, XRD analysis by CSIRO of a surface sample and a sample at 240 cm depth showed similar mineralogy (Raven, 2016). The surface sediment was co-dominantly quartz and smectite (likely montmorillonite) (sum >60 %), and at 240 cm the sediment was dominantly smectite (likely montmorillonite) (>60 %) and sub-dominantly quartz (20<60 %). BP sediments also exhibited the characteristics of vertosols. No visible charcoal was evident in the cores or bank exposures. Bedrock was not reached in this transect.

3.3 Soil carbon

SOM % content of soils across the three wetlands ranged from 1.7 to 18.0 % with a mean of 7.1 % ± 1.2 (SD) (Table 3.2). Soil carbon content was 97.3 % organic for all samples, with a mean 2.7% ± 0.7 (SD) carbonate material present (Table 3.2). As the majority of samples contained >35 % clay content, LOI_{950} was adjusted with the clay correction factor of 0.09 (Hoogsteen et al., 2015). Mean SOC stock to 100 cm depth across the three wetlands was 51.5 kg C m^2 ± 7.8 (SD), and 54.0 kg C m^2 ± 2.0 (SD), 51.6 kg C m^2 ± 3.6 (SD), 48.7 kg C m^2 ± 12.8 (SD) for BP, TP and KL, respectively (Figure 3.5, Table 3.2). Variability of SOC stock was highest in KL cores, compared to the relatively consistency of SOC stock in BP cores. There was no significant difference in SOC stock (to 100 cm) amongst the three wetlands (Table 3.3). Mean SOC stock to 30 cm depth across the three wetlands was 16.6 kg C m^2 ± 3.3 (SD), and 15.7 kg C m^2 ± 0.9 (SD), 15.9 kg C m^2 ± 2.8 (SD), and 18.3 kg C m^2 ± 4.7 (SD) for BP, TP and KL, respectively (Table 3.2). There was no significant difference in SOC stock (to 30 cm) amongst the three wetlands. Mean SOC stock in the 3 m cores was 84.0 kg C m^2, 87.6kg C m^2, 83.9 kg C m^2 for BP, TP and KL, respectively (Figure 3.3, Table 3.2). SOC stock in these cores declined with soil depth (Figure 3.5). SOC stock for 0-20 cm compared to the deepest 20 cm of the core declined by 64.5 %, 56.0 % and 87.6 % for BP, TP and KL, respectively (Table 3.2).

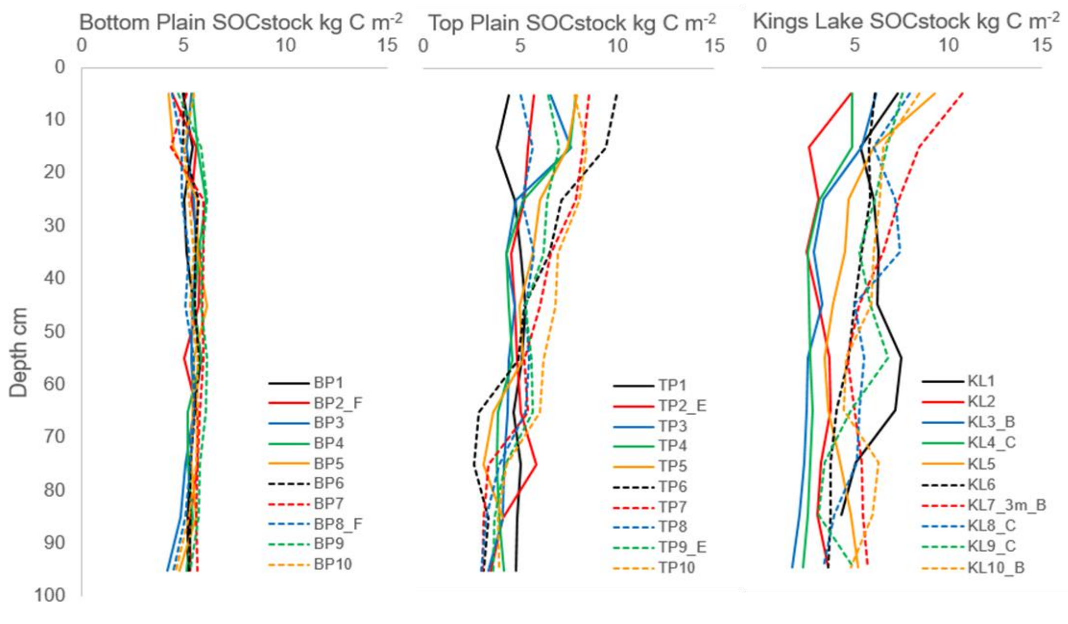


Figure 3.5 Soil organic carbon stock (SOCstock) (kg C m^{-2}) at 10 cm increments to 100 cm depth of sediment cores from Bottom Plain (BP), Top Plain (TP) and Kings Lake (KL). Subsampling from the full length of the internal portion ensured a clean representative sample, enabling measurement and analysis of SOC linearly rather than incrementally. All 100 cm sediment cores, including those forming part of the transects, were used in SOC analysis. Cores also used for sedimentology analysis of transects include one of the following suffixes (B, C, E, F) on their identifying code. Sediment columns and associated descriptions for all cores are included in Appendices 4 and 5.

Table 3.2. Organic matter, loss on ignition (LOI_{550}, LOI_{950}), moisture content, soil organic carbon (SOC) concentration and stock for the whole study area, and three wetlands, Bottom Plain, Top Plain, and Kings Lake. Mean ± standard deviation, with range and coefficient of variation (%) in parentheses.

	Total[a]	**Bottom Plain**	**Top Plain**	**Kings Lake**
LOI_{550} i.e. SOM %	7.1 ± 1.2 (3.8-9.0, 16)	7.1 ± 0.3 (6.8-7.5, 4)	7.6 ± 0.5 (6.6-8.4, 7)	6.6 ± 1.9 (3.8-9.0, 28)
LOI_{950} (adj by 0.09)[c]	2.7 ± 0.7 (1.0-4.9, 24)	3.4 ± 0.2 (3.2-3.6, 5)	3.4 ± 0.3 (3.0-4.0, 9)	2.2 ± 0.3 (1.7-2.7, 16)
DBD to 100 cm depth (g cm^{-3})	1.5 ± 0.0 (0.8-2.2, 16)	1.5 ± 0.0 (1.1-1.7, 8)	1.4 ± 0.0 (0.8-1.8, 21)	1.6 ± 0.0 (0.9-2.2, 19)
DBD to 30 cm depth (g cm^{-3})	1.3 ± 0.0 (0.8-1.9, 16)	1.4 ± 0.0 (1.1-1.7, 11)	1.2 ± 0.0 (0.8-1.4, 12)	1.3 ± 0.0 (0.9-1.9, 18)
SOC stock to 100 cm depth (kg C m^{-2})	51.5 ± 7.8 (30.4-64.3, 15)	54.0 ± 2.0 (50.6-57.5, 4)	51.6 ± 3.6 (46.0-58.5, 13)	48.7 ± 12.8 (30.4-64.3, 34)
SOC stock to 30 c m depth (kg C m^{-2})	16.6 ± 3.3 (10.4-26.5, 20)	15.7 ± 0.9 (14.3 – 17.2, 6)	15.9 ± 2.8 (12.3-21.4, 18)	18.3 ± 4.7 (10.4-26.5, 25)
SOC stock to 3 m depth (kg C m^{-2})	..	84.0	87.6	83.9
SOC stock 3 m cores (kg C m^{-2}) % change	..	8.9 (0-20 cm) 3.2 (280-300 cm) 64.5 %	10.9 (0-20 cm) 4.8 (280-300 cm) 56.0 %	19.2 (0-20 cm) 2.4 (260-280 cm) 87.6 %
SOC concentration to 100 cm depth (g C kg^{-1})	35.4 ± 11.3 (8.7-90.0, 32)	35.5 ± 0.3 (27.2-44.4, 7)	37.8 ± 8.3 (23.0-64.8, 22)	32.8 ± 17.4 (8.7-90.0, 53)
SOC concentration to 30 cm depth (g C kg^{-1})	44.9 ± 1.3 (16.4-90, 28)	37.5 ± 0.4 (34.6-44.4, 6)	47.3 ± 1.4 (35.4-64.8, 16)	50.0 ± 3.3 (16.4-90.0, 37)
Wet season extent (km^2)	30.77[b]	14.74	5.59	10.44
Carbon pool to 100 cm depth (MT ha^{-1})	1.583[b] (1.343-1.823)	0.796 (0.766-0.825)	0.289 (0.268-0.309)	0.509 (0.375-0.643)
Moisture content wet:dry105°C (%)	25.6 ± 10.7 (10.5-75.8, 42)	21.1 ± 4.8 (11.3-31.1, 23)	26.4 ± 5.1 (10.5-35.9, 19)	29.3 ± 16.1 (12.5-75.8, 55)

[a] average of three wetlands unless indicated, [b] sum of sites, [c] Hoogsteen et al. (2015)

Mean SOC concentration to 100 cm depth across the three wetlands was 35.4 g C kg^{-1} ± 11.3 (SD) and 35.5 g C kg^{-1} ± 0.3 (SD), 37.8 g C kg^{-1} ± 8.3 (SD), 32.8 g C kg^{-1} ± 17.4 (SD) for BP, TP and KL, respectively (Figure 3.6, Table 3.2). There was no statistically significant difference among wetlands for SOC concentration as determined by one-way ANOVA.

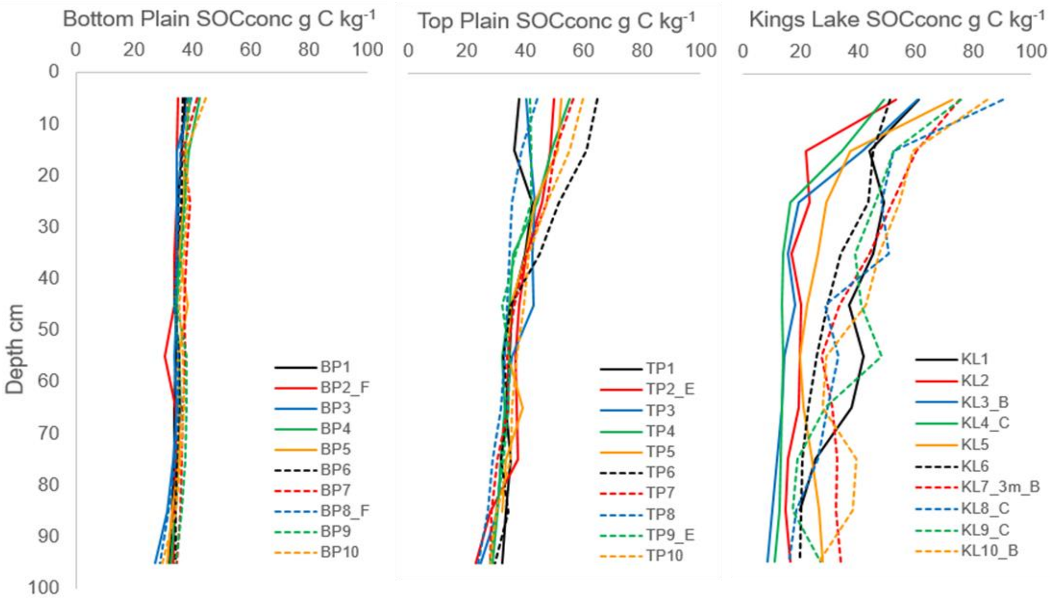


Figure 3.6 Soil organic carbon concentration (SOCconc) (g C kg^{-1}) at 10 cm increments to 100 cm depth of sediment cores from Bottom Plain (BP), Top Plain (TP) and Kings Lake (KL).

Table 3.3 Soil organic carbon (SOC) and soil moisture statistical analyses; significant results in **bold**.

Responses	Test	*P* value
SOC stock (to 100 cm depth) versus wetland	ANOVA	0.3348
SOC stock (to 30 cm depth) versus wetland	ANOVA	0.1379
SOC concentration (to 100 cm depth) versus wetland	ANOVA	0.1640
Log_{10}SOC concentration versus depth	Repeated measures ANOVA	**<0.0001**
Log_{10}SOC concentration versus wetland and depth	Repeated measures ANOVA	**<0.0001**
Soil moisture versus wetland	ANOVA+ Tukey	**0.0178**
Soil moisture versus wetland and depth	Repeated measures ANOVA	**0.0006**

Overall, SOC concentration declined with increasing soil depth across all locations. The greatest decline in SOC concentration occurred in the 0-50 cm range. The overall mean SOC concentration for each 10 cm increment from 0-50 cm (0-10 cm, 10-20 cm, 20-30 cm, 30-40 cm and 40-50 cm) differed significantly from the 90-100 cm increment (p<0.0001, p<0.0001, p<0.0001, p=0.008 and p=0.0352), respectively. There was a statistically significant difference among depths, and interaction between wetland and depth, for log_{10}SOC concentration as determined by repeated measures (GLM procedure) ANOVA (F(9,243) =74.72, p<0.0001), and (F(18,243) =17.65, and p<0.0001) respectively (Table 3.3).

Total C pool to 100 cm depth across the three wetlands was estimated to be 1.583 MT, with 0.796 MT, 0.289 MT, and 0.509 MT for BP, TP and KL, respectively, based on wet season areal extent of the study area (Table 3.2). Based on the SOC stock values in the 3 m cores, total C pool for the 3 wetlands can be extrapolated to an estimated total C pool of 2.604 MT, with 0.1.238 MT, 0.490 MT, and 0.876 MT for BP, TP and KL, respectively. DBD for each sample gradually increased with depth, with a mean across the 3 wetlands of 1.5 g cm^{-3} ± 0.0 (SD) (Figures 3.3, 3.7, Table 3.2).

Moisture content (wet:dry 105°C) ranged from 10.5 to 75.8 %, with a mean of 25.6 % ± 10.7 (SD) across the cores and to 100 cm depth (Figures 3.3, 3.7, Table 3.2). There was a statistically significant difference between wetlands for moisture as determined by one- way ANOVA ($F(2,27) = 4.79$, $p = 0.0178$) (Table 3.3). Post-hoc Tukey's test showed that KL and BP, differed significantly ($p = 0.0146$), however, KL and BP were not significantly different from TP. KL had significantly higher levels of moisture than BP. Moisture levels of surface sediments (0-30 cm) declined downstream through the three wetland areas, with high moisture levels of average 44 % around KL sediments, transitioning to an average 26 % at TP, and a uniformly low 16 % on BP. Moisture content declined to 20 % at a depth of 100 cm in the KL sediments, remained relatively constant in TP sediments, and increased to 26 % in the BP sediments (Figure 3.7). There was a significant difference in soil moisture when comparing the surface 30 cm and the bottom 30 cm amongst the wetlands as determined by one-way ANOVA ($F(1,58) = 13.23$, $p = 0.0006$) (Table 3.3).

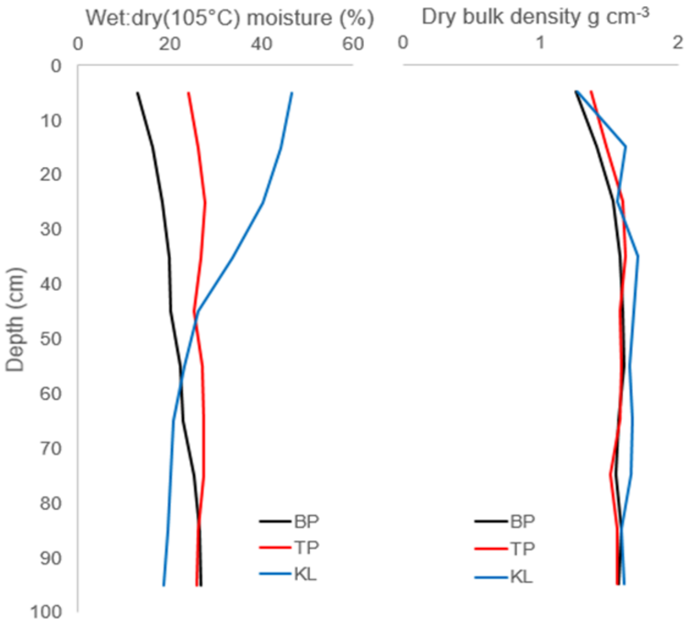


Figure 3.7. Mean soil moisture (wet:dry 105°C) content (%) and dry bulk density (DBD) (g cm^{-3}), at 10 cm increments to 100 cm depth across the three wetlands, Bottom Plain (BP), Top Plain (TP) and Kings Lake (KL).

3.4 Aquatic metabolism

For dry season samples, GPP varied 46-fold (0.2-9.3 mg C m^{-3} h^{-1}), and PR varied 5-fold (10.8-55.0 mg C m^{-3} h^{-1}), over the study area (Table 3.4). There was no statistically significant difference among the wetlands for GPP and PR as determined by one-way ANOVA ($F(2,23) = 2.95$, $p = 0.0720$ for $\log_{10}$GPP) and ($F(2,23) = 0.36$, $p = 0.7030$ for $\log_{10}$PR) (Table 3.5). GPP/PR ratio varied

16-fold (0.02-0.31), and was <1 for all samples, representing a range restricted to the heterotrophic spectrum (Figure 3.8). There was a statistically significant difference between wetlands for GPP/PR as determined by one-way ANOVA ($F(2,23) = 3.93$, $p = 0.0341$) (Table 3.5). Post-hoc Tukey's test showed that KL and TP differed significantly ($p = 0.04$), however, BP was not significantly different from TP or KL. Four samples with negative GPP values were excluded from data analysis (after Pace and Cole, 2000). Negative GPP values occurred where a sample's post-inundation dark bottle respiration reading was higher than its corresponding light bottle reading and was possibly due to organic content differences between the two bottles. Overall, there was a significant positive relationship between $\log_{10}$GPP and $\log_{10}$PR ($r = 0.62$, $p = 0.0007$, n=26), and were log-transformed to enable comparison with results from other Australian wetlands studies. When log-transformed, several $\log_{10}$GPP values were negative, and excluded from further analysis, hence the small number of samples for BP and TP.

Table 3.4. Dry and wet season GPP (mg C m^{-3} hr^{-1}), PR (mg C m^{-3} hr^{-1}), and GPP/PR ratio of Kings Plain study area, at three wetlands, Bottom Plain, Top Plain and Kings Lake. Mean ± standard deviation, with range and coefficient of variation (%) in parentheses. Negative GPP values excluded.

	Total	**Bottom Plain**	**Top Plain**	**Kings Lake**
Dry season	(n=26)	(n=9)	(n=7)	(n=10)
GPP	2.1 ± 0.4 (0.2-9.3, 95)	1.5 ± 0.2 (0.4-2.7, 48)	1.9 ± 1.0 (0.2-7.5, 135)	2.9 ± 0.8 (1.2-9.3, 81)
PR	19.1 ± 2.0 (10.8-55.0, 53)	16.3 ± 1.4 (11.3-23.8, 25)	20.4 ± 5.8 (11.3-55.0, 76)	20.6 ± 3.1 (10.8-37.8, 48)
GPP/PR	0.1 ± 0.0 (0.0-0.3, 58)	0.1 ± 0.0 (0.0-0.2, 50)	0.1 ± 0.0 (0.0-0.1, 61)	0.2 ± 0.0 (0.1-0.3, 47)
Wet season	(n=29)	(n=9)	(n=10)	(n=10)
GPP	5.1 ± 0.6 (0.3-15.0, 68)	3.5 ± 0.6 (1.0-5.7, 51)	5.7 ± 1.1 (0.3-9.1, 59)	6.1 ± 1.4 (1.4-15.0, 69)
PR	26.1 ± 2.0 (14.0-51.1, 42)	19.4 ± 0.9 (15.2-22.5, 14)	28.9 ± 4.0 (14.1-49.5, 43)	29.5 ± 3.7 (18.6-51.1, 39)
GPP/PR	0.2 ± 0.0 (0.0-0.5, 50)	0.2 ± 0.0 (0.1-0.3, 42)	0.2 ± 0.0 (0.0-0.5, 64)	0.2 ± 0.0 (0.1-0.3, 37)

For the wet season samples, GPP varied 48-fold (0.3-15.0 mg C m^{-3} hr^{-1}) and PR varied 4-fold (14.1-51.1 mg C m^{-3} hr^{-1}), over the whole study area (Table 3.4). There was no statistically significant difference in GPP and GPP/PR ratio between wetlands as determined by one-way ANOVA ($F(2,27) = 2.30$, $p = 0.1206$ for $\log_{10}$GPP) and ($F(2,27) = 0.13$, $p = 0.8752$ for GPP/PR) (Table 3.5). GPP/PR varied 22-fold (0.0-0.5 mg C m^{-3} hr^{-1}), and was <1 for all samples, representing a range restricted to the heterotrophic spectrum (Figure 3.8). There was a statistically significant difference in PR between wetlands as determined by one-way ANOVA ($F(2,27) = 3.41$, $p = 0.0477$ for $\log_{10}$PR) (Table 3.5).

Post-hoc Fisher LSD test showed that BP differed from KL significantly (p=0.02), and from TP significantly (p=0.04), however, KL was not significantly different from TP. One sample with a negative GPP value was excluded from data analysis (Pace and Cole, 2000). Overall, there was a significant positive relationship between $\log_{10}$GPP and $\log_{10}$PR (r=0.74, p=0.0001, n=29) and GPP/PR were log-transformed to enable comparison with results from other Australian wetlands studies. When log-transformed, several $\log_{10}$GPP values were negative, and excluded from further analysis.

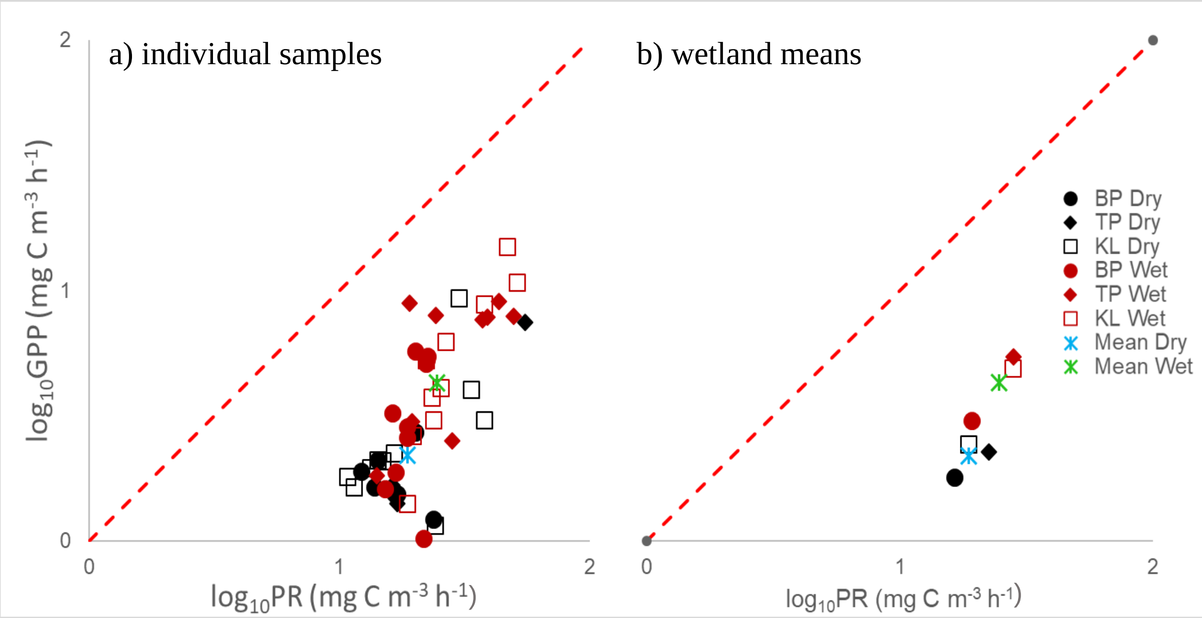


Figure 3.8 Relationship between GPP and PR of study area in dry and wet seasons, showing a) individual samples and b) wetland means. GPP/PR <1 for all samples (below red line), indicating a heterotrophic system for all samples, in both seasons. Bottom Plain (BP) (closed circles), Top Plain (TP) (closed diamonds), Kings Lake (KL) (open squares), dry season (black markers), wet season (red markers)), mean value of study area (✱ star) in blue (dry season) and green (wet season).

There was a statistically significant difference in GPP, PR and GPP/PR between seasons, across the wetland system, as determined by one-way ANOVA (F(1,53) =24.44, p<0.0001 for $\log_{10}$GPP), (F(1,54) =10.14, p=0.0024 for $\log_{10}$PR), and (F(1,54) =14.28, p=0.0004 for GPP/PR) (Table 3.5). Post-hoc Tukey's test showed that dry and wet season GPP, PR and GPP/PR differed significantly (p<0.0001, p=0.0002) and p=0.0004), respectively. Five samples with negative GPP values, and 4 samples with negative PR values, were excluded from data analysis (Pace and Cole, 2000).

There was a statistically significant difference in GPP between seasons for BP, TP and KL as determined by Kruskal-Wallis (H(1) =6.4, p=0.0114 for BP), (H(1) =6.4, p=0.0112 for TP), and (H(1) =4.97, p=0.0257 for KL) (Figure 3.8, Table 3.5). There was no statistically significant difference in PR between seasons for BP, TP and KL as determined by Kruskal-Wallis (H(1) =3.08, p=0.0792 for BP), (H(1) =3.44, p=0.0637 for TP) and (H(1) =3.29, p=0.0696 for KL) (Figure 3.8,

Table 3.5). There was a statistically significant difference in GPP/PR between seasons for both BP and TP as determined by Kruskal-Wallis ($H(1) = 4.86$, $p=0.0275$ for BP) and ($H(1) = 5.95$, $p=0.0147$ for TP) (Figure 3.8, Table 3.5). There was no statistically significant difference in GPP/PR between seasons for KL as determined by Kruskal-Wallis ($H(1) = 2.06$, $p=0.1509$ for KL) (Figure 3.8, Table 3.5). For the dry season 5-day inundation, mean relative humidity was 44.28 %, and mean temperature was 26.44°C. For the wet season 5-day inundation, mean relative humidity was 46.41 %, and mean temperature was 24.6°C.

Table 3.5 Summary table of GPP and PR statistics for dry and wet seasons, with significant results in **bold**.

Season	Responses	Test	P value
Dry	GPP dry versus wetland	ANOVA	0.2855
Dry	log_{10}GPP dry versus wetland	ANOVA	0.0720
Dry	PR dry versus wetland	ANOVA	0.6164
Dry	log_{10}PRdry versus wetland	ANOVA	0.7030
Dry	**GPP/PR dry versus wetland**	ANOVA + Tukey	**0.0341**
Wet	GPPwet versus wetland	ANOVA	0.1767
Wet	log_{10}GPPwet versus wetland	ANOVA	0.1206
Wet	PR wet versus wetland	ANOVA	0.0556
Wet	**log_{10}PR wet versus wetland**	ANOVA + Fisher LSD	**0.0477**
Wet	GPP/PR wet versus wetland	ANOVA	0.8752
Dry vs wet	**log_{10}GPP dry versus log_{10}GPP wet**	ANOVA + Tukey	**<0.0001**
Dry vs wet	**log_{10}PR dry versus log_{10}PR wet**	ANOVA + Tukey	**0.0024**
Dry vs wet	**GPP/PR dry versus GPP/PR wet**	ANOVA + Tukey	**0.0004**
Dry vs wet	**GPP BP dry vs wet**	Kruskal-Wallis	**0.0114**
Dry vs wet	**GPP TP dry vs wet**	Kruskal-Wallis	**0.0112**
Dry vs wet	**GPP KL dry vs wet**	**Kruskal-Wallis**	**0.0257**
Dry vs wet	PR BP dry vs wet	Kruskal-Wallis	0.0792
Dry vs wet	PR TP dry vs wet	Kruskal-Wallis	0.0637
Dry vs wet	PR KL dry vs wet	Kruskal-Wallis	0.0696
Dry vs wet	**GPP/PR BP dry vs wet**	Kruskal-Wallis	**0.0275**
Dry vs wet	**GPP/PR TP dry vs wet**	Kruskal-Wallis	**0.0147**
Dry vs wet	GPP/PR KL dry vs wet	Kruskal-Wallis	0.1509

4 Discussion

Carbon dynamics in the sediments and aquatic ecosystems of wetlands are shaped by geology and climate setting, geomorphology, hydrology, vegetation structure and residence time (Kayranli et al., 2010, Marín-Muñiz et al., 2014, Sutfin et al., 2016). However, to date there are no accurate global estimates of the tropical freshwater wetland carbon pool (Villa and Bernal, 2018). Within this wetland category, research has focused on humid tropical wetlands and less on wet-dry tropics wetlands. Within wet-dry tropics wetlands there is significant variation in SOC concentration and retention, dependent on wetland type, and sediment distribution and composition. For example, during the inundation phase, carbon may have longer residence time in depressional wetlands compared to more rapid export from flow-through wetland systems (Marín-Muñiz et al., 2014). Seasonal hydrology differences are the key driver in aquatic metabolism of wet-dry tropics wetlands, with the balance in aquatic metabolism dependent on the balance of processes occurring at that time (Pettit et al., 2017a).

4.1 Geomorphological context

The landscape setting of the Kings Plains study area, with alternating sequences of resistant bedrock and alluvium filled low-slope wide valleys (Figure 3.1), has parallels with African floodplain wetlands (Tooth et al., 2002). The Kings Plains longitudinal profile is a result of flow restriction due to abrupt protruding chert ridges, leading to an accumulation of sediment at KL's outlet, increased channel sinuosity at TP's downstream end, and fine sediment accumulation in BP's floodout zone. Cessation of source headwater flow from the ancestral Annan River through river capture some 1-3 million years ago (Heidecker, 1973), has produced the basins in which a low flow, depositional, wetland system now occurs (Best and Dallwitz, 1963). This low flow environment is also accentuated by the seasonality of wet-dry tropics rainfall, resulting in seasonally disconnected channels and recurrent sediment deposition.

The geomorphic unit and sediment composition of the three wetlands at Kings Plains reflect varied hydrogeomorphic characteristics (Tooth et al., 2002). The wetlands are seasonally connected, yet their sediment composition and SOC concentrations and stock vary across the system. In general, sediments fine downstream, reflecting an overall decline in slope. During the wet season, once water levels breach KL's outlet, very fine to coarse sediments are transported into the steeper, higher energy channel zone, and medium and coarse particles are deposited (Brierley and Fryirs, 2005). Stratigraphy in the channel zone indicates an alternating sequence of low flow regimes, with channel infill of waning flood deposits, and higher energy coarser deposition. Finer sediments continue to be transported downstream to the low relief TP and BP. Here, the TP floodplain light medium clays are vertically accreted by overbank deposition. Flow continues via the highly sinuous channel to the low relief BP. Following the cessation of rainfall post wet season, suspended load, heavy clay deposition occurs as low relief and declining energy produce distributary channels and a large floodout (Brierley and Fryirs, 2005, Tooth et al., 2002). Further evidence of the low energy slow depositional environment is the sinuosity of the palaeochannel, buried in the middle of BP.

The valley fill of the Kings Plains setting is representative of other zones found in the wet-dry tropics of northern Australia that are dominated by black cracking clays (Cook et al., 2010, Hutley et al., 2011). The dominant clay mineral montmorillonite found in BE1_F infers an elevated water-holding capacity from its high cation-exchange characteristics (Ahmad, 1983). The lack of stratigraphy on TP and BP demonstrates the reworking, or self-mulching, of cracking clays common in periodically wet environments (McKenzie et al., 2004, p. 364). Ferromanganese nodules found in Kings Plains sediments, along with mottling formed through the removal of iron, are indicative of restricted drainage and periodically saturated soil layers (McKenzie et al., 2004, p. 9).

At the downstream end of BP, water flows into a seasonally disconnected channel via a series of return channels. These return channels have the potential to be disturbed through knickpoint retreat. Changes in the geomorphic structure of BP from floodout to continuous channel due to knickpoint retreat would lead to wetland drainage and loss. This would convert BP from a sediment accumulating soil carbon sink to a carbon source through the fluvial export of carbon (Cowley et al., 2018, Sutfin et al., 2016, Villa and Bernal, 2018).

4.2 Soil carbon in wetlands

4.2.1 SOC concentration and stock

There is a paucity of research available on soil carbon storage in Australian wet-dry tropics freshwater wetlands. Comparison to the few studies undertaken on wet-dry tropics freshwater wetlands globally needs to be approached cautiously due to the diversity of the wetland types and their geographical range. Whilst there are estimates of SOC concentrations and stock to a depth of 1 m, most studies have only sampled to depths of between 30 and 80 cm. Therefore, to make comparison of Kings Plains to other studies, adjustments were made to account for the different sampling depths.

The Kings Plains mean SOC concentration of 35 g C kg^{-1} (to 100 cm depth) aligns with wet-dry topics wetlands in Costa Rica and Botswana (Bernal and Mitsch, 2013), and is lower than wetlands in the Mexican wet-dry tropics (Marín-Muñiz et al., 2014). The Mexican wetlands have higher SOC concentrations because of differences in their geomorphology, hydroperiodicity, and vegetation structure (Marín-Muñiz et al., 2014). At Kings Plains, SOC concentration declines through the soil profile to 1 m, however, there is variation amongst the three wetlands, with the greatest decline in the KL cores. During the wet season, inundation of wetlands, in general, drives carbon through the systems (Marín-Muñiz et al., 2014) to varying degrees, dependent on geomorphology. The lower energy environment of TP and BP supports the retention of carbon, and hence a slower decline in SOC concentration through the soil profile than in KL. Kings Plains SOC concentrations are low when compared to wetland studies from other climate zones, especially if those wetlands contain peat, such as NSW temperate highland peat swamps (Cowley et al., 2018), and humid tropics forested wetlands of Costa Rica (Bernal and Mitsch, 2013). SOC concentrations are also 6 times lower than temperate zone depressional wetlands found in Victoria (Pearse et al., 2018) (Table 1.1).

In comparison, the Kings Plains wetland system has considerable SOC stocks when compared to the few available wet-dry tropics wetland studies undertaken (Table 1.1). Total SOC stock to 1 m depth across the three wetlands is 1.58 MT. However, stock could be a lot greater if carbon is also being stored deeper in the profile, especially for BP and TP. Kings Plains SOC stocks are either similar in magnitude to the Mexican wetlands (Marín-Muñiz et al., 2014) or considerably higher than Costa Rican and Botswanan wetlands (Bernal and Mitsch, 2013), when adjusted for sampling depth. The SOC stocks of Kings Plains are comparable to those of temperate Australian wetlands (Cowley et al., 2018, Pearse et al., 2018) (Table 1.1). Within Kings Plains, spatial variability of SOC stocks is greater when comparing values to 30 cm depth rather than 100 cm depth (Figure 3.5), similar to Pearse et al. (2018) Variability may be due to greater wet-dry cycling at shallower soil depths compared to sub-surface soils (Clarkson et al., 2014) and differences in vegetation structure among the wetlands (Pearse et al., 2018). This SOC stock variability could also be explained by the greater variability among the wetlands in the dry season-measured soil moisture in the 0-30 cm range (Figure 3.7), and vegetation community differences, particularly between KL and its downstream plains. SOC stock and concentration are less variable at BP and more variable at KL, probably due to the different extremes of inundation, from driest to wettest, for the wetlands, indication of anaerobic conditions at KL, and more varied vegetation structure and geomorphology at KL compared to TP and BP. KL’s lower SOC stocks from 30-100 cm depth compared to TP and BP may also be due to more mixing of TP and BP sediments from seasonal shrink-swell in the soil column. There are implications for the control of SOC storage and release across the wetlands if future inundation patterns change. For example, if KL were to become drier, would its SOC concentration become less variable, and decline, and thus lead to lower SOC stocks? We expected higher SOC stock in KL, especially in cores KL4_C and KL8_C, due to the presence of Gley colours (Munsell Color, 2000) indicating anaerobic conditions. This may be due to the same local influences that are driving variability with depth.

The usual pattern of SOC stock decline in non-peat soils from 0-100 cm depth occurs at Kings Pains (Jobbágy and Jackson, 2000). However, surprisingly there is only a gradual decline from 100-250 cm in the 3 m cores from TP and BP, suggesting that these soils retain their carbon levels at depth. However, caution is advised due to a small number (3) of 3 m cores taken. For cores to 75 cm, Pearse et al. (2018) found that the decrease in SOC stocks through the profile is lower in wetland soils, compared to terrestrial soils, due to slow decomposition under saturated conditions. The strong adsorption of organics to clay particles compared to sandy particles is a factor in the stability of carbon within the soil in tropical regions (Hassink, 1997). Additionally, cracking clays allow direct organic matter input to deeper soil layers during the dry season, which decompose under saturated conditions once swollen (DES, 2013). The extent to which this decomposed organic matter adds to the SOC concentration at depth is not well documented. Further investigation would determine if the higher than expected levels of carbon at depth are more widespread across TP and BP and would aid understanding of the volume of total carbon pool in this wetland type.

4.2.2 The effect of dry bulk density

DBD has a strong effect on the levels of SOC stock derived from SOC concentrations (Köchy et al., 2015). For example, the high SOC stock values of peat-rich soils are characterised by their organic matter rich concentrations and low bulk densities (Köchy et al., 2015). In contrast, Kings Plains' high SOC stock is largely driven by the elevated DBD levels in the clay-rich wetland sediment, resulting in higher SOC stocks compared to other wet-dry tropics wetlands, despite similar SOC concentrations (Table 1.1). Kings Plains' high mean DBD of 1.5 g cm^{-3} is between 1.5 and 5 times higher than the average DBDs of wet-dry tropics wetlands reported in Table 1.1, and shows the usual increase in DBD with increasing depth (McKenzie et al., 2004, p. 22). This aligns with the DBD of floodplain silty clay sediments of the nearby Annan River (Kelly, 1997). In Mexican wetlands (Marín-Muñiz et al., 2014), mean SOC concentrations twice that of Kings Plains results in slightly lower mean SOC stocks due to very low mean DBD, even when adjusted for sampling depth. Kings Plains SOC stock is comparable to other Australian temperate wetlands, but again this is due to their higher SOC concentrations offset by generally lower DBDs (Cowley et al., 2018, Pearse et al., 2018). The influence of DBD on SOC concentration in determining SOC stock underscores the importance of accurate bulk density calculations when estimating larger scale carbon pools (Köchy et al., 2015). For wet-dry tropics wetlands, detrital SOC storage (small particles) is critically important, as seen at Kings Plains, rather than macro carbon (plant matter) as found in peatlands.

Within Kings Plains, the highest SOC stock is found on BP, and the lowest on KL, which is unexpected. Despite the lowest SOC concentration across the three wetlands being found on BP, it's high DBD values boosts BP SOC stocks. The study area's high DBD is a direct result of cohesive clay dominated sediments, especially the cracking clays found on TP and BP. DBD normally range between 1.0-1.8 g cm^{-3}, however, soils with abundant organic matter can have DBDs less than 0.5 g cm^{-3} (McKenzie et al., 2004, p. 22). Tropical peat soils have average bulk densities of 0.09 g cm^{-3} compared to boreal peatlands of 0.112 g cm^{-3} (Köchy et al., 2015). Generally in soils where DBD is >1.6 g cm^{-3}, and specifically for clay soils >1.4 g cm^{-3}, poor aeration and physical resistance inhibits root growth (McKenzie et al., 2004, pp. 47-48, Weil and Brady, 2016, pp. 161-166). This is evidenced with Kings Plains soils where the minimal occurrence of fine roots disappears by ~70 cm depth.

4.2.3 Relationship between geomorphology, sedimentology and SOC

The role of geomorphology and hydrology in determining SOC concentration and stock is evident at Kings Plains. Despite the lowest SOC concentrations, the highest mean SOC stock to 1 m is found on BP, due to the highest mean bulk density from the deposition of uniform, cohesive, very fine, heavy clay sediments. The presence of cracking clays on TP and BP would allow the soil to remain saturated for extended periods, potentially resulting in higher SOC concentrations at depth. Soil moisture on BP declines most slowly due to the physicochemical structure of the heavy clay. Despite the relatively dry surface layer, BP holds more moisture than KL in the 70 -100 cm depth range. Both SOC stock and SOC concentration within KL is more variable, due to greater geomorphological complexity compared to BP and TP (Sutfin et al., 2016). The northern edge of the lake abuts the valley margin. The southern edge is located midway across the valley, with ~200 m distance subject to wet-dry season lake recession and expansion, meaning it is subject to a larger area of seasonal hydrological change. Three samples (KL2, KL3_B, and KL4_C) were taken from the southern edge, the remainder from the northern edge. One sample (KL10_B) from the northern edge was taken near a backswamp. SOC stock and concentration results from KL's northern edge are generally higher than those from the lake's southern edge (Figure 3.5). In this location of KL, sediments dry out as the lake recedes during the dry season, and combined with silt and sandy clays that hold less moisture, contribute to the faster decline in soil moisture for KL compared to BP and TP. High moisture content delays vegetation decay, preserving carbon longer (Bernal and Mitsch, 2012, Sutfin et al., 2016), with the highest moisture content found in the northern edge locations.

The relationship between sedimentology and SOC is clear. The variation in SOC concentration is a function of the different geomorphological drivers affecting each wetland (Aufdenkampe et al., 2011, Wohl et al., 2017). In the slow vertically accreting, uniform heavy clays across BP, SOC values show little variability between sampling locations. In contrast, at KL where there is variation in sediment classes, SOC is also more variable across sampling locations and through the profile. KL is a more complex sampling environment, due to backswamps, proximity to input from the northern hillslope, and an extensive gradual drainage area on the southern lake edge. The sediments vary, with a higher proportion of sandy clays, which store less carbon than the fine clays of TP and BP.

4.2.4 Residence time of SOC

Little is known about the composition and structure of the deeper strata of the Kings Plains alluvial fill, except for information from drill holes drilled at the eastern end of the study area (Best and Dallwitz, 1963). However, Bik and Lucas (1963) and Heidecker (1973) estimate that the alluvial fill is tens of metres deep (maybe up to 50 m) and has been in place since at least the late Pleistocene when the ancestral Annan River changed course.

This raises interesting questions about the extent and residence time of soil carbon at depth. The processes by which carbon perseveres in soils at depth is not fully understood (Bernal et al., 2016). A range of biophysicochemical interactions are implicated, including limitations of microbial activity on organic matter (Kuzyakov, 2010), rising temperature (Hagerty et al., 2014), and physicochemical interactions with clays (Schmidt et al., 2011). Disturbance of deeper (to 3 m) stores of carbon in a sub-tropical sandy soil can be activated by the response of microorganisms to ‘priming’ by inputs, such as organic nitrogen (Bernal et al., 2016). Whether this priming would occur in BP and TP where carbon is strongly bonded to clay is not understood. At KL, carbon would not be as strongly bonded to the coarser clay sediments (McKenzie et al., 2004). Additionally, the vegetation structure of KL’s aquatic plants and woody riparian zone allows a deeper root profile, exposing the deeper sediments to bioturbation and increased diffusion of oxygen (Bernal et al., 2016). These are important considerations when estimating the total carbon pool. The estimate of mean 1.58 MT ± 0.24 (SD) of carbon stock at Kings Plains could therefore be considered a minimum.

4.2.5 Measurement methods and research comparison

Soil carbon measurement can be problematic due to spatial variability of carbon, non-standardised analytical methods, inclusion of SOM in SOC analysis, the use of SOM:SOC conversion factors, and LOI accuracy issues:

a) natural spatial variability of carbon occurring in soils, even within localised areas, renders spatial or temporal comparison difficult (Allen et al., 2010). This requires sampling replication to adequately analyse carbon stocks. In this context, LOI is a suitable cost-effective method for larger sample quantities (Pallasser et al., 2013);

b) non-standardised and potentially conflicting analytical methods confound results (Allen et al., 2010). Soil carbon analysis methods are diverse, ranging from simple to complex, including destructive *in situ* (physical sampling) or non-destructive remote measures (Allen et al., 2010), and providing precise quantitative to broad qualitative results;

c) soil analysis is restricted to the SOC component of SOM (Rayment and Lyons, 2011). The occasional unavoidable inclusion of SOM in analysis (Conyers et al., 2011) can render SOC measurements unreliable. By convention, 58 % of SOM is considered SOC, however, this is not universally accepted due to soil type differences (Pribyl, 2010), which makes study comparison problematic. More recent studies utilise elemental analysis of representative samples to determine the appropriate conversion factor for a particular soil type; and

d) LOI accuracy issues can occur due to differing individual sample weight, furnace temperature, firing durations, structural water loss from clays at high temperatures, and in-furnace sample re-positioning. However, there are techniques to overcome these issues (Hoogsteen et al., 2015). Clay correction factors, used in this study, can compensate for overestimation of SOC in clay-rich soils due to structural water loss (Hoogsteen et al., 2015).

Despite these issues LOI was considered the most appropriate SOC analysis method to use for this project due to the large number of samples, precision required, use of LOI correction factors and laboratory facilities available. Differences in SOC calculation methods and wetland definitions, for example, peatlands categorised as soils rather than wetlands (Junk et al., 2014), lead to uncertainties in global estimates and modelling.

4.2.6 Limitations of soil carbon analysis

Direct comparison to other soil carbon studies should be approached with caution, due to differences in geomorphological, hydrological, vegetation and soil structures of other study areas. The lack of comparable data demonstrates the need for further research of Australian wet-dry tropics wetlands.

4.3 Aquatic metabolism

Aquatic metabolism of wetlands varies temporally due to seasonal hydrology differences, depending on the balance of processes occurring at that time. Wetlands can display significant spatial and temporal variation in the balance between autotrophy and heterotrophy, even in wetlands with similar characteristics (Beringer et al., 2013, Bortolotti et al., 2016, Hoellein et al., 2013). Taking one set of measurements only provides a snapshot. Nevertheless, if GPP is greater than PR, i.e. GPP/PR >1, the system is currently autotrophic, and the wetland aquatic ecosystem functions as a carbon sink. Where GPP/PR ratio is <1, the system is heterotrophic and functions as a carbon source (Kayranli et al., 2010). Despite the variations in GPP, PR and GPP/PR within and between wetlands in the Kings Plain wetland system, all three wetlands were determined to be heterotrophic when inundated under controlled conditions based on samples taken during both the dry and wet seasons.

There is the possibility of significant spatial heterogeneity in aquatic metabolism, and carbon dynamics due to varying inundation regimes(Cardoso et al., 2012), between the three Kings Plains wetlands, as during the dry season, the wetlands become separate, hydrologically disconnected systems. Post-wet season, as water recedes, wetlands can reform as diverse individual systems. The degree of hydrological connectivity differences between water bodies can produce variation in phytoplankton abundance. Differences in GPP/PR may relate to variations in inundation duration each year, which is in part determined by each wetland's geomorphology. Due to its geomorphic setting, KL is inundated for more months each year compared to BP and TP. The significant difference between dry season GPP/PR of KL and TP may be explained by anecdotal evidence that TP dries out earlier than BP (Hughes, 2017). Whilst Kings Plains' wet season aquatic productivity was greater than the dry season, overall levels were still relatively low, and remained heterotrophic. These results are at the low end of the productivity range of other wet-dry tropics wetland systems within northern Australia (Butler, 2008), and globally (Hamilton et al., 1995, Lindholm et al., 2007) (Table 1.2).

In northern Australian wet-dry tropics wetlands, the low overall productivity levels are likely due to several factors (Butler, 2008). A key factor is turbidity from suspended clay particles in a phytoplankton dominated system (Butler, 2008, Ward et al., 2013). A dominance of clays in suspension, as evidenced at Kings Plains, limits light penetration into the water column restricting aquatic productivity. However, the extent of this effect is dependent on particle size distribution, chemical composition, and shape of the suspended solids (Bilotta and Brazier, 2008). Wetland turbidity is also increased when sediments are disturbed by the foraging of feral animals (Doupé et al., 2010). Additionally, nutrient deficiency in these waters can also limit productivity levels (Butler, 2008, Pettit et al., 2017b).

Sub-tropical wetlands can exhibit similar responses in carbon cycling and transformation as the wet-dry tropics. Sub-tropical wetlands are also characterised by seasonal wet-dry periods (Peel et al., 2007), and can display higher levels of productivity, but still with a dominance of heterotrophy (Hagerthey et al., 2010). However, the factors impacting productivity differ for wet-dry tropics compared to these systems. In sub-tropical wetlands, primary productivity can be suppressed through shading induced by the presence of dense floating aquatic plants (Hagerthey et al., 2010) rather than turbidity in the water column. A conceptual model in Figure 4.1 shows how these systems vary in terms of GPP and PR.

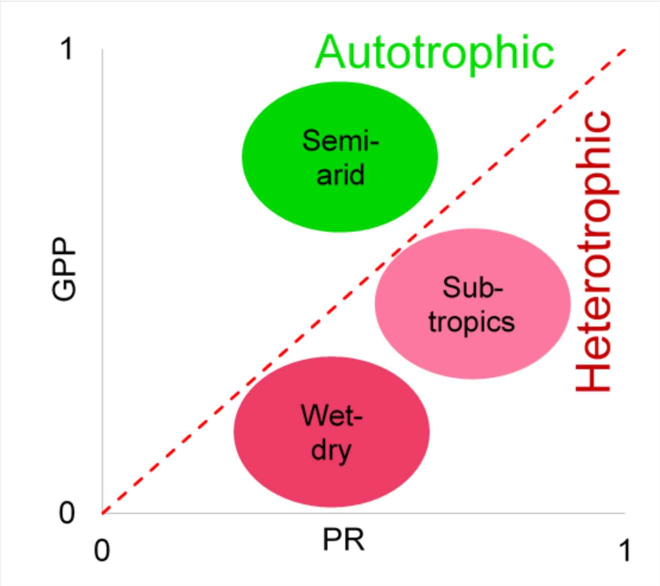


Figure 4.1. Conceptual model of variations in gross primary productivity (GPP) and planktonic metabolism (PR) for seasonal wet-dry, semi-arid inland, and sub-tropical wetlands.

The strongly seasonal hydrological pulse of wet-dry tropics wetlands such as Kings Plains contrasts markedly with the highly variable and intermittent hydrological regimes of semi-arid inland floodplain wetland systems (BOM, 2018). Previous research shows that intermittent hydrological regimes impact the timing of planktonic metabolism, a key component of aquatic carbon cycling and transformation (Kobayashi et al., 2013). Carbon cycling and transformation has been found to be highly variable in channel and non-channel floodplain environments, in semi-arid inland systems, although usually these systems have a dominance of autotrophic conditions (Kobayashi et al., 2013) (Figure 4.1).

Wet-dry tropics wetlands such as Kings Plains possibly have a naturally low productivity baseline compared to other wetland types, perhaps due to the muted response of phytoplankton, zooplankton and other organisms that are adapted to seasonal inundation, and also due to turbidity from suspended clays which can limit light penetration into the water column, thereby limiting photosynthesis at depth. Regular seasonal inundation of Kings Plain possibly elicits a lower productivity response compared to the ‘boom-bust’ productivity pattern of semi-arid inland wetlands systems (Arthington and Balcombe, 2011, Sheldon et al., 2010), where inundation may only occur inter-annually or decadally. Animal activity (e.g. kangaroo, cattle and feral pig grazing) affects water quality conditions and in turn GPP and PR (Kobayashi et al., 2009), while a suite of anthropogenic activities also affects water quality and wetland conditions leading to changes in aquatic metabolism.

Whilst aquatic consumers are largely supported by algal carbon, nutrients from sediments such as nitrogen and phosphorous also support aquatic primary production (Fellman et al., 2013). However, the relationship between sediment nutrients and aquatic productivity of Kings Plains wetlands has not been explored in this study. The low nitrogen environments of many Australian wet-dry tropics wetlands may be also be supported by terrestrial organic matter inputs. Whilst a relatively small source, these terrestrial inputs are considered as potentially locally important in the stimulation of algal productivity, particularly in disconnected wetlands (Pettit et al., 2017a). The relationship between sediments, inputs of terrestrial organic matter and aquatic productivity could form the basis for further study.

4.3.1 Limitations of aquatic metabolism analysis

Other studies have used a variety of different methodologies for assessing metabolism both *in situ* and laboratory-based, leading to varying results. For this study, only part of the carbon production was measured, not including, for example, production due to epiphytic algae on macrophytes (Pettit et al., 2016), or dry season benthic algae production in the littoral zone (Bunn et al., 2006). This could impact whether the system was heterotrophic or autotrophic. Removal of sediments and inundation with DI water would also impact production measures due to DI water's lack of N and P. Identifying the biological sources of carbon was also outside the scope of this study.

4.4 Conservation of wet-dry tropics floodplain wetlands

Australia has some of the last relatively undisturbed wet-dry tropics floodplain wetlands in the world (Pettit et al., 2017a). Despite recognition of their importance, significant knowledge gaps of this extensive, complex and variable wetland class exist (Pettit et al., 2017a, Warfe et al., 2011). There is a paucity of literature on soil carbon dynamics and aquatic productivity (Beringer et al., 2013). On the global scale, estimates of the soil carbon pool of tropical freshwater wetlands are considered inaccurate (Villa and Bernal, 2018). Understanding the drivers of ecosystem function (Warfe et al., 2011) and soil carbon storage in these relatively unmodified systems, is an important baseline for their conservation. This understanding is needed for both national and global conservation efforts, providing insight for assessment and management of changes associated with future economic development and climate impacts.

4.4.1 Economic development

Appropriation of wetlands for economic development disturbs their hydrogeomorphic balance and alters their biophysical characteristics, including carbon cycling. This potentially reduces the soil carbon pool and increases carbon gaseous fluxes. There is renewed interest in the promotion of irrigated agricultural development in northern Australia (Ash et al., 2017). Water extraction for economic development is highly likely to impact the complex wetland systems associated with Australia's northern rivers (Ward et al., 2013). Wetland drainage and carbon release will change the distinctive ecological character of these rivers. Page and Dalal (2011) estimate that ~25 % of SOC in the top 1 m of wetland soils drained for economic development will be lost within the first 50 years of drainage. The conversion of forested freshwater wetlands to flooded grasslands for cattle grazing in a Mexican wet-dry tropics zone was found to cause SOC loss and increased CO_2 and CH_4 fluxes, within 20 years of conversion, especially during the dry season (Hernandez et al., 2015). Livestock grazing of wetlands on the Tibetan plateau has reduced primary productivity and enhanced CH_4 emissions (Hirota et al., 2005). In Australia's wet-dry tropics, the most productive areas for economic development, especially grazing, include regions with cracking clay soils (Cook et al., 2010). Soil carbon losses occur as a result of soil structure degradation and erosion, and modification of vegetation cover from cattle grazing and feral animal foraging (Baker et al., 2000). Increases in GHG production through loss of the SOC pool and livestock emissions (Baker et al., 2000) present issues for GHG management. The impact of exotic species contributes to turbidity and alters aquatic metabolism, however, this effect is dwarfed by the impact of natural seasonal hydrological variability (Doupé et al., 2010).

4.4.2 Climate change impacts

Climate change impacts on Australia's wet-dry tropics zone are expected to significantly change hydrological regimes, particularly floodplain and wetland inundation. Increased rainfall predicted under various future climate scenarios for northern Australia (IPCC, 2014) may be offset by increased evapotranspiration rates under warmer temperatures (Hamilton, 2010). The relatively cloud-free dry season environment means that increased temperatures will raise water temperatures in shallow wetland environments. Both future hydrological and temperature patterns will influence the soil carbon pool, and also ecological structure and aquatic productivity, however, the biogeochemical interactions are complex (Hamilton, 2010). Higher temperatures tend to result in increased heterotrophic activity. The effect of this in the nutrient-limited algal-dominated wet-dry tropics is not considered as important compared to the vascular plant-dominated humid tropics, where temperature effects will have a greater negative impact on aquatic productivity (Hamilton, 2010). More broadly, and outside the scope of this study, soil moisture content was found to be the primary environmental factor controlling GHG emissions from a wet-dry tropics wetland in northern Australia (Beringer et al., 2013). Increasing temperatures and longer dry periods, albeit with potentially less frequent but more intense rainfall and cyclone activity, may lead to wetland desiccation and erosion, causing a change from carbon sink to source (Finlayson et al., 2013, Larkin et al., 2016).

4.4.3 Wetland conservation

Economic and climate change pressures on wetland systems underline the importance of preserving those that remain in their undisturbed and unmodified form. For the persistence of SOC within a wetland, environmental and biological controls dominate over the molecular structure of soils (Schmidt et al., 2011). On King Plains, this is illustrated by potential geomorphic changes at the downstream end of the wetland system. Prevention of knickpoint retreat to maintain BP as an intact wetland would preserve its SOC stores (Cowley et al., 2018). Old, slow cycling SOC stores in undisturbed soils are vulnerable to disturbance. Increased soil aeration from alteration of the hydrological regime can lead to increased enzyme activity, and decomposition of recalcitrant SOC (Bernal et al., 2016, Villa and Bernal, 2018). Disturbance or cultivation can increase root density, breaking soil aggregates, decrease bulk density, and increase soil oxygen diffusion rates, leading to SOC loss and gaseous emissions (Lal, 2008). The wet-dry tropics are dependent on seasonal flood pulses for river-floodplain-wetland connectivity and ecosystem functioning (Irvine et al., 2016, Junk et al., 2013). Changes to the hydrological regime will affect aquatic carbon transformation and cycling, and the balance between heterotrophy and autotrophy.

5 Conclusion

This study provides an understanding of the soil carbon and aquatic productivity characteristics of a relatively unmodified wet-dry tropics freshwater wetland system. The wetlands of Kings Plains are diverse and display variability in their geomorphic, sedimentological and hydrological characteristics, and vegetation communities. These factors contribute to the considerable, but spatially variable, soil carbon stores and low aquatic metabolism responses measured in this study.

For soil carbon, hydrogeomorphology drives sediment composition and distribution, resulting in a considerable carbon pool in the low energy clay-rich sediment accumulation environments of the wetlands. Bulk density is a key factor. The highest soil organic carbon stocks occur where sediments are the finest. This underscores the importance of bulk density when estimating larger scale carbon pools. Relatively undisturbed alluvial fill wet-dry tropics wetlands like Kings Plains hold 51.5 kg C m^{-2} of carbon stock with the possibility of large carbon pools at depth. Further study of soil carbon stores is needed to accurately quantify the total carbon pool, for this and other equivalent wetland systems.

Aquatic productivity was low in both the wet and dry season and all three wetlands were heterotrophic. The importance of planktonic metabolism in the aquatic ecosystems of wet-dry tropics wetlands in providing carbon storage is established. However, to determine the extent to which carbon cycling and transformation are unevenly distributed both spatially and temporally in the Kings Plains wetland system requires a more comprehensive study of ecosystem processes over time. In this study, carbon measurements only included soil carbon within sediments and carbon associated with aquatic metabolism, and did not aim to measure carbon flux, i.e. CH_4 and CO_2 emissions. Naturally, the inclusion of other carbon cycle components is critical for understanding carbon dynamics of wet-dry tropics wetlands and could form an extension to this study.

Land use changes through economic exploitation of northern Australian wet-dry tropics wetlands for agricultural development will potentially release carbon through irrigation-induced altered hydrological regimes, fire regime modification, and changed ecosystem functioning. This study’s findings contribute to this growing body of research and provide a baseline from which further analyses can be undertaken and be used in effective wetland monitoring and management at the regional scale. Of key importance is prevention of wetland deterioration and preservation of their significant carbon storage potential as part of conservation efforts in northern Australia.

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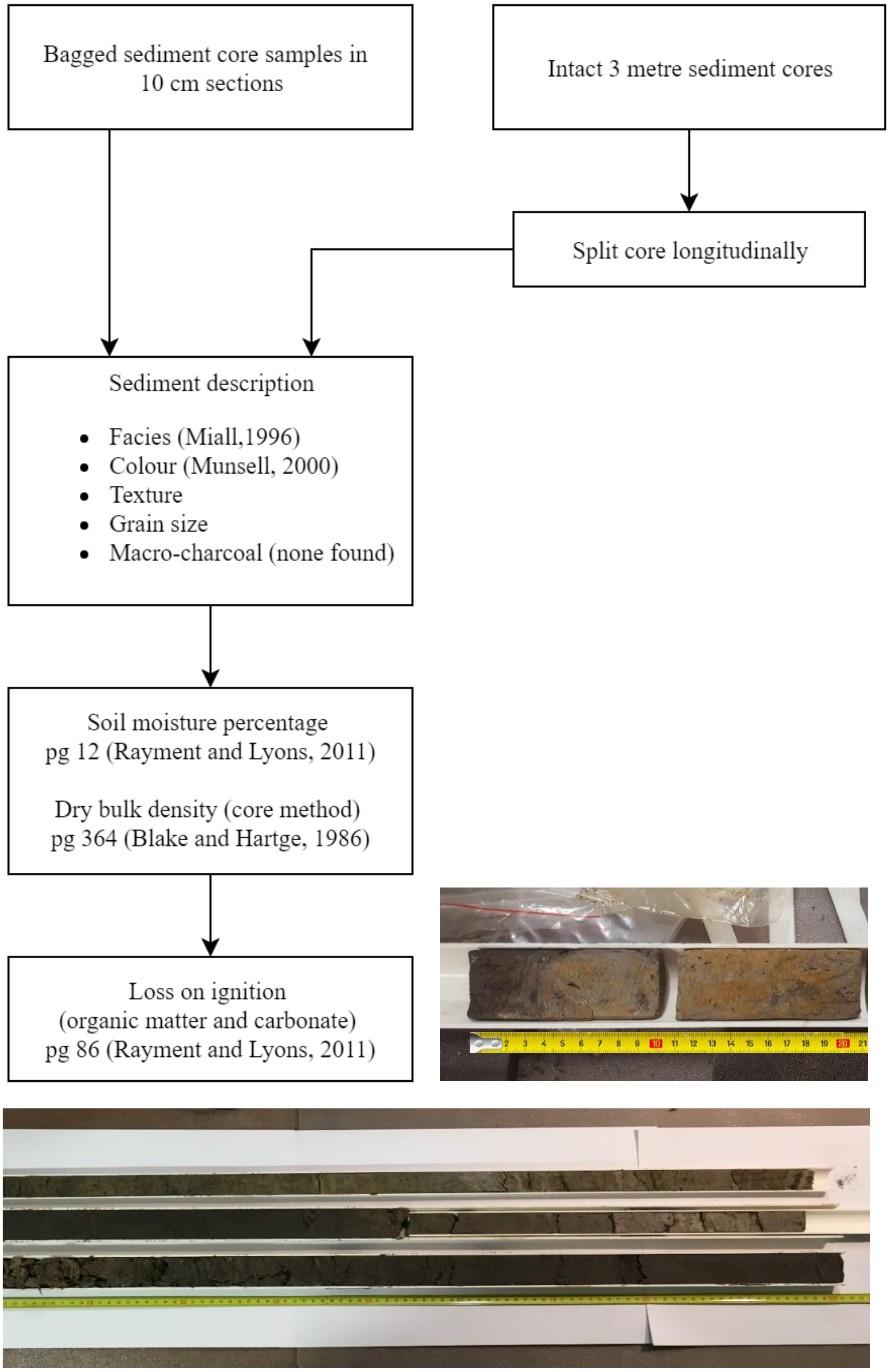
6 Appendices



Appendix 1. Flowchart demonstrating laboratory processing steps used for sediment core characterisation, soil moisture, dry bulk density and loss on ignition, and images showing 10 cm section split samples (top) and split 3 metre sediment core (bottom).

Appendix 2. Formulae used

2.1 Loss on ignition (LOI)

LOI represents the soil organic matter (SOM) % of a soil sample i.e. mass oxidised and volatized. LOI or SOM % includes carbon, hydrogen, nitrogen, oxygen etc.

$$\text{LOI (\%)} = \frac{\text{(dry mass before combustion (g) – dry mass after combustion (g)}}{\text{dry mass before combustion (g)}} \times 100$$

(Fourqurean et al., 2014)

2.2 Soil organic carbon (SOC)

SOC represents the organic carbon portion of a sample's SOM using an appropriate conversion factor from scientific literature. For this thesis, the conversion factor of 0.50 recommended by Pribyl (2010) was used.

SOC (%) = SOM % x 0.50 conversion factor

(Pribyl, 2010)

2.3 Dry bulk density (DBD)

DBD is the mass of dry soil relative to original sampled volume (i.e. after moisture removed) for a given sample. For this thesis, the sample was collected in a PVC tube, and soil was dried at 105°C for 2 hours.

$$\text{DBD (g cm}^{-3}\text{)} = \frac{\text{dry soil mass of sample (g)}}{\text{original soil sample volume (cm}^3\text{)}}$$

where:

Original soil sample volume (cm^3) = tube radius^2 (cm^2) x π x tube length (cm)

Dry soil mass (g) = soil dried at 105°C for 2 hours

(Fourqurean et al., 2014)

2.4 Soil organic carbon (SOC) density

SOC density represents the density of carbon within a given sample analysed

$$\text{SOC density (g cm}^{-3}\text{)} = \text{DBD (g cm}^{-3}\text{)} \times \frac{\text{SOC \%}}{100}$$

(Pearse et al., 2018)

Appendix 2. (continued) Formulae used

2.5 Soil organic carbon (SOC) concentration

SOC concentration is the concentration of soil organic carbon in g C kg^{-1} of a given sample.

SOC concentration (g C kg^{-1}) = SOC % x 10

(Bernal and Mitsch, 2008)

2.6 Soil organic carbon (SOC) stock

SOC stock is the conversion of SOC density, to a volumetric measure, i.e. mass of carbon contained in a given sample.

SOC stock (g C m^{-2}) = SOC density (g C cm-3) x T (cm)

where:
T = sample tube length (sample thickness) in cm of a given sample

To enable comparison with other studies, convert g C m^{-2} to kg C m^{-2}:

SOC stock in Kg C m^{-2} = g C m^{-2} * 0.001

(Pearse et al., 2018)

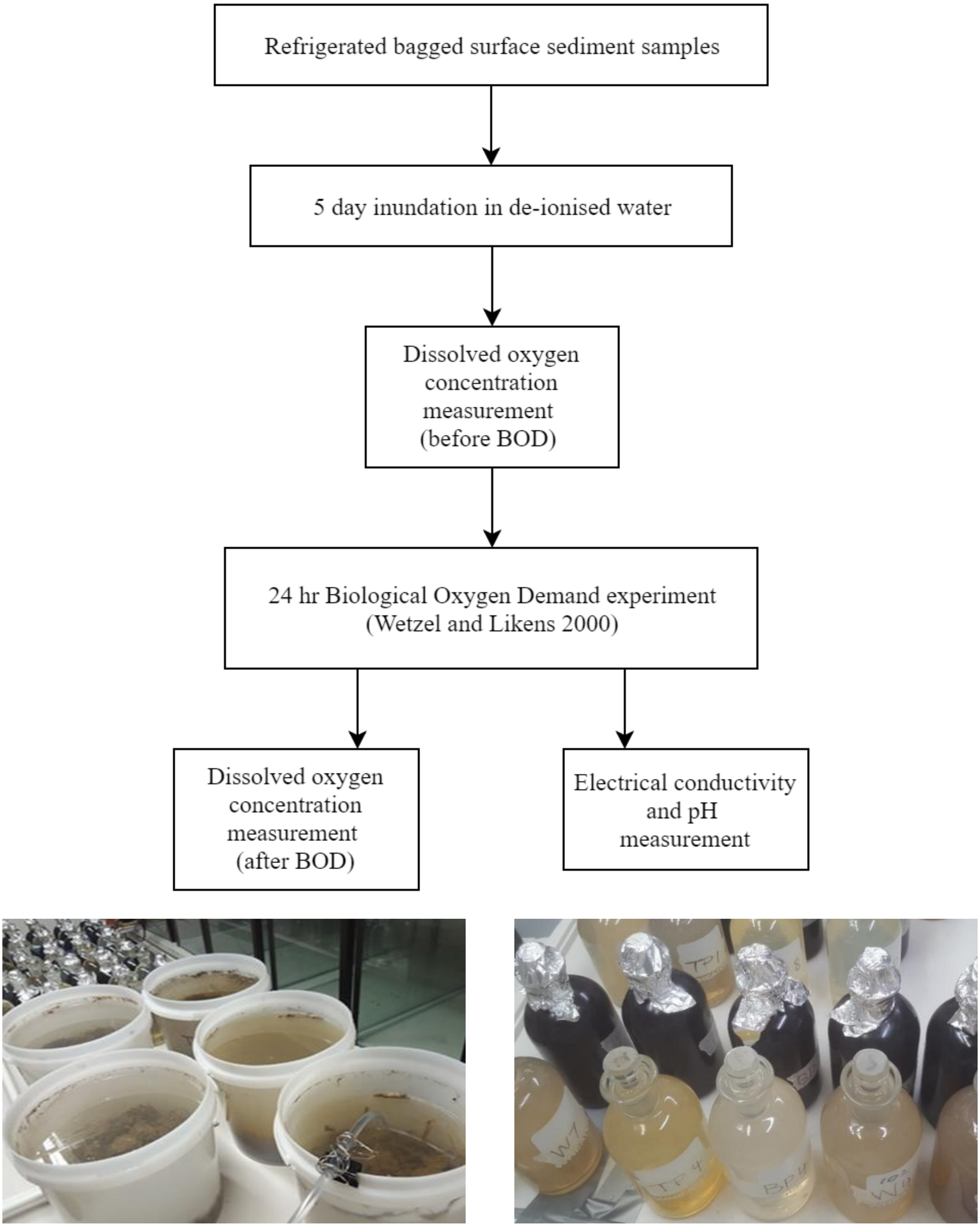
2.7 Gross primary productivity (GPP) and planktonic respiration (PR)

GPP and PR represent the conversion of dissolved oxygen to carbon. Carbon production for each sample can be calculated using the following formulas:

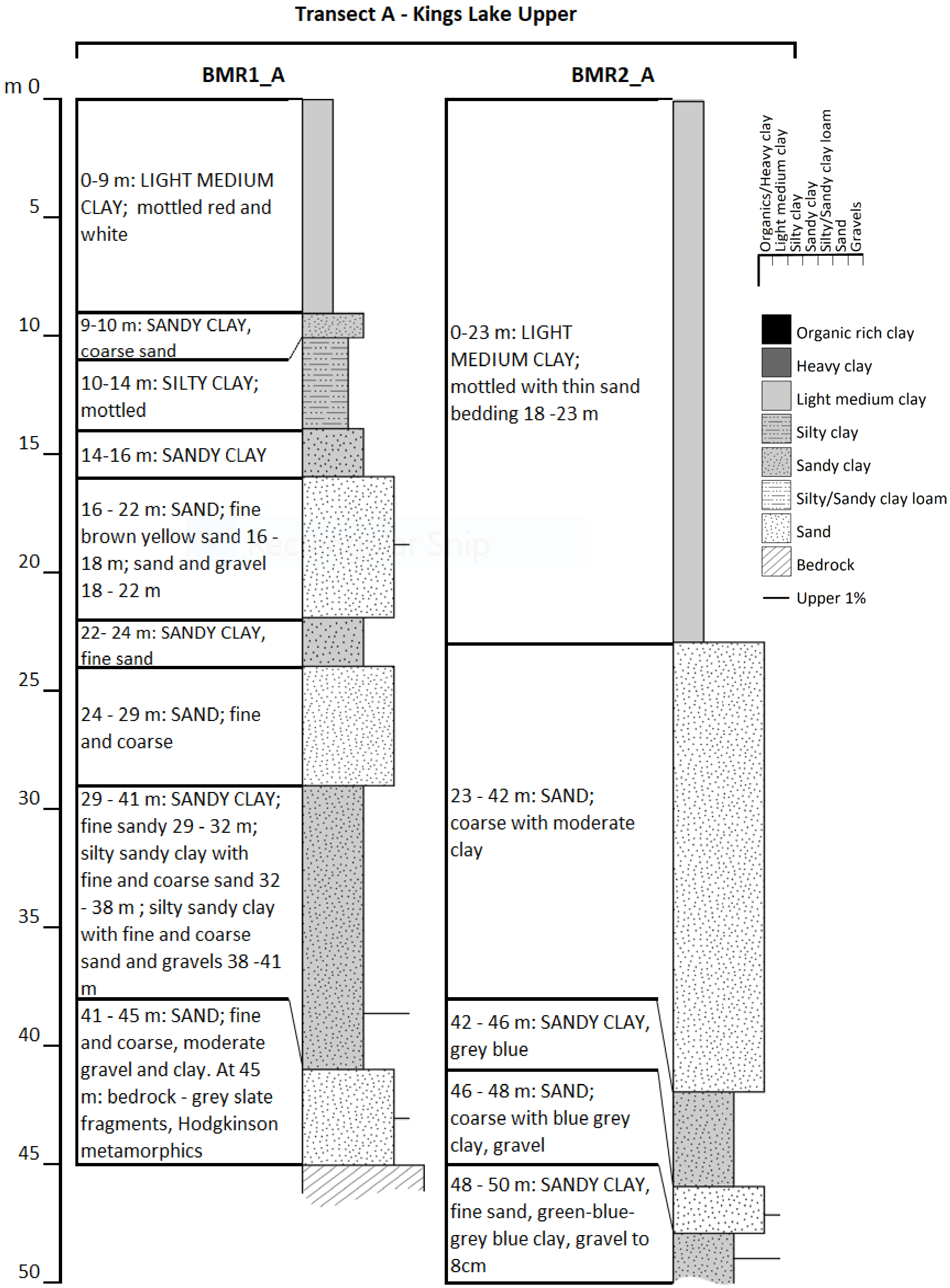
GPP = [(LB – DB) x 1000 x 0.375]/(PQ x t), and
PR = [(IB – DB) x 1000 x RQ x 0.375]/t

where:
IB = DO concentration at beginning (mg L^{-1})
LB = DO concentration in light bottle at end (mg L^{-1})
DB = DO concentration in dark bottle at end (mg L^{-1})
0.375 = factor = ratio of moles carbon: moles oxygen, converts mass of oxygen to mass of carbon
PQ = 1.2 = photosynthetic quotient i.e. relative amounts of oxygen and carbon in photosynthesis
RQ = 1.0 = respiratory quotient i.e. relative amounts of oxygen and carbon in respiration
t = time (hours)

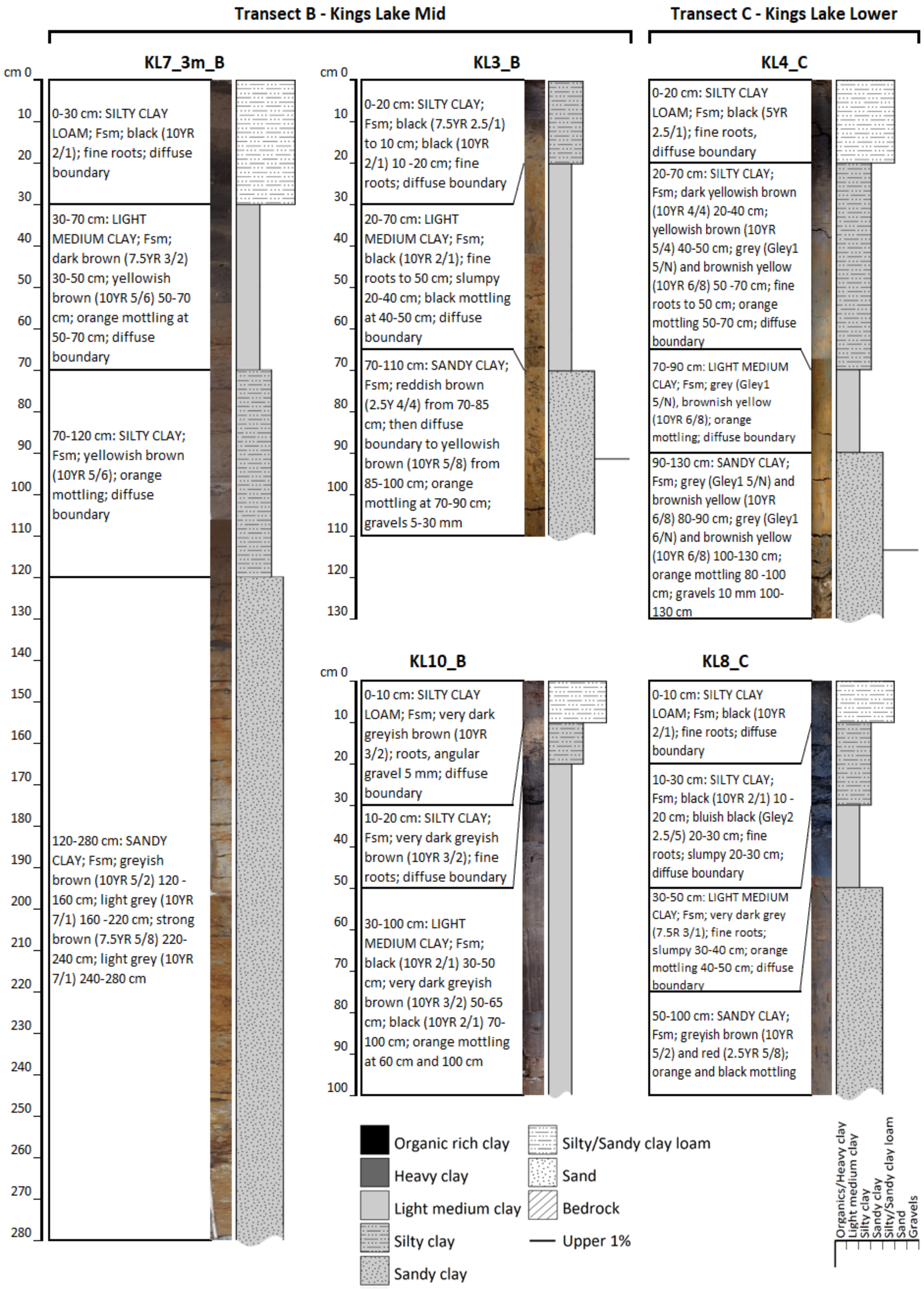
(Wetzel and Likens, 2000)



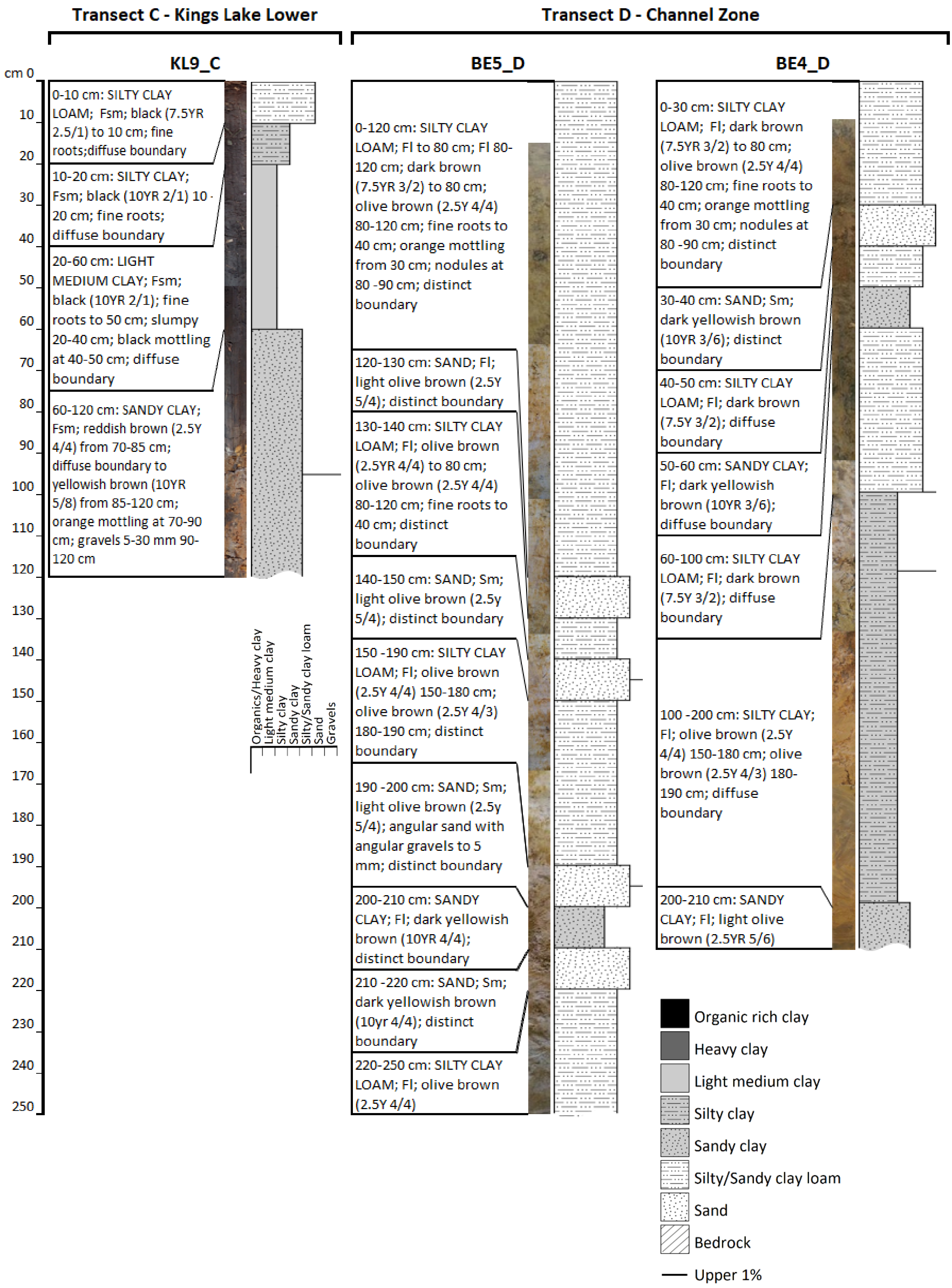
Appendix 3. Flowchart demonstrating processing steps used for laboratory aquatic gross primary productivity experiment, and images showing sediments inundated in buckets (bottom left), and light/dark bottles of BOD experiment (bottom right).



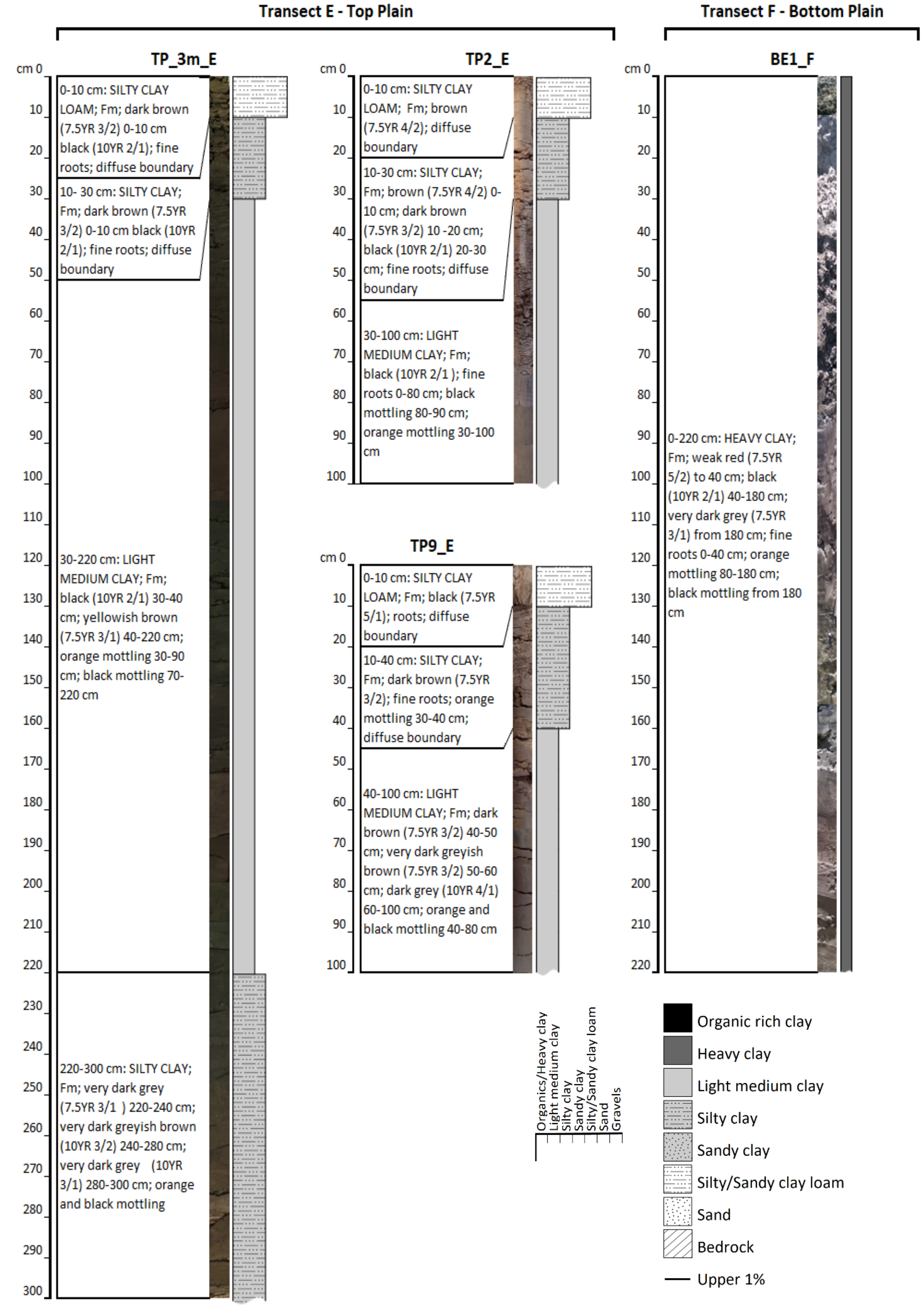
Appendix 4. Sediment columns of drill holes, bank exposures and cores, for Transects A – F at Kings Plains. Transect A scale in metres and Transects B-F scale in centimetres. All sediment columns display texture classes, facies (Miall, 1999), features and colour (moist) (Munsell Color, 2000). This page displays Transect A Kings Lake Upper.



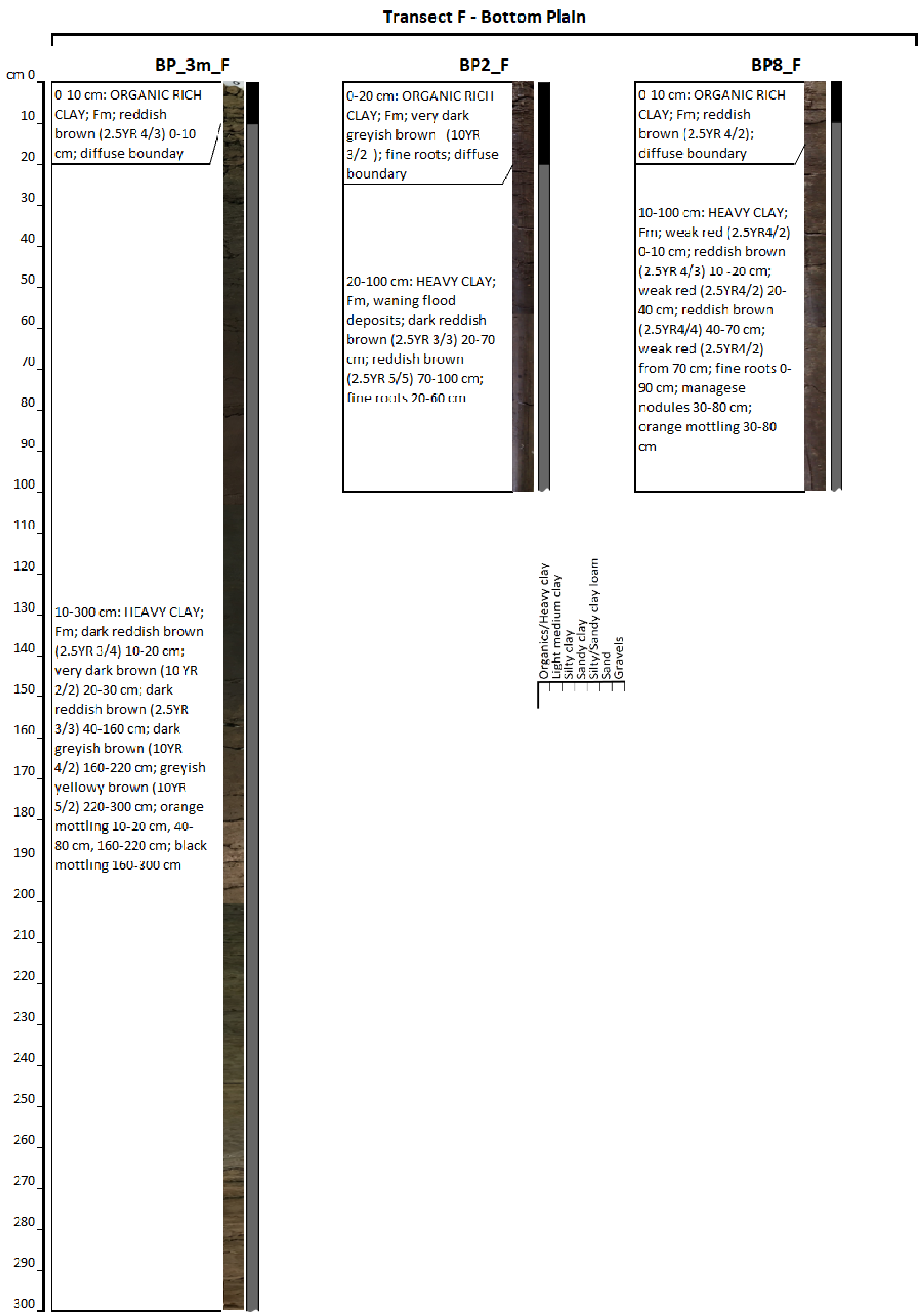
Appendix 4. (continued) Transect B Kings Lake Mid, and Transect C Kings Lake Lower (part 1/2).



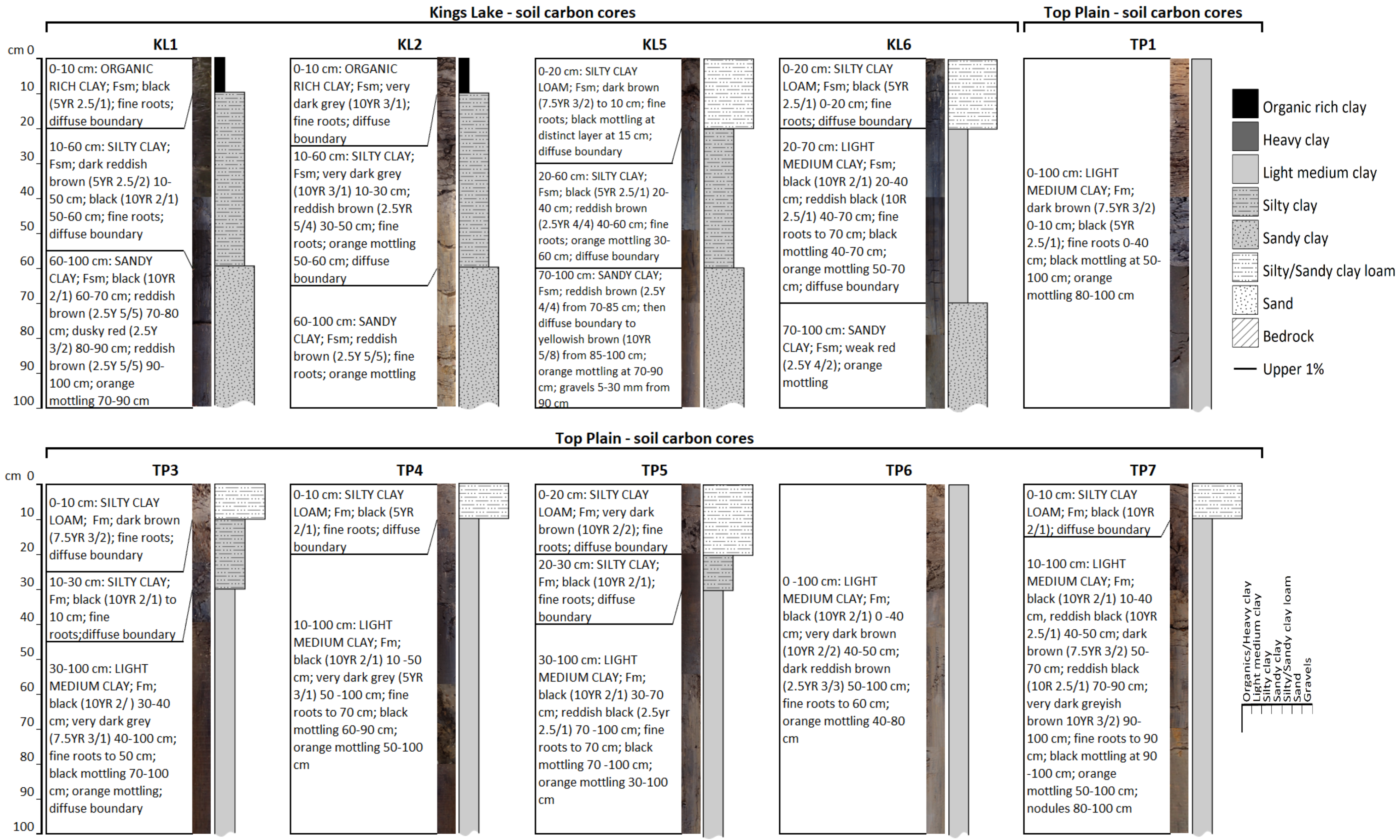
Appendix 4. (continued) Transect C Kings Lake Lower (part 2/2) and Transect D Channel Zone.



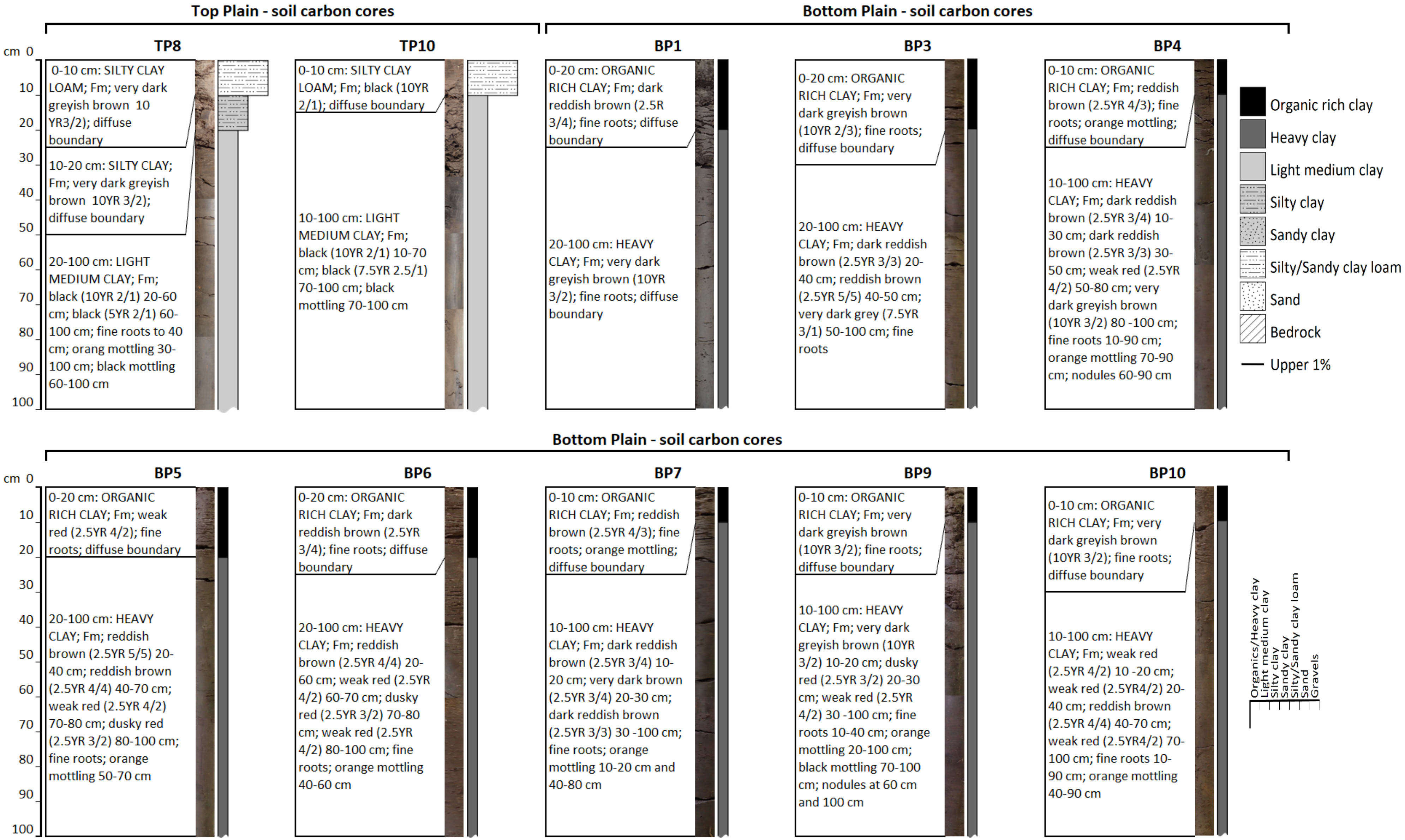
Appendix 4. (continued) Transect E Top Plain and Transect F Bottom Plain (part 1/2).



Appendix 4. (continued) Transect F Bottom Plain (part 2/2).



Appendix 5. Sediment columns of additional cores sampled for soil carbon at Kings Plains. Scale in centimetres. All sediment columns display texture classes, facies (Miall, 1999), features and colour (moist) (Munsell Color, 2000). This page displays cores from Kings Lake and Top Plain (part 1/2).



Appendix 5. (continued) Sediment columns of cores sampled for soil carbon from Top Plain and Bottom Plain (part 1/2).