Chapter 1
Introduction to Wetland Processes, Rehabilitation and Acid Sulfate Soils, with Reference to Estuarine Floodplain Wetlands

1.1 Introduction

Wetlands are important ecosystems as they cycle nutrients, trap sediment, attenuate floods, support a diverse range of vegetation, and provide breeding and feeding grounds for many fish, other aquatic animals and waterbirds. They may also provide drought relief in agricultural areas by retaining water for long periods of time. Wetlands are found all over the world, however, due to the variety of wetland types and lack of consistent definitions between countries, estimates of their worldwide extent range from 4.5 million km^2 to 9 million km^2 (Mitsch & Gosselink 2000; Ramsar Convention Secretariat 2006).

Wetlands throughout the world are very diverse and as a result, it is difficult to develop a definition which accounts for all possible characteristics of a wetland (Cowardin et al. 1979; Mitsch & Gosselink 2000). One of the most general definitions of a wetland was developed at the Ramsar Convention on Wetlands in 1971, which defines them as:

> "… areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six meters" (Ramsar Convention Secretariat 2006: 7).

While the Ramsar definition encompasses a wide variety of habitats (including rocky shores and coral reefs), it is internationally recognised and used as a basis for coordination between countries (Finlayson et al. 1999). Furthermore, the coastal and riparian zone adjacent to the wetland may be included as part of the wetland, as are islands or areas of marine water deeper than six metres at low tide located within the wetland (Ramsar Convention Secretariat 2006).

Other definitions of wetlands available in the literature are based on hydrology, soil type and aquatic fauna and flora, as these are the three main components common between all wetland types (Mitsch & Gosselink 2000). The definitions describe wetlands as areas where hydrology is the dominant influence on the development and establishment of hydric soils, hydrophytic vegetation and biological activity (e.g. Cowardin et al. 1979; NWWG 1997).

While there are a number of characteristics which vary between wetland types, there are also a number of factors which may vary slightly between wetlands of the same type (e.g. topography, shape of the wetland, vegetation density, soil and water chemistry, etc.). Hence, hydrological characteristics may vary between wetlands of the same type, making it important to develop an understanding of individual wetland hydrology prior to attempting rehabilitation or restoration works. However, often these projects begin without a thorough knowledge of the hydrology and therefore long-term monitoring is required to ensure rehabilitation efforts continue to positively progress. Development of wetland hydrology models pre- and/or post-rehabilitation provide valuable information for ongoing management of the wetland based on predictions or past experiences of the rehabilitation effort.

1.2 Aim and objectives of thesis and overview of chapters

The aim of this thesis was to investigate the hydrology and changes in water quality characteristics of an estuarine floodplain wetland, which was subject to a trial of reintroducing saline water and fresh water back into the wetland. The specific objectives were to:

i) determine if restored tidal exchange improved discharge water quality;
ii) investigate spatial and temporal patterns of surface water and groundwater chemistry;
iii) quantify the impact of tidal exchange management and climate on within-wetland water quality;
iv) investigate the hydrological characteristics of the wetland;
v) compare water quality characteristics between the rehabilitated wetland and other wetlands along the river;
vi) develop a conceptual model of the rehabilitated wetland; and
vii) provide recommendations for ongoing management to achieve the most desirable outcomes in terms of wetland water quality.

The remainder of this chapter provides background information about the hydrology and sediments (specifically acid sulfate soils (ASS)) of coastal floodplain wetlands. Management of these wetlands in terms of rehabilitation/remediation and the importance of hydrological models are also discussed.

A detailed description of the study area is provided in **Chapter 2**, including a history of the formation of the Clarence River and flood mitigation works on the floodplain, flora and fauna at the study site (identifying the important habitat provided by the wetland), and climate patterns over the study period. A preliminary investigation of water quality at the study site is provided in **Chapter 3**, the results of which aided in the establishment of a 2-year surface water quality monitoring program which is presented in **Chapter 4**. The longer-term study compared pre- and post-rehabilitation discharge water quality, examined spatio-temporal variation of within-wetland water quality, and quantified the impact of tidal exchange management and climate on surface water quality. Groundwater quality and hydrodynamics, along with linkages between groundwater and surface water at the study site are discussed in **Chapter 5**. The study examined spatio-temporal variation of groundwater chemistry, the effect of climate on groundwater quality, and the effectiveness of tidal exchange management and ponding to reduce oxidation of ASS.

In **Chapter 6**, spatial variation of water quality is compared and contrasted between three wetlands on the Clarence River floodplain, including the main study site. The purpose of the study was to determine what characteristics and external factors were common between wetlands, and thus the applicability of a conceptual model (developed for the main study site) to other coastal floodplain wetlands.

The conceptual model of wetland hydrology, focussing on water quality, at the main study site is presented in **Chapter 7**. The results from Chapters 3 - 6 are incorporated into the development of the model. Observations in the field, such as changes in vegetation type and cover (and therefore habitat and waterbird species), are also included in the model and related to the zonation of surface water quality identified in Chapter 4. Three main management options are discussed for Little Broadwater in terms of viability and positive or negative impacts. The applicability of the model to future floodplain wetland rehabilitation projects is discussed. **Chapter 8** provides general conclusions to the study and recommendations for future work at Little Broadwater.

1.3 Hydrology and the water balance of wetlands

Wetland ecosystems are comprised of three intrinsically linked components (Figure 1.1): i) hydrology; ii) the physiochemical environment, which includes sediments, and soil and water chemistry; and iii) biota (Hollis & Thompson 1998; Mitsch & Gosselink 2000). Each component has a feedback response to the other two, but ultimately all three components are controlled by climate and geomorphology. Hydrology is the principal component, as it determines the development, maintenance and productivity of wetlands (Bedford 1996; Callaway 2001) and the habitat of wetland types (Mitsch & Gosselink 2000). Hydrology also influences the development and dynamics of sediments, plant growth, diversity and dispersal, and access by fauna to breeding and feeding grounds (Callaway 2001).

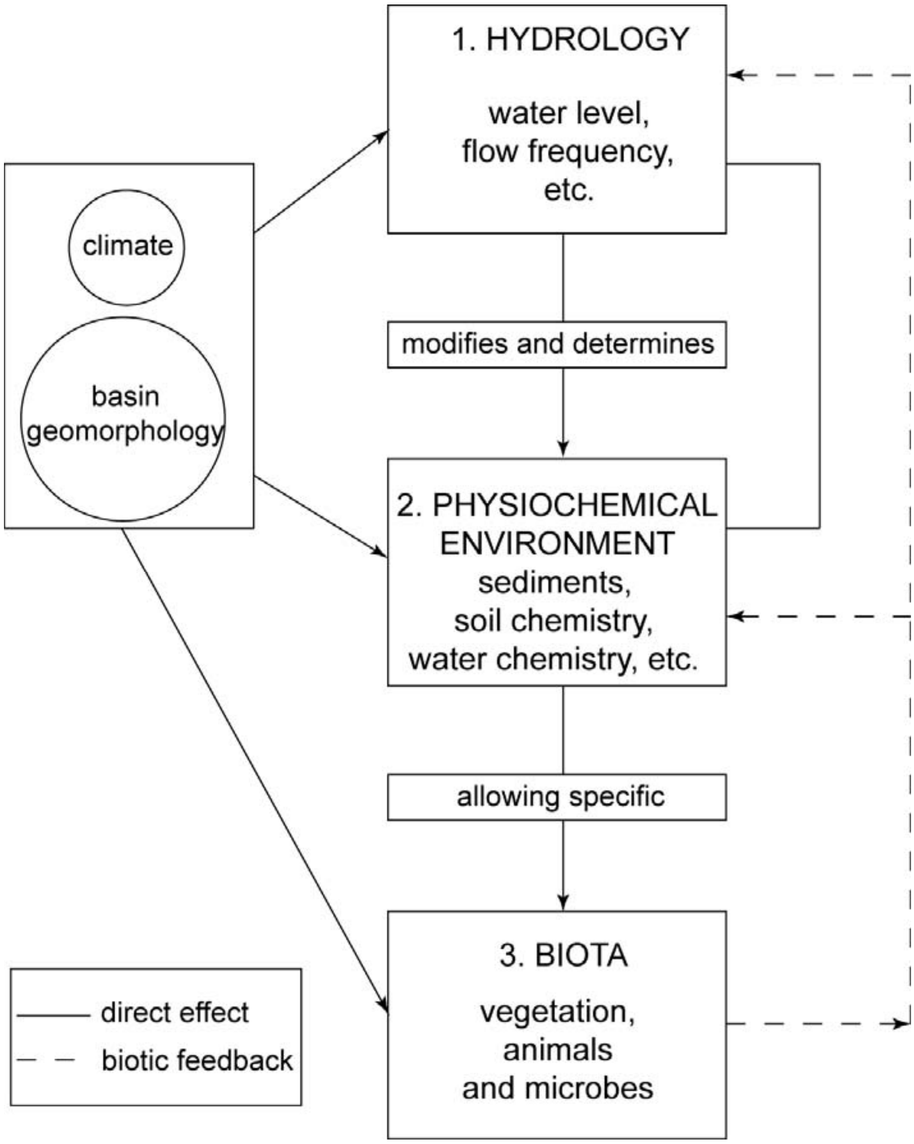

Figure 1.1: Conceptual diagram of the three functioning components of wetland ecosystems. Adapted from Mitsch and Gosselink (2000: 109).

The hydrology of wetland ecosystems is complex and is a function of many interacting factors, including seasonal variation, extremes in climate, variations in flow (groundwater, tidal regime, etc.) and vegetation (Hughes, Binning & Willgoose 1998). Hollis and Thompson (1998) identified six key variables of wetland hydrology that should be considered for the management and assessment of impacts on wetlands:

i) water level regime – the main factor controlling ecological processes and represents the seasonal patterns of surface and groundwater levels;
ii) level-area-volume relationship – refers to the connection between water level, the area of inundation and the volume of water within the wetland, and is important in hydrological modelling;
iii) water budget (balance) – includes precipitation, evapotranspiration, surface runoff, river and groundwater inflow/outflow and tidal exchange. The water balance is very important in understanding the functioning of a wetland (see Section 1.3.1);
iv) turnover rate – the flushing rate of a wetland, which influences the biota and chemistry of the wetland;
v) extremes – floods and droughts can have a significant impact on wetlands, including excessive erosion or deposition (during floods), or during droughts, desiccation or concentration of salts; and
vi) water quality – represents biochemical processes occurring within the wetland, natural and anthropogenic processes in the catchment, and determines the diversity of wetland plants and animals.

There are three main processes in wetland hydrology – inflows, storage and outflows – and the importance of each varies between wetland types (Walton, Chapman & Davis 1996). Despite the importance of these processes, the inflows and water balance of an individual wetland is often not measured (Gilvear et al. 1993), even though quantitative analysis of wetland hydrology is necessary for development of models and effective management (Walton, Chapman & Davis 1996).

Surface water flow is an important variable to be considered when examining the turnover rate, water quality and water budget of a wetland. Flow is driven or affected by a number of environmental factors, including vegetation type and density (Restrepo, Montoya & Obeysekera 1998), temperature, wind, water density (related to salinity) and river flows. Wind is a strong flow mechanism in wetlands, especially shallow systems with large areas of open water. Geyer (1997) found that the turnover rate of shallow estuaries could be increased three-fold due to wind alone. When wind-driven flow is coupled with other mixing process, the flow of a wetland system may be dramatically increased or decreased.

1.3.1 The wetland water budget

Water budgets are an important component of wetland hydrology as they provide the framework of hydrological, biogeochemical and ecological linkages between wetlands and their surrounding environment (Drexler et al. 1999). A water budget is generally an equation, or series of equations, balancing inflow, storage and outflow of individual wetland systems, and is the simplest form of a hydrological model. More complex hydrological models use the water budget as a starting point, and also include factors such as water quality, flushing rate of the wetland, specific water flow paths, solute balance and vegetation (refer to Section 1.7 for more detail).

A range of wetland water budgets are described in the literature, although they are all based on the main principals of inputs and outputs. Mitsch and Gosselink (2000) express the wetland water budget as:

$$\Delta V/\Delta t = P_\mathrm{n} + S_\mathrm{i} + G_\mathrm{i} - ET - S_\mathrm{o} - G_\mathrm{o} \pm T \qquad \text{(Equation 1.1)}$$

Where:

$\Delta V/\Delta t$ = the change in volume of water storage in wetland per unit time

P_n = net precipitation (Equation 1.2)

S_i = surface inflows

G_i = groundwater inflows

ET = evapotranspiration

S_o = surface outflows

G_o = groundwater outflows

T = tidal inflow (+) and outflow (–)

All parameters can be expressed in terms of depth per unit time (e.g. cm yr^{-1}) or in terms of volume per unit time (e.g. m^3 yr^{-1}).

$$P_\mathrm{n} = P - I \qquad \text{(Equation 1.2)}$$

Where:

P = total precipitation

I = interception of rainfall by vegetation

More detail regarding evaporation processes are provided by Thompson and Finlayson (2001) (Equation 1.3) in their wetland water budget. Evaporation from open water is separate to evapotranspiration from soils and vegetation and the equation also includes evaporation of water intercepted by vegetation and water inputs or outputs by humans.

$$V_t = V_{t-1} + (P - I) + Q_\mathrm{i} + G_\mathrm{i} - E_\mathrm{o} - E_\mathrm{t} - Q_\mathrm{o} - G_\mathrm{o} \pm T \pm H \qquad \text{(Equation 1.3)}$$

Where:

V_t = volume of water in the wetland at time t

V_{t-1} = volume of water in the wetland at time interval $t - 1$

The following are inputs and outputs between time $t - 1$ and t:

P = precipitation directly onto the wetland

I = evaporation of water intercepted by vegetation within the wetland

Q_i = contributions from rivers and surface runoff to the wetland

G_i = groundwater inflow to the wetland

E_o = evaporation from areas of open water

E_t = evapotranspiration from wetland soils and vegetation

Q_o = outflows from the wetland carried by streams and rivers

G_o = groundwater recharge from the wetland

T = tidal inputs (+) or outputs (–) associated with the wetland

H = human water withdrawals (e.g. for irrigation) or returns of water to the wetland (e.g. effluent discharges).

Precipitation and evapotranspiration are the dominant controls on the wetland water budget, as these ultimately determine the freshwater inputs into a wetland. Surface runoff is a function of rainfall, evapotranspiration and infiltration, which is dependent on the saturation of sediments (watertable depth) within the local catchment. Watertable depth is also dependent on rainfall and evapotranspiration patterns. Furthermore, rainfall and evapotranspiration directly affect the salt balance within a wetland either via direct input/output or by influencing the tidal forcing in the adjacent creek or river.

1.3.2 Groundwater

Of all the components of the wetland water budget, groundwater has been the most difficult to quantify due its complex nature of flow through sediments (Drexler et al. 1999) and the number of factors which affect the flow. Groundwater flow in a wetland is controlled by surface flows, subsurface inflows and outflows, infiltration, evapotranspiration and/or tidal flows (Duever 1988; Greenblatt & Sobey 1999). Groundwater flows were considered insignificant in older studies (e.g. Duever 1988) due to the complexity and amount of time required to understand the flows, which often contributed much less to the overall hydrological functioning of the wetland than surface water flows. However, some wetlands are dependent on groundwater as their primary source of water (Gilvear et al. 1993), rather than surface runoff or tidal flows. Therefore, an assessment of groundwater flows (e.g. hydraulic conductivity) should be undertaken when developing wetland water budgets to determine the contribution of groundwater to the system.

1.4 Flood mitigation and the impact on wetland hydrology

Floodplain hydrology has been severely altered due to flood mitigation works and drainage policies. Flood mitigation works include weirs, drains, one-way floodgates, culverts and levees. On coastal floodplains mitigation works are used to control flooding by reducing flood frequency, aid in the removal of flood waters and exclude tidal inflows (Haskins 2000; Pressey & Middleton 1982; Smith 2002). These mitigation works and drainage policies have ultimately led to severe changes in floodplain and wetland hydrology (Table 1.1) by altering the flooding and drying cycles of wetlands, thereby lowering the watertable and changing salinity regimes (Portnoy & Giblin 1997b; Todd & Mays 2005; Turner & Lewis 1997). As a result, the diversity of wetland types has been reduced (Kentula et al. 1992 cited in Bedford 1996: 57).

Changes to wetland hydrology have had both short- and long-term impacts on wetlands. Short-term impacts include a reduction in total wetland habitat, loss of mangroves and other aquatic vegetation, reduction in access for aquatic animals and increased aquatic weeds in drains (Haskins 2000). Long-term impacts include oxidation of ASS due to lowering of the watertable, poor water quality (due to a reduction in exchange and oxidation of ASS), increased decomposition of organic matter, and compaction and subsidence of sediments (Haskins 2000; Smith 2002).

To reduce or prevent flooding of low-lying areas (i.e. backswamps), the hydraulic gradient of many landscapes has been increased by a factor of at least 5 through channel straightening and the installation of one-way floodgates (White et al. 1997). One-way floodgates prevent flushing of wetlands and natural channels with river and/or tidal water, and only allow water to drain off the floodplain at low tide or when the hydraulic head is higher in the wetland than in the river. The roughness coefficient of channels has also been reduced by a factor of 5 due to clearing (White et al. 1997), which also contributes to faster removal of water from the floodplain than would have naturally occurred. As a result of these flood mitigation schemes, flood waters are now drained from many areas within 5 days, compared to 100 days prior to modification (White et al. 1997). Many wetlands are now dry for the majority of the year and those that still have some water have often been converted from one wetland type to another, e.g. a permanent brackish wetland to a seasonally inundated freshwater wetland.

Table 1.1: Comparison of floodplain wetlands before and after drainage (derived from Smith 2002).

Component	Before Drainage	After Drainage
Hydrology	▪ Natural wet/dry cycle based on seasonal rainfall and evaporation. ▪ Remained flooded for 3-6 months during and after the wet season. ▪ Naturally produce humic acid from decaying organic matter. During prolonged droughts may have produced some sulfuric acid.	▪ Drainage has made the majority of floodplain wetlands permanently dry. ▪ Floodwaters are removed within 6-10 days. ▪ Acidic groundwater can discharge for months. ▪ Black (deoxygenated) water can discharge for 1-2 weeks after a flood.
Flora and Fauna	▪ Major habitat and nursery for fish ▪ Waterbird habitat.	▪ Increase in dryland pasture. ▪ Invasion of trees such as *Casuarina glauca* (swamp oak) and *Melaleuca quinquenervia* (tea tree). ▪ Decline in aquatic wildlife due to acidic waters, loss of protective vegetation for waterbirds and limited access.
Management	▪ Fires not frequent and due to moisture were low intensity, retaining peat. ▪ During spring droughts cattle were moved to backswamps where there was an abundance of vegetation.	▪ Fires occur more frequently and intensely due to drier conditions, which remove vegetation and peat layers. ▪ Drainage has lead to ASS exposure and acid scalds. ▪ Depending on intensity of drainage, backswamps provide limited or no drought reserve.

As a direct result of increased drainage and removal of surface water, the watertable has been lowered in many wetlands. Groundwater adjacent to drains is lowered to low tide level

(Johnston, Slavich & Hirst 2005b) and during dry periods evapotranspiration can lower the watertable further, changing the hydraulic gradient so groundwater flows away from the drain (Blunden & Indraratna 2000). The impact of lowering the watertable is particularly important in wetlands underlain by ASS, as these may be exposed and have considerable impacts on water quality when the watertable increases above the oxidised layer during high rainfall.

Another impact of flood mitigation schemes has been the increased formation of iron monosulfidic black ooze (MBO), particularly in drains. Iron monosulfides occur naturally in reducing conditions such as estuaries, mangroves, coastal lakes, salt marshes and tidal wetlands (Bush, Fyfe & Sullivan 2004). Decaying organic matter combined with sulfate-reducing bacteria and soluble iron leads to the formation of MBOs (Bush, Fyfe & Sullivan 2004). Drains provide an ideal environment for these processes to occur. If mobilised, such as during flood events when high velocity flows occur in drains, MBO can completely deoxygenate the water column (Sullivan et al. 2002 cited in Bush et al. 2004: 603). As little as 1 mg MBO L^{-1} can completely deoxygenate water (Bush et al. 2004). Furthermore, as the dissolved oxygen concentration increases after the initial deoxygenation, acidification may occur as ferrous iron is oxidised to ferric iron and sulfur is oxidised to sulfate (Bush, Fyfe & Sullivan 2004).

1.5 Acid sulfate soils

The impact of flood mitigation works on wetlands is greatly increased when implemented in areas containing ASS. As discussed earlier, ASS are often exposed when the watertable is lowered due to drainage and increased evapotranspiration of the groundwater. Oxidation of ASS can lead to a severe decline in water quality and cause disease and mass mortality of aquatic animals due to acid and metals mobilised from the exposed soils. While ASS most commonly occur on coastal floodplains (termed Coastal Lowland Acid-Sulfate Soils (CLASS) by Burton et al. (2008)), sulfidic sediments also occur in inland freshwater wetlands (Hall et al. 2006). Due to the location of the study site, however, the following discussion of the formation of ASS, the chemical reactions that take place during oxidation, the resultant impact on water quality, and the effect of flood mitigation works on the oxidation process is in relation to coastal floodplains only.

1.5.1 Formation of sulfidic sediments and oxidation processes

Sulfidic sediments were formed in estuarine lowlands during the last sea-level rise in the Holocene period, less than 10 000 years before present (Dent 1986; Sammut, White &

Melville 1996). They are formed by the reduction of dissolved sulfate (SO_4^{2-}) to sulfide by bacteria breaking down organic matter in low energy tidal environments (Sammut, White & Melville 1996), such as estuarine wetlands and coastal backswamps. Bacterial reduction of SO_4^{2-} can only occur in anoxic environments (Berner 1984). In NSW the predominant sulfidic sediment is pyrite (FeS_2), but it can also be in the form of iron monosulfide (FeS) or greigite (Fe_3S_4) (Indraratna, Tularam & Blunden 2001). Pyrite forms when detrital iron minerals react with hydrogen sulfide (H_2S) (Equation 1.4; Figure 1.2).

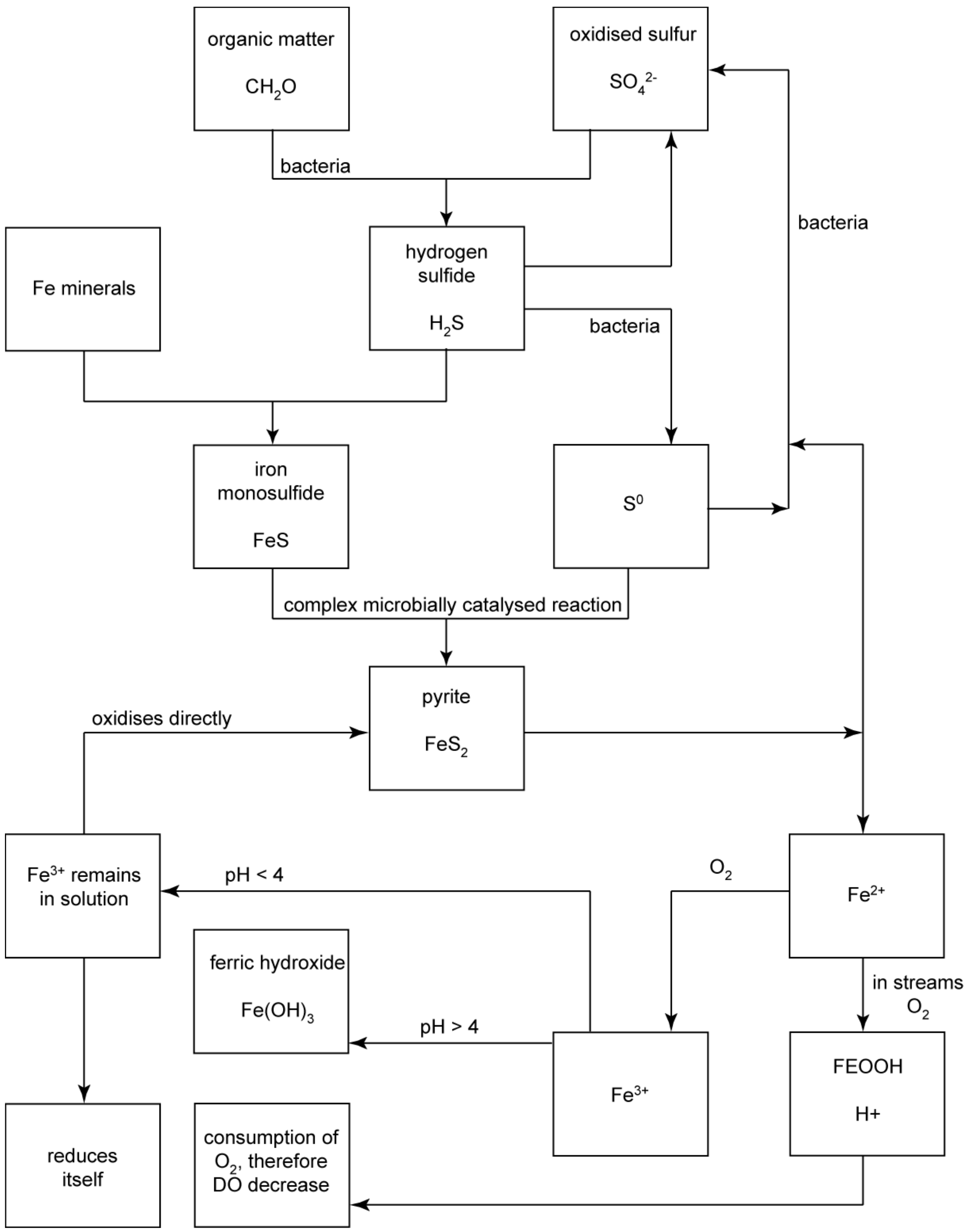

Figure 1.2: The development and oxidation of pyrite and subsequent formation of acid sulfate soils (derived from Berner (1984) and White et al. (1997)).

$$Fe_2O_3 + 4SO_4^{2-} + 8CH_2O + \tfrac{1}{2}O_2 \xrightarrow[\textit{reduced conditions}]{\textit{bacteria}} 2FeS_2 + 8HCO_3^- + 4H_2O \qquad \text{(Equation 1.4)}$$

Under anoxic reducing conditions, sulfidic sediments are stable, however, oxidation converts pyrite to ferrous iron (Fe^{2+}) and SO_4^{2-}, and produces acid in the form of hydrogen ions (H+) (Blunden & Indraratna 2001; White et al. 1997) (Equation 1.5; Figure 1.2). The acid may then react with estuarine clays and mobilise metals and salts such as aluminium (Al), iron (Fe), magnesium (Mg), potassium (K), sodium (Na) and manganese (Mn) into the soil (Green et al. 2006; White et al. 1997). Oxidation of 1 tonne of FeS_2 can produce 1.6 tonnes of sulfuric acid (Melville & White 2000).

$$FeS_2 + 14/4O_2 + 7/2H_2O \rightarrow Fe(OH)_3 + 2SO_4^{2-} + 4H^+ \qquad \text{(Equation 1.5)}$$

Although unoxidised sulfidic sediments are stable in anoxic conditions, soluble ferric iron (Fe^{3+}) present in partially oxidised sulfidic sediments can continue to oxidise pyrite in the absence of oxygen (White et al. 1997) (Equation 1.6, Figure 1.2). This form of oxidation can be increased five- or six-fold if bacteria such as *Thiobacillus* are present (Preda & Cox 2004).

$$FeS_2 + 14Fe^{3+} + 8H_2O \rightarrow 15Fe^{2+} + 2SO_4^{2-} + 16H^+ \qquad \text{(Equation 1.6)}$$

1.5.2 Acid sulfate soils, water quality and flood mitigation structures

Oxidation of ASS can drastically degrade water quality in wetlands and rivers, leading to serious health implications for flora and fauna. Sulfuric acid produced during the oxidation of ASS is leached into surface water and groundwater during rainfall events, along with other metals and salts released by the sediments in response to the sulfuric acid. Iron flocs may smother and kill vegetation and acid-tolerant species such as *Eleocharis* spp. (spikerush) and the *Nymphaea caerulea* ssp. *zanzibarensis* (cape waterlily) may invade and dominate the wetland (Sammut, White & Melville 1996). Mass mortality of fish, crustaceans and worms may occur under acidic conditions (Sammut, White & Melville 1996). Chronic effects of drain and estuary acidification and mobilisation of heavy metals include accumulation of iron on the gills of fish, oysters and crustaceans, eventually fusing the gills together; triggering diseases such as epizootic ulcerative syndrome (red-spot disease) in fish; disturbance to reproduction and recruitment; barriers to migration; and reduced species diversity (Sammut, White & Melville 1996).

The effects of acidification on habitat and aquatic animals is dependent on the quantity of acidic water and the duration and frequency of acidification (Sammut, White & Melville 1996). A number of factors determine how much acidic water is discharged from a wetland system, including rainfall patterns and watertable depth. Large, intense rainfall events can result in the acid water being released as a large slug, while smaller rainfall events can cause the acidic water to be released as a constant flow into the estuary (Atkinson 2000; Sammut, White & Melville 1996).

Watertable depth is also an important factor affecting the amount of acidic water discharged from a wetland (Johnston, Slavich & Hirst 2004a; Wilson, White & Melville 1999). Groundwater is a major source of acid water in drained wetlands and can continue to seep into drains and be discharged for months (Smith 2002). Research by Wilson, White and Melville (1999) suggests that if a large rainfall event occurs and the watertable is low, there is available pore space where the runoff can be stored and therefore not release acid into the surface water. However, if the watertable is high and a large rainfall event occurs, there is no available pore-space to absorb the runoff and a flush of acidic discharge may occur. More recent studies have determined that acid export is predominant when the groundwater level is in a narrow elevation range, termed the acid export window (Johnston, Slavich & Hirst 2004b). This window is defined by the backswamp topography and the local tidal minima (Johnston, Slavich & Hirst 2004a), thus making the range different for each wetland.

Although oxidation of ASS is a natural process that often occurs during droughts, the drainage of wetlands and other low-lying areas on coastal floodplains has increased the occurrence, duration and severity of oxidation and acidification events. Drains increase the amount of acidic water transported into the estuary through increased rates of surface water and groundwater drainage. One-way floodgates prevent dilution and neutralisation of acid water by saline water available in the river, and may also retain acidic water within the drain for months (Atkinson 2000), especially during dry periods. Increased evapotranspiration of groundwater due to the lack of surface water can draw acid salts to the sediment surface, causing acid and salt scalds (Green et al. 2006; Rosicky et al. 2006). This can then cause large areas of vegetation to die and thus expose groundwater to increased evaporation.

1.6 Wetland rehabilitation

The terminology for improving degraded wetlands is imprecise and vague throughout the literature (Kentula 2000). Terms such as restoration, rehabilitation and remediation are used interchangeably and generally refer to a reversal of the hydrological changes made to wetlands (Turner & Lewis 1997). Restoration implies that the wetland has been returned to its pre-disturbance condition in terms of physical, chemical and biological processes (Cairns 1988; Lewis 1989). However, restoring wetlands to their original condition may not be practical or possible due to social and/or economic costs (White et al. 1997), or the level of disturbance. In addition, restoration may not be technically possible within the given time frame for most projects. Therefore, the term ‘rehabilitation’ used henceforth, refers to improving or returning ecological functioning of the wetland (National Research Council (U.S.) & Committee on Restoration of Aquatic Ecosystems - Science, Technology, and Public Policy 1992). The term restoration is used only in relation to returning tidal flows to the altered wetland.

Rehabilitation of drained estuarine wetlands often involves restoring tidal exchange to improve water quality, improve habitat for fauna and increase flora and fauna species diversity. Rehabilitation can be initiated deliberately or through natural events such as storms breaching levees (e.g. Eertman et al. 2002). Many projects involving tidal restoration assume that the wetland will return to its natural state of structure, function and stability (Neckles et al. 2002) if the primary cause of altered hydrology is removed. However, restoring the correct hydrology to a wetland often requires more than just removing levees, floodgates or weirs to allow exchange (Callaway 2001).

Rehabilitation by restoring tidal exchange has been achieved at wetlands around the world. For example, in Puerto Rico at the Laguna Boca Quebrada, Vieques a road was blocking tidal exchange and had caused the death of more than 100 ha of mangroves (Turner & Lewis 1997). Reconnection of the lagoon to the ocean resulted in the regrowth of the mangroves. At the Sieperda Tidal Marsh in The Netherlands fields and pastures were converted back to tidal marsh when the dike was breached by a storm, restoring tidal exchange to the marsh area (Eertman et al. 2002). Studies in the north-eastern USA have also shown that reintroducing tidal flushing to restricted salt marshes can lead to the restoration of ecological functions (Roman, Garvine & Portnoy 1995). In Australia, a study by Streever and Genders (1997) examined the effect of removing culverts to allow increased tidal flow to a salt marsh on Kooragang Island, NSW. They found there was considerable expansion of salt marsh over a single growing season due to the increased exchange.

When restoring tidal exchange to wetlands, it is essential to understand how the wetland functions in terms of hydrology to ensure the appropriate amount of exchange is achieved. An example where this problem occurred is the Barn Island marshes in Connecticut, USA, where tidal flushing was restored by installing a four foot wide culvert to allow water to pass through a dike (Rozsa 1995). Salt marsh vegetation began to reappear in the southern part of the marsh, but not in the middle or upper marsh areas (Rozsa 1995). It was found that not only was the culvert too small to allow a sufficient amount of water to flood the marsh, but that it was also restricting drainage of water back into the Sound (Rozsa 1995). A seven foot wide culvert was later installed and as a result the salt marsh vegetation advanced to the upper marsh area (Rozsa 1995).

Prior to wetland rehabilitation it is important to consider factors such as changes to sediment structure and subsidence. Estuarine muds and clays found in wetland systems are dependent upon available water in the pore space to retain the structure of the sediment. When the wetland is drained the sediment structure collapses, resulting in subsidence, shrinking and cracking of the mud. For example, in Indonesia the kaolinite clays shrink in dry conditions but have limited swelling during wet periods due to the low absorption capacity of the clay (Hamming & van den Eelaart 1993). Changes to sediment structure and the wetland topography may result in water spreading further over the floodplain when reflooded than prior to drainage. Areas that were previously only inundated during flood events could now be permanently under water, which may change the vegetation type and habitat within and surrounding the wetland.

1.6.1 Rehabilitating acid sulfate soil wetlands

Many studies have examined the effect of flood mitigation drains on acid production and water quality in coastal ASS landscapes, with the majority suggesting either installing weirs or tidal buffering as solutions to improve water quality. The technique used for rehabilitation or remediation depends upon whether the wetland is naturally a freshwater or estuarine wetland, the desired outcome (i.e. improvement of fish access, vegetation health and/or discharge water quality), and other limitations such as poor circulation and feasibility. Rehabilitation strategies in ASS landscapes can be in the form of transformation, dilution, neutralisation and containment (Atkinson 2000), and can be applied individually or in combination. Transformation involves reducing the acid into other stable compounds, however, it has not yet been tested in the field (Atkinson 2000).

Dilution requires adding enough fresh or saline water to raise the pH, or allowing gradual leakage of acid into the river (Atkinson 2000). The latter is often a suitable strategy for large freshwater rivers (Atkinson 2000) which have limited neutralisation capacity. Dilution strategies are essentially a combination of containment and neutralisation strategies (Tulau 2007). Containment strategies attempt to prevent the acid from entering the environment by reflooding the affected area and holding the acid in the soil profile, natural depression, artificial ponds or drains (Atkinson 2000; Smith 2002; White et al. 1997). This may be achieved with devices such as weirs or dropboard culverts, resulting in stable and higher watertables, thereby decreasing oxidation of pyrite and reducing acid discharge (Glamore & Indraratna 2001; Johnston, Slavich & Hirst 2004b; White et al. 1997). Maintaining a high watertable is a more common containment strategy than completely reflooding the backswamp due to the large scale of agriculture on many drained coastal floodplains (White et al. 1997). Social and economic costs also make it impossible or undesirable to return all drained wetlands to their original state (White et al. 1997). However, to effectively reduce acid export without affecting current agricultural land use, soils need to have low hydraulic conductivity, the land often needs to be laser levelled, the drainage redesigned and there needs to be responsive watertable control (Tulau & Henderson 2001).

There are a number of other limitations associated with containment strategies which maintain a high watertable to reduce or prevent oxidation of ASS. Maintaining the watertable above the oxidised ASS may not increase pH or reduce the redox potential if there is low organic matter content and iron-reducing bacteria are absent (Equation 1.6; Danh 1991 cited in Tuong 1993: 267; van Breeman 1976 cited in Tuong 1993: 267). Oxidation of ASS can also continue in the absence of oxygen by Fe^{3+} and thus acid production can continue to occur with containment strategies. Another limitation of watertable management is that the sulfidic sediments may be at different depths throughout the management area, and therefore the watertable needs to be maintained at different heights to ensure the soils are not exposed (Tuong 1993). The watertable would need to be maintained at the shallowest depth, however, this may result in loss of land due to surface water flooding because of the floodplain topography. Furthermore, regions with long dry seasons and poor groundwater quality may not benefit from a raised watertable as the high evaporation could continue to draw acid salts and metals to the sediment surface (Tuong 1993). Maintaining a high watertable may also not prevent the leaching of Al and Fe (adsorbed in the soil matrix) into drains (Indraratna, Tularam and Blunden 2001).

Containment strategies which reflood the wetland can often be more beneficial than just maintaining a high watertable, as there are improvements in water quality and the provision of ecological benefits (Eertman et al. 2002). Acid scalds are remediated by the regrowth of water-tolerant plants, providing both increased grazing potential and restored habitat for aquatic fauna and waterbirds. Research has shown that wet pasture species are more nutritious than dry land species and also have a higher grazing potential (Clay et al. 2007).

Neutralisation techniques involve buffering acidic water with saline water (and possibly freshwater) or chemicals, and is often used in a similar way and/or in conjunction with containment and dilution techniques. Chemical neutralisation involves the addition of lime, dolomite or magnesite to areas with acidic discharge (Atkinson 2000; White et al. 1997). However, this can be expensive if the affected area is large and reapplication is needed. For example, the Tuckean Swamp floodplain in northern NSW covers an area of 4000 ha and has an estimated potential of 1.3×10^6 tons of sulfuric acid, which the cost to neutralise with lime would be approximately $250 million (White et al. 1997). Neutralisation with saline or freshwater is more economically viable than chemical neutralisation (White et al. 1997) and is often referred to as ‘flushing’ as there is regular exchange of drain and river water (Tulau 2007).

Neutralisation techniques work best in conjunction with containment and dilution strategies. This generally involves the installation of a weir to control the surface water level within the wetland, dropboard structures within the weir to allow exchange between the wetland and the river, and modification of the one-way floodgates by installing two-way tidal gates to allow control the amount of tidal exchange. As with other management strategies, there are benefits and negative impacts associated with both saline and freshwater neutralisation techniques.

Rehabilitation and management via freshwater reflooding is beneficial for drained floodplain areas which are used for grazing. Benefits of freshwater reflooding include: reduction of acid generation, discharge and groundwater movement to drains; increased vegetation and organic matter cover in the backswamp; increased grazing and habitat values of the backswamp; and a reduction in occurrences of low dissolved oxygen due to increased vegetation cover (Tulau & Henderson 2001). However, there are a number of problems associated with freshwater rehabilitation of ASS wetlands. Firstly, freshwater has a lower neutralising capacity than brackish water and therefore requires large amounts of water to be exchanged frequently (i.e. dilution) (White et al. 1997). Secondly, freshwater neutralisation may lead to the formation of MBO's through a microbially catalysed reaction, where SO_4^{2-} is reduced to hydrogen sulfide (H_2S) and precipitated Fe^{3+} is turned back into dissolved Fe^{2+} (Equation 1.7) (White et al. 1997). While this reduces soil acidity the potential for deoxygenating of the water column during floods is increased due to increased MBO's. Furthermore, in regions subject to variable climatic conditions, limited freshwater availability and increased evaporation during dry periods can lead to desiccation, exposing MBO's which re-oxidise and aid in pyrite oxidation (White et al. 1997).

$$SO_4^{2-} + 2H^+ + 2CH_2O \rightarrow H_2S + 2H_2O + 2CO_2Fe(OH)_3 + 2H^+ + \tfrac{1}{4}CH_2O \rightarrow Fe^{2+} + {}^{11}/_4\, H_2O + \tfrac{1}{4}CO_2 \qquad \text{(Equation 1.7)}$$

Neutralisation via brackish water reflooding is a more effective technique than freshwater neutralisation. Bicarbonates, which are found naturally in saline water, have the potential to buffer and neutralise acidic water (Equation 1.8) through the generation of sulfides and bicarbonate alkalinity (Equation 1.9) (Berner 1971 cited in Portnoy & Giblin 1997a: 1060). Ferric oxyhydroxides are reduced to form makinawhite (a sulfidic sediment) and release base (Equation 1.10) (Fenchel & Blackburn 1979 cited in Portnoy & Giblin 1997a: 1060). However, the effectiveness of tidal buffering is dependent upon a number of factors including the bicarbonates and acid concentrations, salinity levels and the hydrodynamics of the wetland (Indraratna, Glamore & Tularam 2002).

$$Ca^{2+} + HCO_3^- + H^+ + SO_4^{2-} \rightarrow H_2CO_3 + Ca^{2+} + SO_4^{2-} \qquad \text{(Equation 1.8)}$$

$$2CH_2O + SO_4^{2-} \rightarrow H_2S + 2HCO_3^- \qquad \text{(Equation 1.9)}$$

$$2FeO{\cdot}OH + 3HS^- \rightarrow 2FeS_{mack} + S^0 + 3OH^- + H_2O \qquad \text{(Equation 1.10)}$$

When restoring tidal exchange to drained ASS wetlands, there are a number of factors which should be taken into consideration, particularly in relation to metals and nutrients. Reflooding the wetland can mobilise arsenic (As), Al, Fe, Mn, nickel (Ni) and zinc (Zn) which can have negative impacts on the environment (Section 1.5.2). However, an increase in pH to near neutral reduces Al toxicity (Portnoy & Giblin 1997a) and the formation of acid volatile sulfide (AVS) reduces the mobility of Fe^{2+}, Mn Ni and Zn (Burton et al. 2008). Reflooding with brackish water has the potential to mobilise nutrients which could lead to a reduction in dissolved oxygen due to increased algal growth (Portnoy & Giblin 1997a). While these negative effects may only be temporary, it is important to understand the biogeochemical consequences of tidal restoration to limit these detrimental outcomes. Other short-term negative effects may be the die-back of non-saline tolerant vegetation which could lead to a drop in dissolved oxygen levels, and increased acid discharge as surface acids are removed from the system (Haskins 2000). However, as the wetland adjusts to the new tidal regime (e.g. an increase in salt-tolerant plants and improving access for aquatic fauna) the negative impacts will decrease in severity.

A number of studies have shown the effectiveness of restoring tidal exchange for improving drain water quality behind floodgates. Glamore and Indraratna (2001) found that there was an increase in drain water quality, and was most notable during drier periods when freshwater flows were reduced and did not prevent ingress of saline water. They reported more than a 10-fold increase in water quality in terms of pH, and dissolved oxygen levels remained above 6 mg L^{-1}, thereby reducing secondary oxidation of Fe^{2+}. Haskins (2000) also noted that when floodgates are managed to allow tidal flushing, the water behind the gates is of better quality both chemically and in terms of habitat value. However, there are few examples of restoring tidal inundation/exchange to large-scale ASS wetlands (Johnston et al. 1999).

The frequency, magnitude and duration of floodgate opening will control the extent and continuance of water quality improvement, as will the interaction between the volume and quality of inflow and outflow water (Johnston, Slavich & Hirst 2002; Johnston, Slavich & Hirst 2005a). Research has shown that short duration and less frequent floodgate openings with large magnitudes of water only result in short-term improvements (Johnston, Slavich & Hirst 2002), with water quality rapidly reverting to previous conditions when floodgates are closed (Johnston, Slavich & Hirst 2005a). Portnoy (1999) suggests that tidal flushing should be restored in small, incremental stages and water quality should be monitored for changes in pH, sulfide concentrations, dissolved nutrients, Fe^{2+} and dissolved oxygen.

There are a number of other advantages and disadvantages associated with tidal reflooding to rehabilitate ASS wetlands. Advantages include the submergence of sulfidic layers, bicarbonates aid in neutralising acid in the soil and water, there is a twice-daily exchange of water which increases the neutralising capacity of the water, excess salts are removed and nutrients are provided (Mitsch & Gosselink 2000; Portnoy & Giblin 1997a; Tularam & Glamore 2004; White et al. 1997). During dry periods exchange can continue to prevent desiccation of the wetland. Disadvantages of tidal reflooding include limited saline inflow during high rainfall (and therefore reduced buffering capacity), and salinisation of surrounding higher land due to existing drainage structures which readily transport water into the back areas of the floodplain (Mitsch & Gosselink 2000; Smith 2002; White et al. 1997).

Existing drainage structures may increase the amount of water that can be exchanged which may lead to increased salinisation of soils (White et al. 1997). Increased salinisation of upland areas is a common concern when proposing tidal reflooding. Wetlands with high soil hydraulic conductivity have an increased risk of groundwater salinisation (Johnston, Slavich & Hirst 2005b), so it is essential that the conductivity be tested before reintroducing brackish or saline water. However, many coastal floodplain soils (especially in NSW) have relatively low hydraulic conductivity; therefore, lateral transport of salt water would be restricted to a distance of only several metres from the drain (Johnston & Slavich 2004). Furthermore, ingress of saline water may be limited by the watertable range, local topography and watertable depth (a function of precipitation and evapotranspiration patterns), even though the watertable may be subject to tidal forcing (Johnston, Slavich & Hirst 2005b).

1.6.2 Assessing the success of rehabilitation

Determining the success of wetland rehabilitation is dependent upon the objectives of the rehabilitation project (Kentula 2000). It is therefore important to have frequent and ongoing assessments of rehabilitation efforts to ensure the project is progressing as planned and negative impacts are identified as early as possible. There are a number of techniques for assessing the success of rehabilitation strategies, including reference sites, functional equivalence trajectories and monitoring. Reference wetlands must be carefully selected to ensure similarities to the rehabilitating wetland in terms of hydrogeologic setting, size, geomorphology, tidal range, position in the landscape, adjacent land use and water quality (Neckles et al. 2002). Due to high spatial and temporal variability in natural wetlands, a large number of reference wetlands may be needed to account for this variation (Neckles et al. 2002; Simenstad & Thom 1996). There are three main advantages to using reference sites: (i) data from the reference wetland(s) is used to define the goals of the restored wetland; (ii) the data can then be used as a template for restoring wetlands; and (iii) reference wetlands can be used to establish a framework by which estimations of decline and recovery of functions can be compared (a functional equivalence trajectory) (Brinson & Rheinhardt 1996).

The importance of wetland sites to establish a functional equivalence trajectory is exemplified by a study of the restoration of the Gog-Le-Hi-Te Wetland, Puget Sound, Washington USA. Simenstad and Thom (1996) regularly monitored 16 ecosystem attributes over the first 7 years of restoration, however, many attributes were inconclusive or suggested that they were progressing differently to reference wetlands. They came to the conclusion that it is necessary to understand the natural variability and functioning of reference wetlands before selecting attributes to use for functional equivalency. However, there are often financial and practical constraints to sampling a large number of wetlands and in some regions a suitable reference site may not exist. If no suitable reference sites can be found, it is important to establish long-term monitoring to understand the temporal and spatial variation of the restoration site for directing management efforts (Boyer, Fourqurean & Jones 1997; Maher, Cullen & Norris 1994).

1.7 Wetland hydrodynamic models

The application of wetland hydrodynamic models in restoration projects is an alternative option to reference wetlands and functional equivalence trajectories for determining the successful progress of rehabilitation strategies. A thorough knowledge of the functioning of an individual wetland is required if the model is to be used for developing appropriate management strategies prior to restoration efforts. Results of long-term monitoring and research can be used to develop site specific models (Walton, Chapman & Davis 1996) or calibrate pre-existing models.

Models of wetland hydrology and hydrodynamics can range from simple conceptual models to fully-integrated simulations which require complex data sets. The purpose of the model will determine the type used. Simple conceptual models require only a basic knowledge of the linkages and interactions between components (e.g. Figure 1.3), and are the initial step in the development of more complex models. Conceptual models can range from simple water balance models (Al-Khudhairy et al. 1999) to those incorporating various ecosystem components such as habitat, catchment disturbance, geomorphology and water quality.

Numerical models, like conceptual models, can be simple or complex depending on the purpose. Simple numerical hydrological models include the wetland water budget (Section 1.3.1) and can consist of one equation or a series of equations. More complex numerical models can be used to predict components such as tide height levels within a wetland to ensure that adjacent land is not flooded (e.g. Roman, Garvine & Portnoy 1995), or circulation and mixing to aid in the development of monitoring programs (e.g. Milford & Church 1977). However, one of the major problems with numerical models is that they are spatially restricted and therefore the model needs to be run several times for spatial representation of wetland hydrology (Al-Khudhairy et al. 1999). Furthermore, complex components such as differences in soil horizon properties (and therefore groundwater flow) are often not accounted for in numerical and conceptual models (Al-Khudhairy et al. 1999).

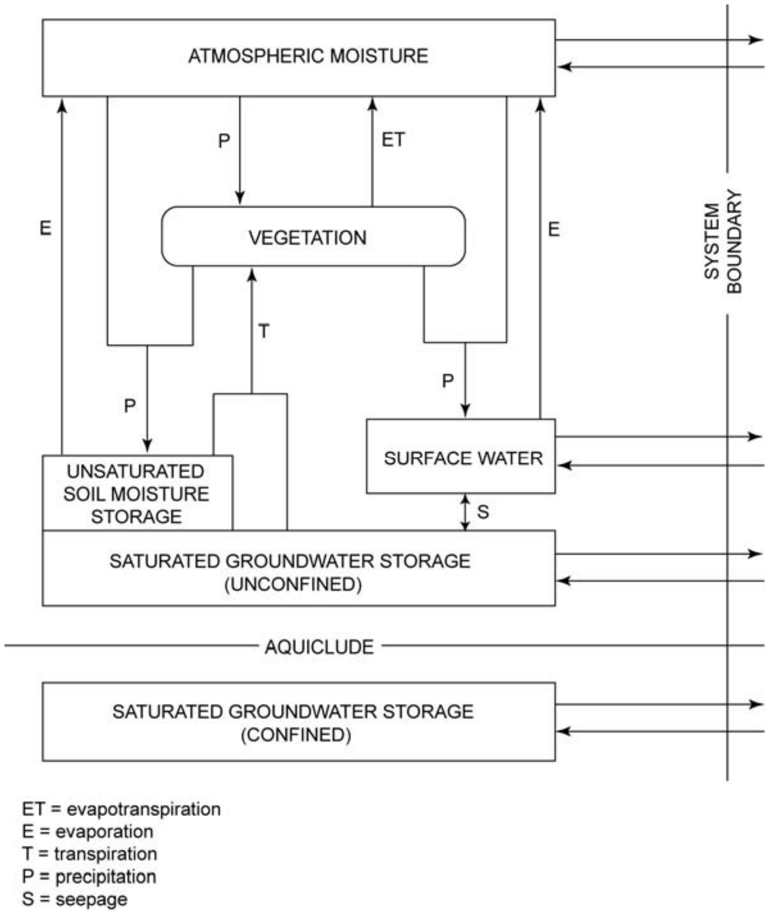

Figure 1.3: Conceptual hydrological model of a freshwater wetland (Duever 1988: 11).

The use of fully-integrated simulation models in complex wetland systems is becoming more common, with a range of computer programs available such as MIKE SHE (e.g. Al-Khudhairy et al. 1999; Refsgaard et al. 1998), Water Quality Mapping and Analysis Program (WQMAP) (e.g. Spaulding, Mendelsohn & Swanson 1999) and Marsh Response to Hydrological Modification (MRHM) (Boumans, Burdick & Dionne 2002). A comprehensive coverage of available hydrological models is provided by Singh (1995). While the use of simulation models provides the ability to predict the effect of different rehabilitation and management strategies (Al-Khudhairy et al. 1999), there is still a lot of uncertainty associated with the predicted outcomes (Restrepo et al. 2006). Much of this uncertainty is due to the lack of data because of the cost associated with obtaining the large and complex data required as input for the model (Johnston, Slavich & Hirst 2005b). The natural complexity of interactions between wetland hydrology and other ecosystem components, which varies between wetlands, also provides a form of uncertainty in the predicted outcomes of these models.

The development of a conceptual model for an individual wetland ensures that there is a thorough understanding of the processes and interactions between hydrological and other ecosystem components such as vegetation, fauna and anthropogenic impacts. While conceptual models do not provide quantitative predictions of the effect of different management strategies (such as numerical or simulation models), their use in wetland rehabilitation allows managers to consider the potential positive and/or negative effects a particular strategy may have. The conceptual model can also be used to create a hazard ranking process which is based on available or easily obtainable information (Johnston, Slavich & Hirst 2005b). This is especially important for projects where funding for collecting large amounts of data and/or developing a model is limited.

Chapter 2
The Study Area

2.1 Introduction

This chapter describes the natural setting of Little Broadwater and the Clarence River, the study site pre- and post-drainage, and the history of flood mitigation in the study area. A description of the structures installed for rehabilitation of the wetland is also given. This information provides vital background for understanding environmental processes at the site and the impact of drainage on the Lower Clarence River floodplain.

2.2 Study site

Little Broadwater is a large coastal floodplain wetland on the Clarence River, NSW North Coast (Figure 2.1). The wetland drains into Sportsmans Creek on the north-western side of the Clarence River. It is approximately 20 km north of Grafton, 40 km upstream of the Clarence River mouth (3.2 km upstream of the confluence of Sportsmans Creek and the Clarence River). The wetland covers an area of approximately 235 ha, with a local catchment of approximately 670 ha (Pressey 1987). The main land use within the Little Broadwater catchment is cattle grazing, with areas of timber, scrub and small crops (Figure 2.2). There is also a golf course and rural residential development to the east of the wetland and a road crosses the northern section, dissecting an area of 63 ha from the main body of the wetland. The majority of Little Broadwater itself is privately owned and used for grazing cattle, with areas of State land in the east and central regions (Figure 2.3). The topography of the wetland is relatively flat, with an average land elevation of -0.1 m AHD (Wilkinson 2003).

Little Broadwater is one of three wetlands that form the Everlasting Swamp Complex – an infilled lagoon system divided into northern and southern sections by Sportsmans Creek. The other two wetlands in the Complex are Imesons Swamp, west of Little Broadwater, and Everlasting Swamp on the southern side of Sportsmans Creek. The Everlasting Swamp Complex, with the exception of a central portion, has been gazetted as State Environmental Planning Policy (SEPP) 14 (Coastal Wetlands) and is listed on the Directory of Important Wetlands in Australia (Environment Australia 2001; Wilkinson 2003).

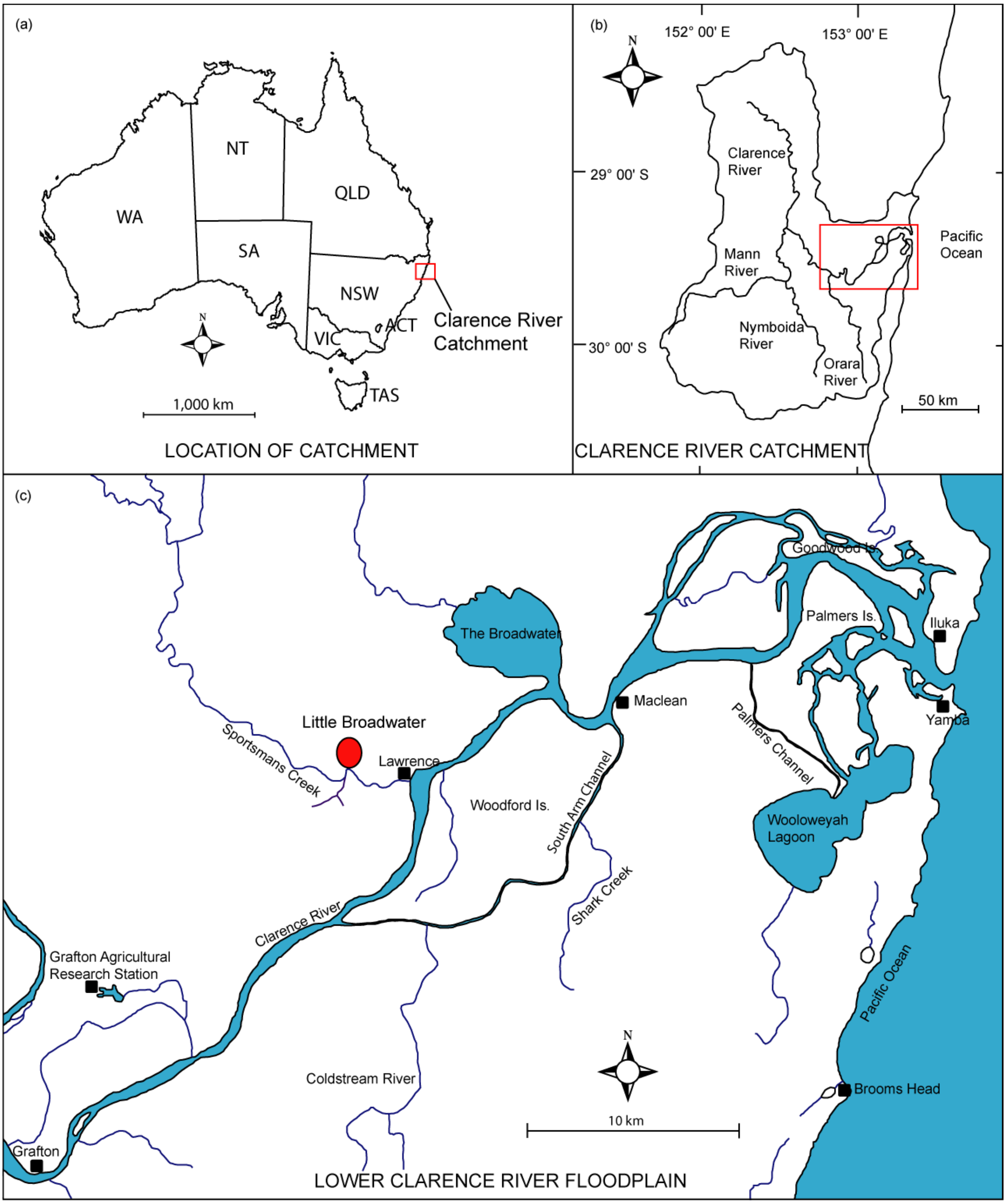

Figure 2.1: (a) Location of Clarence River in NSW, (b) Clarence River catchment and (c) the lower floodplain and location of the study site, Little Broadwater.

Figure 2.2: Quickbird satellite image (May 2004) of Little Broadwater and local catchment showing the location of flood mitigation structures.

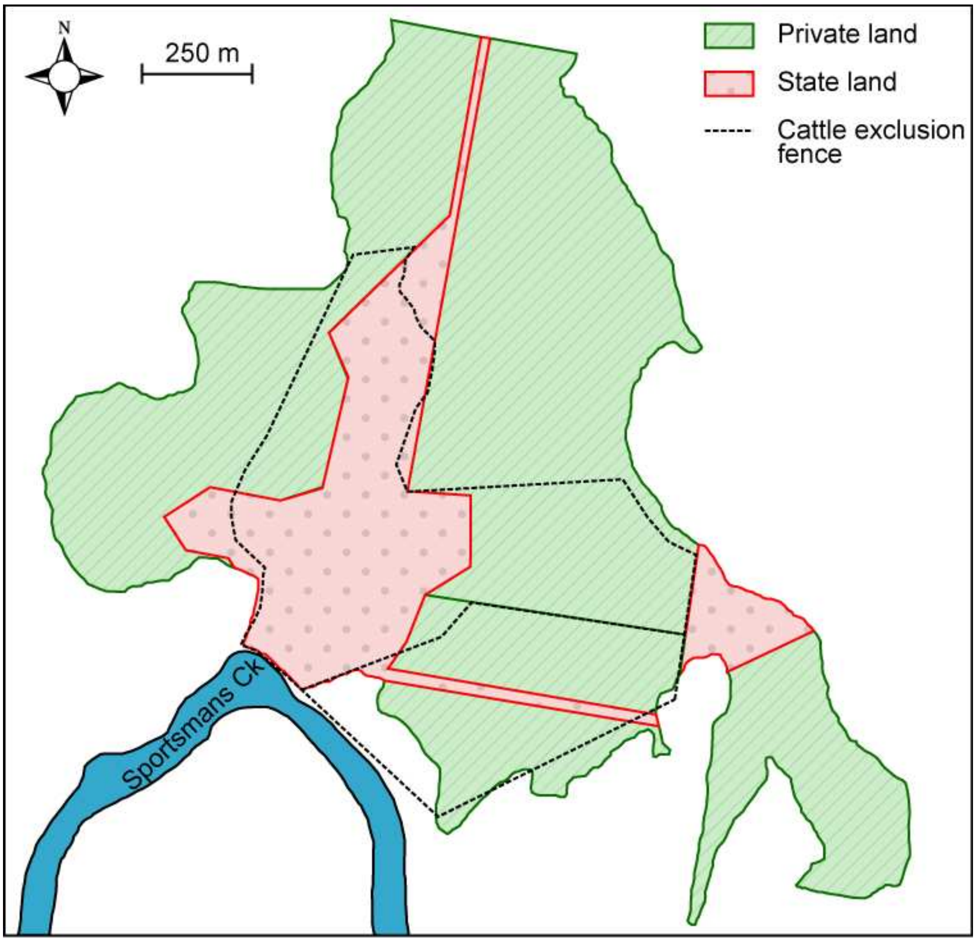

Figure 2.3: The majority of Little Broadwater is privately owned, with the central area and a portion of the eastern side State land (courtesy CVC).

2.2.1 Geomorphology

The Clarence River is located 680 km north of Sydney on the north coast of NSW. With a catchment area of 22 700 km^2 (McSwan & Switzer 2006), main channel length of approximately 250 km, waterway area of 89 km^2 (NSW Department of Natural Resources n.d.) and floodplain area of approximately 500 km^2 (based on maximum flood height; P. Wilson 2008, pers. comm., 22 December) the Clarence River is the largest coastal river in NSW. The main trunk of the Clarence River is located in the southern section of the Clarence-Moreton Basin (Haworth & Ollier 1992), where the Grafton Formation is the major geological landscape and consists predominantly of quartz sandstone (Lin & Melville 1993). The Clarence-Moreton Basin is approximately 450 km long in the north-south direction and 300 km wide in the east-west direction (Ollier & Haworth 1994). The basin is bounded by the South Queensland Block to the north, the New England Block to the west and south, and the Beenleigh Block and the Coast Range to the east (Figure 2.4). The Clarence-Moreton Basin is only open to the coast between Brooms Head and Schnapper Point (Ollier & Haworth 1994).

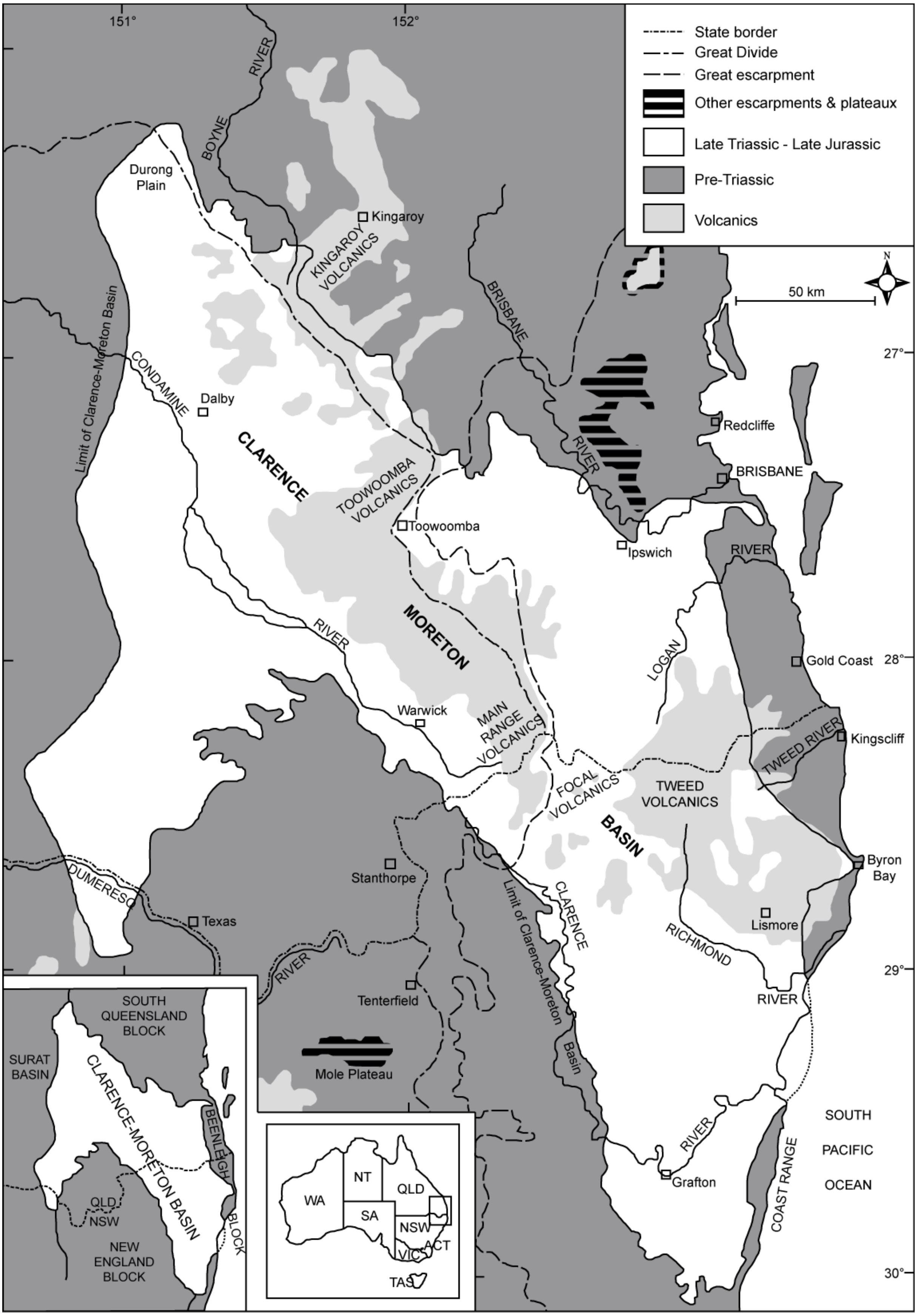

Figure 2.4: Major geomorphic features of the Clarence-Moreton Basin and surrounding area (Ollier & Haworth 1994: 292).

The Clarence-Moreton Basin began forming during the Late Palaeozoic, when part of the generally flat palaeoplain subsided (Haworth & Ollier 1992). Drainage at this time would have been east to west, or southeast to northwest. Terrestrial sediments began to fill the Basin in the Late Triassic Period and deposition continued through the Jurassic and possibly the Cretaceous (Ollier & Haworth 1994). Extensive sedimentation in the Basin ceased approximately 80 million years ago, around the same time that the seafloor began spreading (creating the Tasman Sea). This initiated downwarping of the coast, leading to coastward drainage and coastal erosion of the Clarence-Moreton Basin (Ollier & Haworth 1994).

The Clarence River evolved over a long period of time in parallel with the geological evolution of the region, and does not fit with geomorphic models such as climatic geomorphology, Davisian cycles, or steady-state landforms (Haworth & Ollier 1992). The Clarence River is instead an example of evolutionary geomorphology (Haworth & Ollier 1992). The Clarence River flows predominantly along the Palaeozoic-Jurassic boundary, crossing back and forth occasionally, a course which was established once the Clarence-Moreton Basin had finished filling with Mesozoic sediments (Ollier & Haworth 1994). Drainage was increasingly restricted to the eastern section of the Basin in the Late Cretaceous due to the development of the north-south Continental Divide through the middle of the Basin (Haworth & Ollier 1992). Miocene volcanism in the northern section of the Basin not only further restricted drainage to the south, but also forced the main trunk of the Clarence River closer to the western edge of the Clarence-Moreton Basin, thereby confining the River to its present position on the western edge of the southern arm of the Basin (Haworth & Ollier 1992). The downwarping of the Basin with the creation of the Tasman Sea led to a reversal of the northwest flowing Clarence River between the headwaters and the Orara River, making the section of the Clarence River between the Orara River and the coast an overflow of the recently south-flowing Clarence River and north-flowing Orara River (Ollier & Haworth 1994).

The Clarence River passes through a number of physiographic regions from the coast to Lawrence and Little Broadwater (Figure 2.5). The Bundjalung Dunefield occurs along the coastal strip and contains extensive deposits of Quaternary aeolian and marine sand, dunes, beach ridges and sandsheets. Between this coastal strip and the township of Maclean is the Clarence Delta region. Landform elements include alluvial plains, tidal plains, and swamps and numerous channels. Upstream from Maclean the Clarence River landscape changes from the Clarence Delta to the Clarence Alluvial Floodplain characterised by alluvial plains, abandoned channels, levees and backswamps. Little Broadwater is located in the transition zone between the Clarence Delta and Clarence Alluvial Floodplain, so whilst it is dominated by alluvial processes there are also significant estuarine features (Morand 2001).

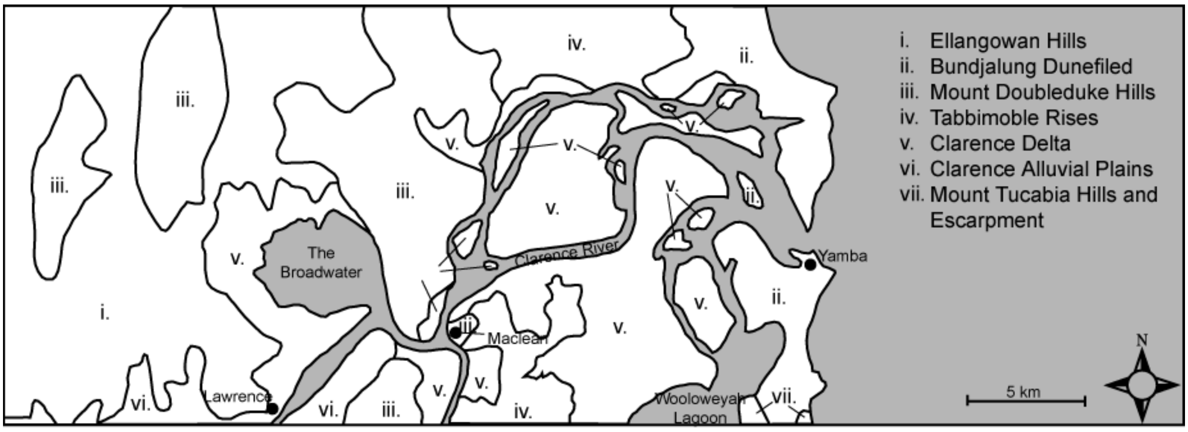

Figure 2.5: Physiographic regions of the Clarence River lower floodplain (adapted from Morand 2001: 2). Little Broadwater is located in the transition zone between the Clarence Delta and the Clarence Alluvial Floodplain.

The Clarence River is classified as a mature barrier estuary. Barrier estuaries have a shore parallel sand barrier that restricts the entrance to the estuary, and are located on broad tidal and backbarrier sand flats, with narrow and elongated entrances and are wave-dominated (Kench 1999; Roy et al. 2001) (Figure 2.6). Mature barrier estuaries are characterised by broad floodplains with backswamps and cut-off bays (Figure 2.6) and also have lower species distribution, diversity and populations of estuarine flora and fauna than younger estuaries (Roy 1984). This is due to steep-sided channels, limiting intertidal zones, and high river discharges that cause rapid salinity and turbidity fluctuations (Roy 1984). As barrier estuaries mature the tidal range is amplified due to increased channelisation of the river (Roy 1984) (Figure 2.6). Salinity gradients also become more distinct as the estuary matures, especially in the lower reaches where side arms can be more saline than the main channel (Roy et al. 2001). In the Clarence River, tidal influence can reach as far as Copmanhurst (105 km upstream) and the average annual discharge from the Clarence River is 3.7 x 10^6 ML yr^{-1} (NSW Department of Natural Resources n.d.).

Since its formation, the Clarence River has been subject to cycles of excavation and infilling due to Quaternary glacial and interglacial changes in sea level. Melting of the continental ice sheets, beginning 17000 years ago, resulted in a sea level rise of 150 m over a period of 10 000 years, in a time known as the Postglacial Marine Transgression (Roy 1984). Since sea level stabilised around 6500 years ago the Clarence River has been infilling (Roy et al. 2001). The saline and reducing conditions in abandoned channels and backswamps on the delta and alluvial floodplain has led to the formation of pyritic sediments (acid sulfate soils) during this infilling period.

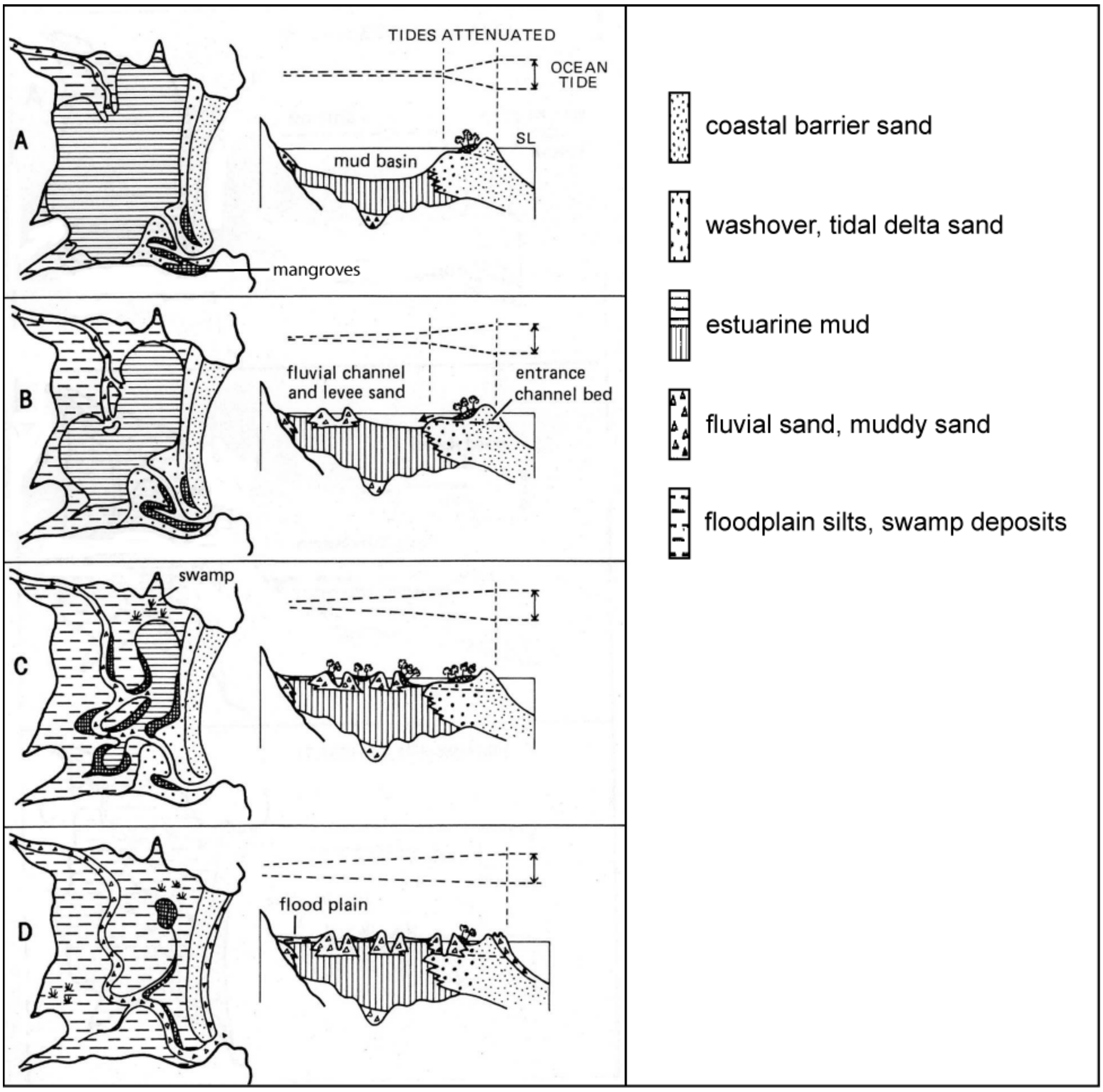

Figure 2.6: Evolution of a barrier estuary. (A) Youthful stage; (B) the estuary begins to infill and becomes shallower; (C) as infilling continues deltas form and embayments begin to be cut-off; (D) the mature estuary is characterised be sinuous channels, smooth levee banks, and broad floodplains (Roy 1984: 112). The Clarence River is at stage (D).

There is an estimated 530 km^2 of high risk ASS underlying the Clarence River floodplain downstream of Grafton (Tulau 1999). The Everlasting Swamp complex was identified as one of seven ASS priority hotspots in NSW by the NSW Department of Land and Water Conservation (DLWC, now Department of Environment and Climate Change NSW). The soils around Little Broadwater are characterised by their strong acidity, high erodibility and low permeability. These soils include the Everlasting (*ev* and *eva*), Lawrence (*la*) and Cowper (*cw*) soil landscapes (Figure 2.7). The *ev* soil landscape is found in the backswamp and is defined as having very poor drainage and a high watertable, is saline and sodic, and has wet bearing strength, high plasticity and low fertility (Morand 2001). The variation *eva*, which incorporates the elevated dry margins of the swamp (Morand 2001), bounds the *ev* soil landscape to the north, northeast and west (Figure 2.7). The south eastern edge of Little Broadwater is bordered by the *la* soil landscape and the southern edge by the *cw* soil landscape. The *la* soil landscape has low permeability, moderate salinity, is strongly acidic and occurs on low hills and overlies the Grafton Formation (sandstone, siltstone, and conglomerate) (Morand 2001). The *cw* soil landscape is alluvial and found in major levees along the Clarence River and is characterised by strong acidity and low permeability (Morand 2001).

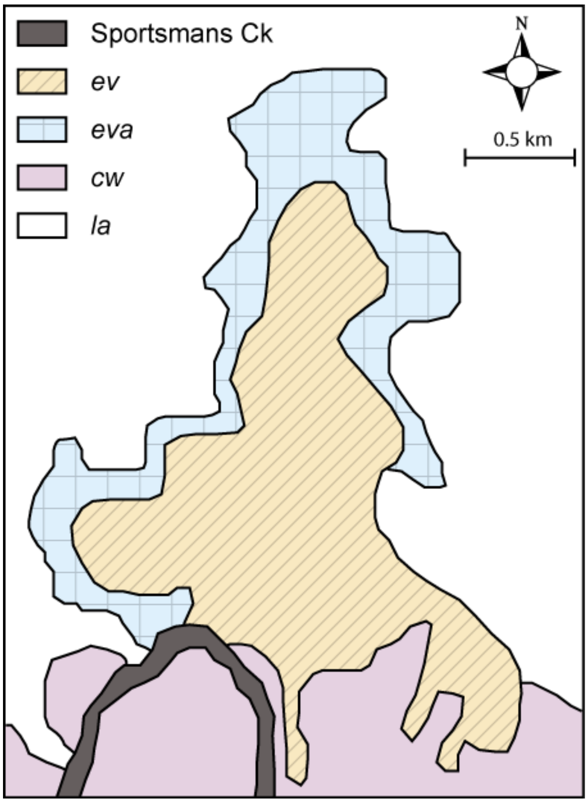

Figure 2.7: Soil Landscapes at Little Broadwater (adapted from Morand 2001). The soils are characterised by their strong acidity, high erodibility and low permeability.

Field investigations of soils in the Everlasting Swamp Complex hotspot by Morand (2002) determined that the predominant soils were classified as Sulfuric/Sulfidic Oxyaquic or Redoxic Hydrosols (Humic Gleys). These soils have 30-50 cm of dark organic-rich clay overlaying wet/saturated soft grey clay, with distinct boundaries between the topsoil and acid sulfate material. The depth to acid sulfate material in the Everlasting Swamp Complex ranged from 20-50 cm with an average depth of 39 cm at Little Broadwater (Morand 2002). Six dominant soil materials were identified across the Everlasting Swamp Complex and the four most common are summarised in Table 2.1. Actual ASS were present at a depth of 20 cm and potential ASS at 60 cm. At the time of the soil investigation by Morand (2002), salt precipitates were noted on the scalded surface and analysis determined they were predominantly magnesium sulfate, calcium sulfate and sodium sulfate (Table 2.2).

Table 2.1: The most common soil materials at the Everlasting Swamp Complex (includes Little Broadwater) (adapted from Morand 2002: 115).

Soil Description	Field pH	Depth (mm)
Fresh alluvium	4.5-5.5	0-50
Dark organic mottled clay	4.5-5.5	50-200
Jarositic unripe clay (actual ASS)	4.0-4.5	200-600
Dark grey unripe clay (potential ASS)	4.5-6.5	> 600

Table 2.2: Results of salt precipitate analysis for Little Broadwater (Morand 2002: 123).

Salt Precipitate	%
Total sodium	1.23
Total potassium	0.30
Total calcium	1.38
Total magnesium	6.84
Total sulfate	27.3
Total carbon	6.78
Total nitrogen	0.49
Total aluminium	2.80
Total iron	1.85
Total manganese	0.14

2.2.2 Historic flood mitigation on the Clarence River floodplain

Prior to European settlement the Clarence River floodplain supported communities of dense sub-tropical rainforest, wetlands and *Casuarina*, *Melaleuca* and *Eucalyptus* forests (Bawden 1979; Farwell 1973 cited in Smith in press). Europeans began harvesting cedar in the mid-

1830s, and clearing of the floodplain increased during the 1840s and 1850s for agriculture (Tulau 1999). In the early 1900s drainage unions were established to co-ordinate drainage networks between landholders, with the belief this would encourage the growth of pasture species and settlement of the region (Pressey 1989).

Early survey plans from 1898 to 1905 indicated that Little Broadwater was previously a large tidal estuarine wetland which formed part of Sportsmans Creek (Wilkinson 2003). Between 1911 and 1927 the wetland was drained and tidal exchange prevented with the construction of the Little Broadwater levee by the Little Broadwater Drainage Trust, for the purposes of agriculture and flood mitigation. A small 900 mm diameter pipe (probably with a one-way floodgate) for discharge was installed at the western end of the levee in approximately the same location as the current floodgate structures. Around 1927 or later a three-cell weir with one-way floodgates (1.22 m x 0.68 m) was constructed at the eastern end of the levee to increase drainage (Wilkinson 2003).

Between the late 1950s to the 1970s flood mitigation schemes on the Clarence River floodplain were rapidly implemented by the Clarence River County Council (CRCC) in response to a series of damaging floods in the 1940s and 1950s (Clarence Valley Council 2006b). These flood mitigation works included the construction of drains, levees and one-way floodgates to prevent flooding of low-lying land and increase drainage of floodwaters (Pressey 1989). Both the Little Broadwater levee and spillway structure were replaced in the 1960s when the CRCC raised the height of the levee to 1.22 m AHD and installed a twin cell (1500 mm) box culvert (Figure 2.8) in the location of the small pipe (the spillway was allegedly buried). The culvert has two 1620 mm one-way floodgates and the drain (Lawrence Whalan’s drain) has a length of 250 m and width of 5 m (Wilkinson 2003).

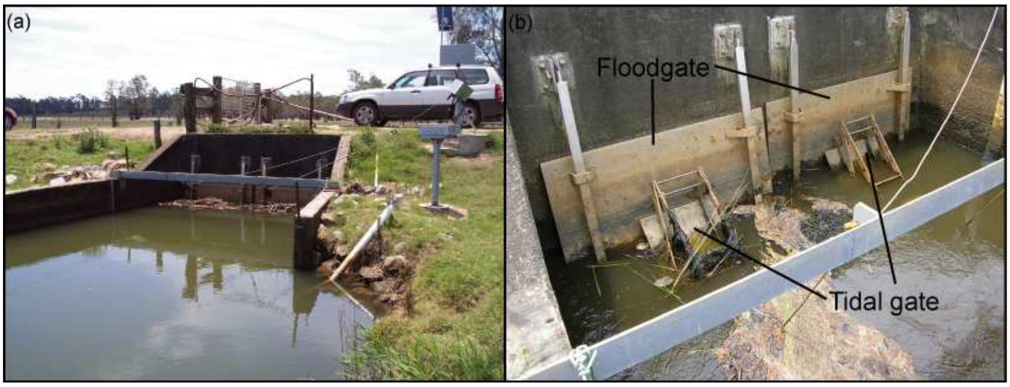

Figure 2.8: (a) Floodgate structure and levee bank built by CRCC in the 1960s (courtesy CVC) and (b) the one-way floodgates were modified with two-way tidal gates in 2003.

Construction of flood mitigation structures on the floodplain by the CRCC continued into the 1980s, with the building of the 16 km long Sportsmans Creek levee from Woody Creek in the north to Blanches Creek in the south to protect crops and improve pasture in Everlasting Swamp (Wilkinson 2003). In total there are 14 major flood mitigation structures in the Everlasting Swamp Complex, consisting of 12 constructed flood mitigation drains with one-way floodgates, and two natural watercourses (Reedy and Woody Creeks) with floodgates (Wilkinson 2003). Along with the 12 major drains, there has also been an extensive amount of private drainage works, which discharge into these larger drains. Presently, there are 110 km of levees, 250 flood mitigation drains and 500 floodgates on the Clarence River floodplain managed by Clarence Valley Council (CVC, previously CRCC) (Clarence Valley Council 2006a), along with numerous other privately managed drains and floodgates (Figure 2.9).

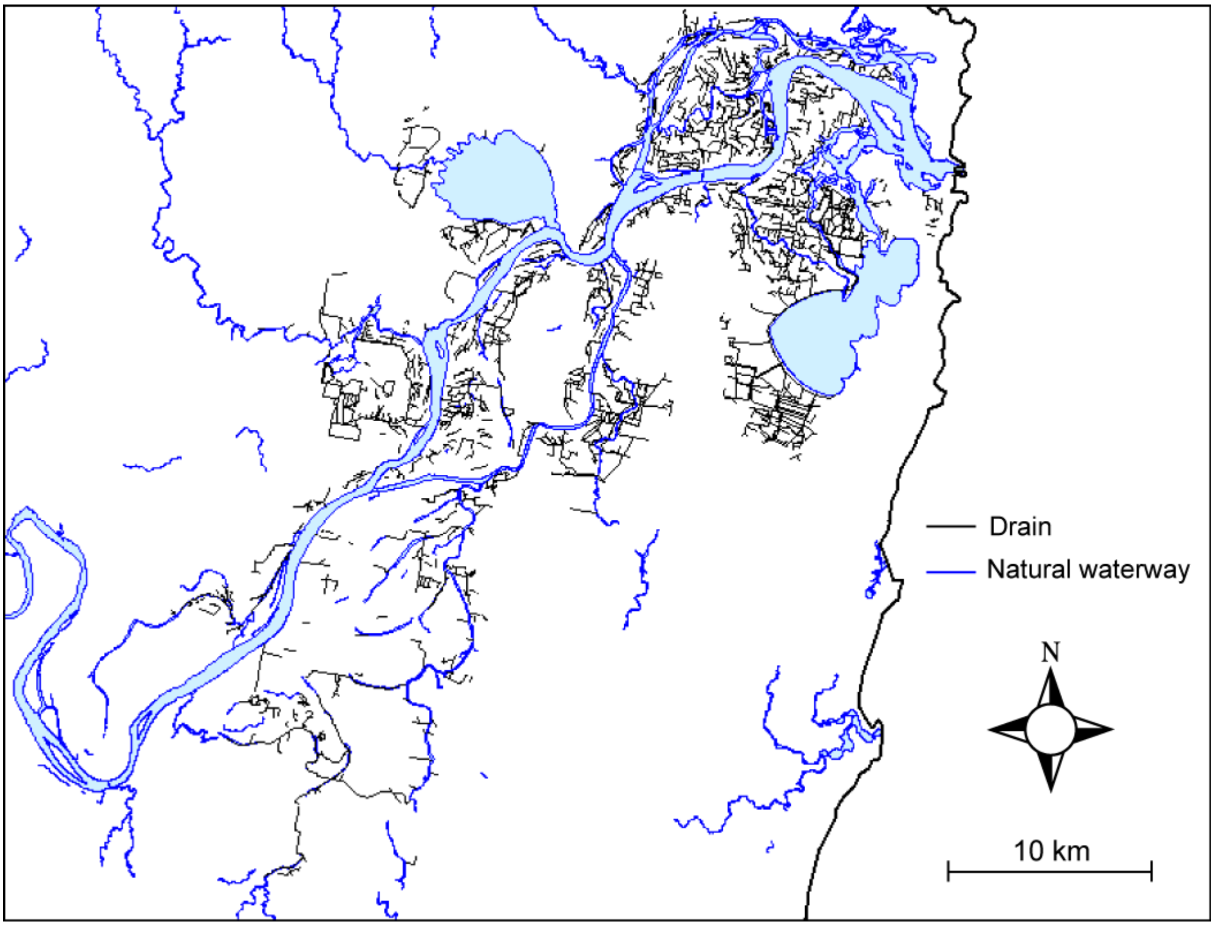

Figure 2.9: The Lower Clarence River floodplain is dissected by a large network of private and public drains (courtesy CVC).

Draining of the floodplain had many negative effects on the environment, including the loss of fish and water bird habitat, oxidation of acid sulfate soils, poor discharge water quality, and in many cases reduced grazing potential (due to acid scalds). After decades of drainage Little

Broadwater was typical of these degraded floodplain wetlands, and in 1997 was identified as a possible rehabilitation site by CVC and other state government departments. In 2000 the Little Broadwater landholders and CVC developed a Drain Management Plan to rehabilitate 172 ha of the wetland by re-flooding and establishing managed tidal exchange. In 2001 continuous monitoring of discharge water quality was established by the DLWC, as part of the Acid Sulfate Soils Hotspots Remediation Program.

The rehabilitation trial of Little Broadwater began in June 2003 with the aim to protect, conserve and promote ecologically sustainable use of the wetland (NSW DPI 2006). The specific objectives of the trial were to: i) protect and enhance surface water and groundwater ecosystems, ecological processes and industries (particularly fisheries); ii) reduce the severity, duration and frequency of acidic discharge; and iii) improve sustainable agriculture of the wetland (NSW DPI 2006). Brackish-saline water exchange was the initial strategy trialled (tidal exchange permitted until Sportsmans Creek reached an EC of 13 dS m^{-1}), however, in early 2006 management was changed to freshwater exchange (tidal exchange permitted until EC reached 3 dS m^{-1} in the creek). The reasons for change in management are discussed in Chapters 4, 5 and 7.

The flood mitigation structures were modified by fitting winches to the floodgates to assist in opening the floodgates to flush the drain. In mid-2003 a concrete weir with removable dropboards and a fish-flap was constructed in the drain (Figure 2.10a, b), and two 1.0 x 1.2 m tidal gates were attached to the one-way floodgates (Figures 2.8b and 2.10c, d). The weir crest causes the water in the backswamp to ‘back-up’ and provides a means of water level control. This control is further refined by dropboards embedded in the weir that may be removed (or replaced) to vary the controlled water level. This also allows rapid lowering and hence drainage in the event of flooding. The fish-flap is a reverse one-way floodgate which is forced open on the rising tide due to head difference between the backswamp and creek (Figure 2.11). Each tidal gate is operated automatically by a float (buoy) (Figure 2.10c, d) which closes the gate as it reaches a pre-determined height on the rising tide. These structures allow for controlled tidal exchange and provide fish access to the wetland while maintaining flood protection. A cattle exclusion fence was also erected to exclude cattle from the centre of the wetland, and was later extended into the southern area of the wetland (Figure 2.3).

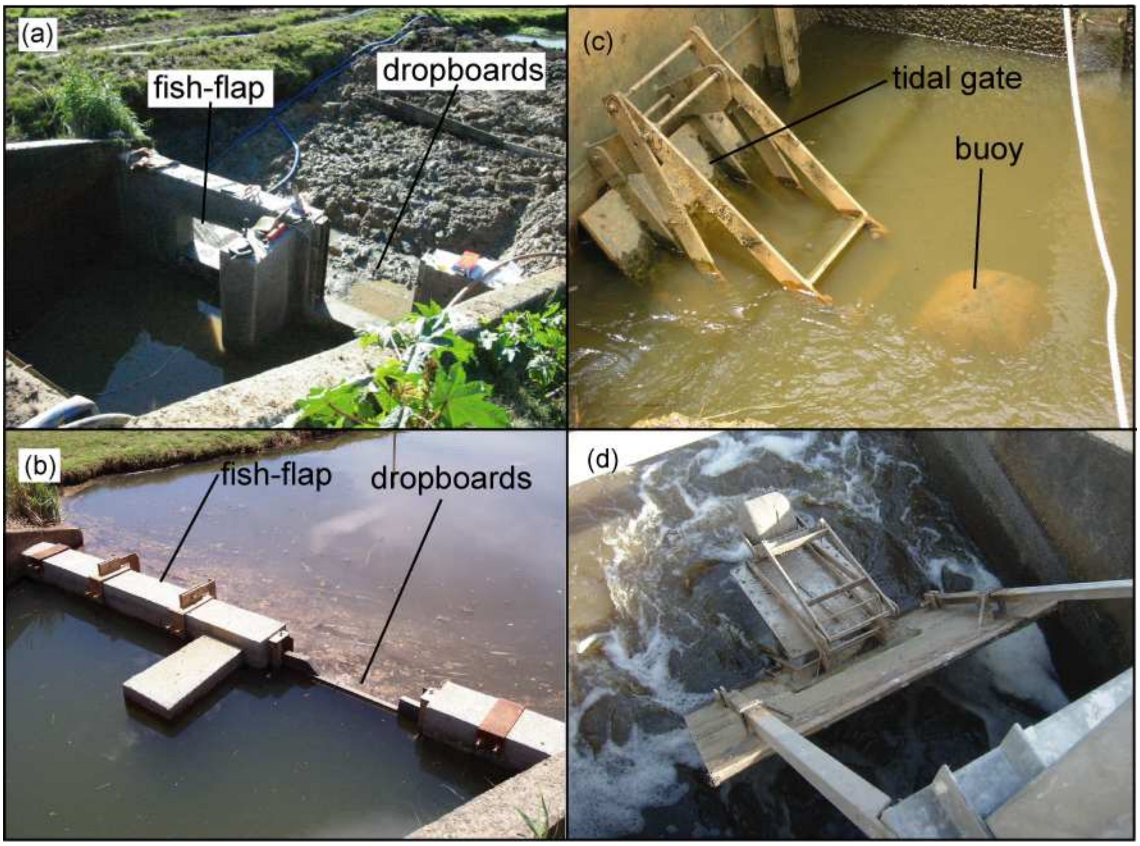

Figure 2.10: Modifications to the original culvert and floodgates for rehabilitation of Little Broadwater. (a) Construction of the in-drain weir, view from the levee bank (courtesy CVC); (b) completed in-drain weir with dropboards and fish-flap in place; (c) the in-drain weir from the bank of the drain; (d) a two-way tidal gate installed on a one-way floodgate, showing the buoy closing device.

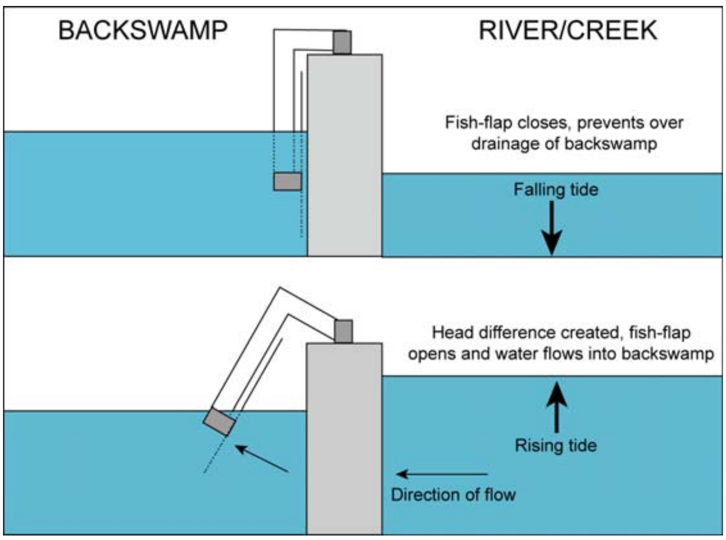

Figure 2.11: The fish-flap at Little Broadwater allows fish and other aquatic fauna access to and from the wetland at high tide (courtesy CVC).

2.2.3 Flora and fauna

Prior to drainage, Little Broadwater was an open tidal wetland, where the central area (approximately 28 ha) was submerged to a depth of 0.5 m on an average high tide with little or no emergent vegetation (Wilkinson 2003). Parish maps from 1898 and 1905 indicate this open area had a mangrove fringe which extended to 930 m from Mantons Road in the north and at least 1.24 km from the current floodgates (Wilkinson 2003). Behind the mangroves was a 5-12 m band of *Casuarina glauca* (swamp oak). The outer wetland was an open reedy swamp which was inundated during spring tides and noted as good grazing country on the 1905 parish map. These open swamps would have generally been fresh to brackish and quite salty during drier periods, however, there is no record of saltmarsh habitat at Little Broadwater (Wilkinson 2003). The Parish maps also indicate that the undulating ground surrounding Little Broadwater was grassy with thick stands of gum oak, ironbark and tea tree.

Draining of Little Broadwater has led to the loss of all but two individual *Avicennia marina* (grey mangrove) trees and the spread of swamp oak throughout the wetland (NSW DPI 2006). There is now a 200-250 m band of thick swamp oak on the western side of the wetland core, and a 150-200 m band of scattered swamp oak fringing the north-eastern side of the wetland core. *Eleocharis* spp. (spikerush) is found throughout the shallow areas, and *Paspalum distichum* (water couch) also occurs in some shallow regions. *Nymphaea caerulea ssp. zanzibarensis* (cape waterlily) often occurs in the core of the wetland where there is relatively deep, open water. Spikerush, water couch and cape waterlily flourish during fresh conditions, and spikerush and cape water lily can also tolerate acidic water. Stands of *Phragmites australis* and *Schoenoplectus litoralis* (river club rush) are dominant to the east of the drain as they can tolerate a range of salinity and water depths. Saline tolerant species such as *Bacopa monniera* (brahmi) and *Cotila coronipifolia* (waterbuttons) occur around the edge of the wetland.

Coastal floodplain wetlands provide important habitat for a diverse range of birds, fish, frogs and reptiles. On the Clarence River floodplain, 97 fauna species are listed on the NSW *Threatened Species Conservation Act 1995*, and 50 of these are birds. The four main threatened wetland bird species are *Ephippiorhynchus asiaticus* (black-necked stork), *Grus rubicundus* (brolga), *Irediparra gallinacean* (comb-crested jacana), and *Anseranas semipalmata* (magpie goose). The black-necked stork and brolga are frequently seen at Little Broadwater, and the comb-crested jacana was recently observed at the wetland. Migratory waders listed under the international Japan-Australia Migratory Agreement (JAMBA) and the

China-Australia Migratory Agreement (CAMBA) have also been recorded at Little Broadwater, specifically *Calidris acuminate* (sharp-tailed sandpiper), *Tringa stagnatilis* (marsh sandpiper), *Tringa nebularia* (common greenshank), and *Gallinago hardwickii* (lathams snipe) (A. Smith 2007, pers. comm., 20 December).

Other water birds which have been observed at Little Broadwater include *Cygnus atratus* (black swan), *Porphyrio porphyrio* (purple swamphen), *Platalea regia* (royal spoonbill), and *Haliaeetus leucogaster* (white-bellied sea-eagle). A number of fish and reptile species have often been seen around the floodgate structures, including *Acanthopagrus australis* (yellowfin bream), *Mugil cephalus* (river mullet), *Anguilla* spp. (freshwater eel) and *Chelodina longicollis* (eastern long-necked turtle).

2.3 Climate

The Clarence River region has a warm temperate climate with a wet summer season and dry winter and spring. Along the coastal fringe the wet season is from November to June, and inland from October to March (Lin & Melville 1993). Average annual rainfall is 1454 mm at the coast (average range at Yamba of 59 mm in September to 182 mm in March) and 1057 mm inland at Grafton (average range of 34 mm in September to 146 mm in February) (Australian Bureau of Meteorology 2007a). The mean annual maximum temperature is 26 °C and the mean annual minimum is 14 °C at Grafton. The coldest months are July to August, with a daily mean maximum temperature of 20-22 °C and minimum of 6-8 °C. The warmest months are from December to March, with a daily maximum temperature of 28-30°C and minimum of 18-20 °C. Frosts may occur several times each year, with a severe frost occurring in early winter 2002 at the study site. Combined with slow regrowth of vegetation from flooding in early 2001, large areas of vegetation at Little Broadwater were killed by the frost (Wilkinson 2003).

2.3.1 Implications for research

Rehabilitation and management of Little Broadwater during this study was dependent on regular flushing and exchange with fresh to brackish water. The salinity of Sportsmans Creek is a function of freshwater inflows from the catchment and tidal forcing. Ultimately, the amount of freshwater inflows is determined by rainfall and evapotranspiration within the local catchment. The salt balance of surface water within the wetland is controlled by tidal exchange, direct rainfall and evapotranspiration.

Groundwater dynamics and management of acid sulfate soils are also affected by climate. Rainfall and evapotranspiration cause fluctuations in the watertable level, and if this level falls below the sulfidic layer then ASS may oxidise. Continuing evapotranspiration can draw acid salts to the surface, which may then be flushed into the wetland and river system by rainfall. Acid products contained within the soil profile may also be mobilised as the watertable is recharged. For ongoing management of Little Broadwater and improvement of water quality and vegetation, it is important to have an understanding of the rainfall and evapotranspiration patterns at the study site.

2.3.2 Weather conditions at the site

Climatic data was sourced from locations as near as possible to the study site through the Australian Bureau of Meteorology (BOM). Rainfall data was sourced from the Lawrence Post Office, approximately 2 km east of the study site. Missing rainfall data was replaced with data collected from Maclean (South Arm) 14 km northeast of the study site, as rainfall patterns were similar to those recorded at Lawrence. Daily evaporation rates were recorded at the Grafton Agricultural Research Station approximately 18 km southwest of the study site. Long-term monthly average daily evaporation rates were from Coffs Harbour, 93 km to the south of the study site. The BOM measures daily evaporation with a Class A evaporation pan,; therefore, the data is representative of open water and bare soil (only when the soil is saturated) with due consideration of the appropriate pan coefficient (Australian Bureau of Meteorology 2007b).

Rainfall patterns differed between years during the pre-modification period (January 2002, when the in-drain water quality logger was installed, to mid-June 2003) and the post-modification period (mid-June 2003 to February 2007), demonstrating the climatic variability of the region. Rainfall records for Lawrence (starting in 1884) indicate that rainfall patterns are highly variable between years and thus differences in monthly rainfall over the pre-and post-modification periods (Figure 2.12a, b) were representative of typical conditions. The only exception to this was during August 2006, when the total monthly rainfall was the highest on record to that date to that date (204 mm, compared to the long-term average of 43 mm). However, total August rainfall of over 150 mm has been recorded in the past and a total of 223 mm fell in 2007 (subsequent to study period), indicating that high rainfall does occur during winter, although not frequently.

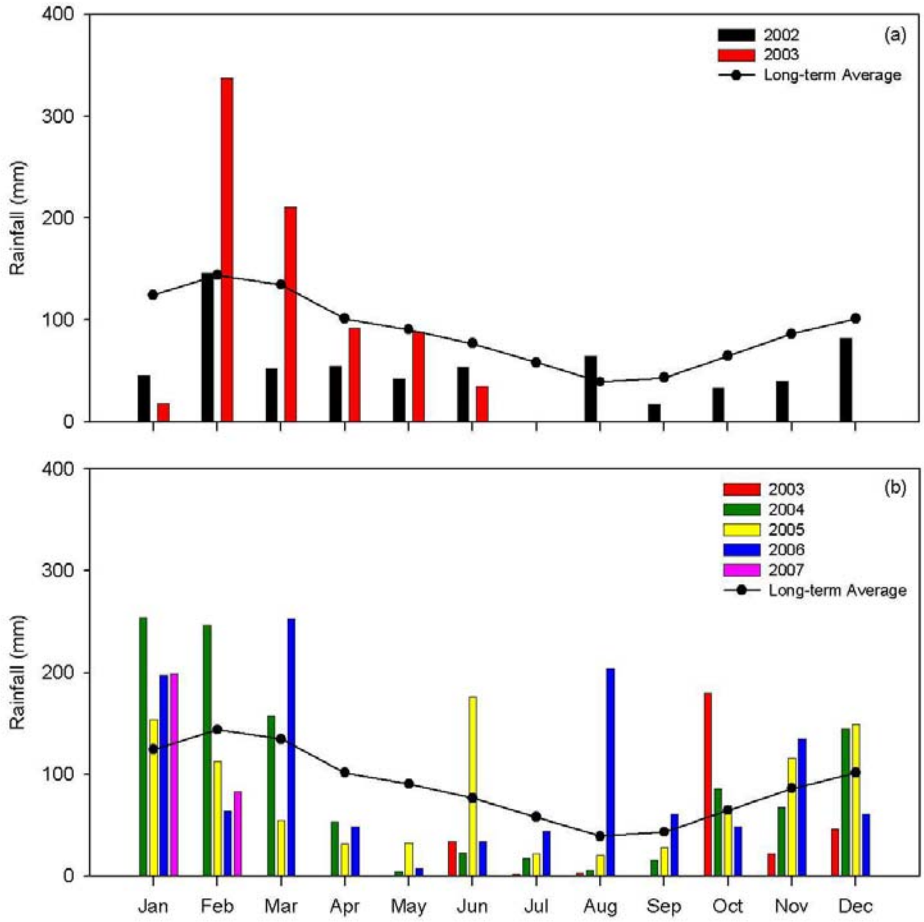

Figure 2.12: Monthly rainfall and long-term average at Lawrence (a) before restoring tidal exchange and (b) post-modification for tidal exchange.

During the pre-modification period, 2002 annual rainfall was much lower than the long-term average (1071 mm), with only 631 mm recorded. Only February and August had similar or greater monthly totals than the long-term average (Figure 2.12a). The 2003 total annual rainfall was similar to the long-term average, with 1035 mm recorded, however, more than 50% of this fell during February and March. A total of 338 mm of rain fell in February 2003, with 53% (179 mm) falling in a 24-hour period (Figures 2.12a and 2.13). Eighty-nine percent of the total February rainfall occurred over a 7 day period. During March 2003, 211 mm of rainfall was recorded which accounted for 20% of the annual rainfall. The majority of the total month's rainfall (59%) occurred over a 4 day period (Figure 2.13).

Annual rainfall during the post-modification period was variable, with 2004 (1077 mm) similar to the long term average, 2005 below average (968 mm) and 2006 above average (1159 mm). During the latter half of 2003 rainfall was well below average, with only 254 mm recorded between July and December. Total monthly rainfall in April and May 2004, 2005 and 2006 was much lower than the long-term average (Figure 2.12b). In contrast, there were a number of months during the post-modification period which accounted for more than 15% of the annual rainfall. These were October 2003, January and February 2004 (24% and 23% respectively), January 2005, June 2005, January 2006, March 2006 (22%), August 2006 and January 2007. Eighty-one percent (199 mm) of the total February 2004 rainfall occurred in a 24-hour period (Figure 2.13).

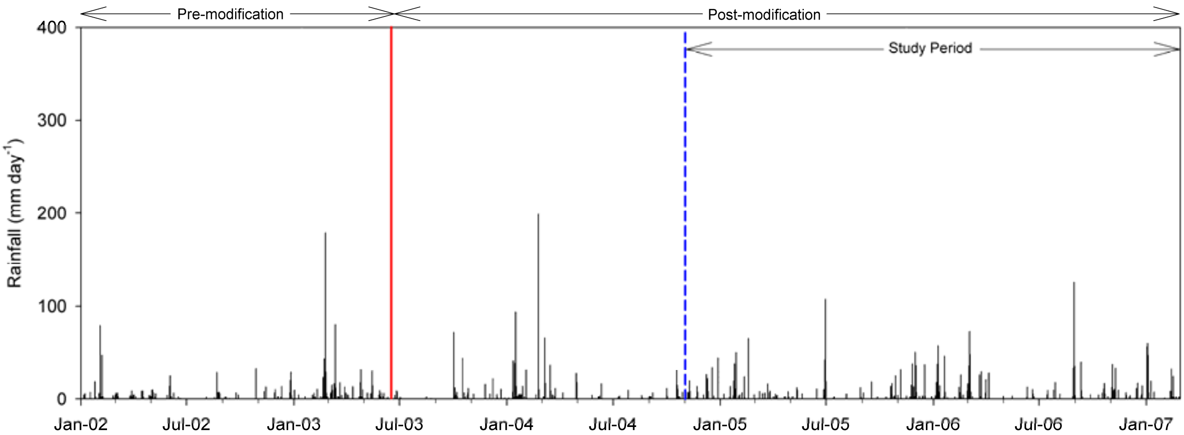

Figure 2.13: Daily rainfall at Lawrence during the pre-modification period (January 2002 to mid-July 2003) and post-modification (mid-July 2003 to February 2007) periods. The study period is also indicated (November 2004 to February 2007).

During the study period (November 2004 to February 2007) there was no 24-hour period with more than 150 mm of rainfall recorded. The majority of rainfall in June 2005 occurred over a 4-day period at the end of the month and accounted for 94% of the total month's rainfall. Similarly, 97% of the total August 2006 rainfall occurred over the last 4 days of the month. In both years the majority of rainfall occurred in a 24-hour period, with 108 mm falling in June 2005 (61% of the total month's rainfall), and 126 mm falling in August 2006 (62% of the total month's rainfall) (Figure 2.13). Both of these rainfall events caused localised flooding and provided unique opportunities to monitor the response of the wetland to fresh conditions during the winter.

Throughout the pre- and post-modification periods there were a number of extended dry phases. Dry phases were defined as a length of three or more weeks with no rainfall recorded. These dry phases were between late autumn to spring, during the driest part of the year (Table 2.3). Dry phases of more than one month occurred over June and August 2002 (44 days), July and August 2003 (43 days), and from August to October 2003 (38 days).

During 2004, prior to the start of the pilot study (Chapter 3), there were five consecutive dry phases of 22-25 days, which were interspersed with periods of low rainfall and short (i.e. less than 21 days) dry phases. The primary implication of these dry phases was the effect on watertable dynamics, the potential for exposure of ASS, and capillary movement of acid salts to the sediment surface, causing scalds (discussed in Chapter 5).

Table 2.3: Dry phases at Little Broadwater pre- and post-modification. Periods of three weeks or more were considered a dry phase.

Number of days without rainfall	Start Date	End Date
44	21 June 2002	3 August 2002
43	4 July 2003	15 August 2003
38	25 August 2003	1 October 2003
22	7 April 2004	28 April 2004
25	1 May 2004	25 May 2004
28	12 June 2004	9 July 2004
22	28 July 2004	18 August 2004
23	8 September 2004	30 September 2004
24	22 May 2005	14 June 2005
23	6 August 2005	28 August 2005
24	7 April 2006	30 April 2006
23	18 May 2006	9 June 2006
22	23 June 2006	14 July 2006
21	7 August 2006	27 August 2006

Evaporation rates at the study site exhibited a seasonal pattern, with high evaporation during summer and low evaporation during winter (Figures 2.14a, b and 2.15). During 2002 the total monthly evaporation was similar to the long-term average, and was greater than 150 mm during summer and late spring (Figure 2.14a). However, in 2003 the total monthly evaporation was generally much lower than the long-term average. During the post-modification period total monthly evaporation rates were much lower than the long-term average and were generally less than 150 mm over summer and late spring (Figure 2.14b). Peak daily evaporation rates during the pre-modification period were approximately 8 mm day^{-1} during summer and 2-3.5 mm day^{-1} during winter (Figure 2.15). In comparison, peak daily evaporation rates during the post-modification period were slightly lower, reaching approximately 7 mm day^{-1} over summer and 2-2.5 mm day^{-1} during winter (Figure 2.15).

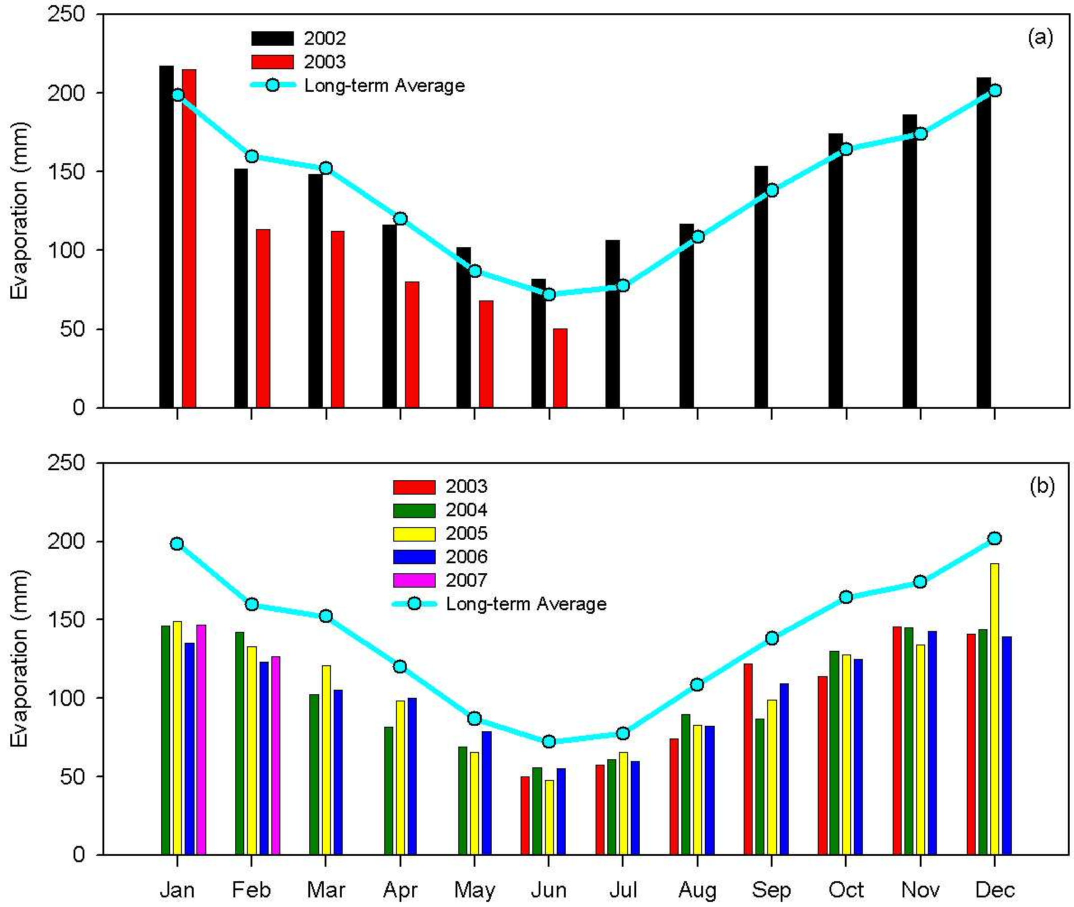

Figure 2.14: Monthly evaporation and long-term average (a) prior to and (b) after restoring tidal exchange. Only 2002 had monthly evaporation similar to the long-term average.

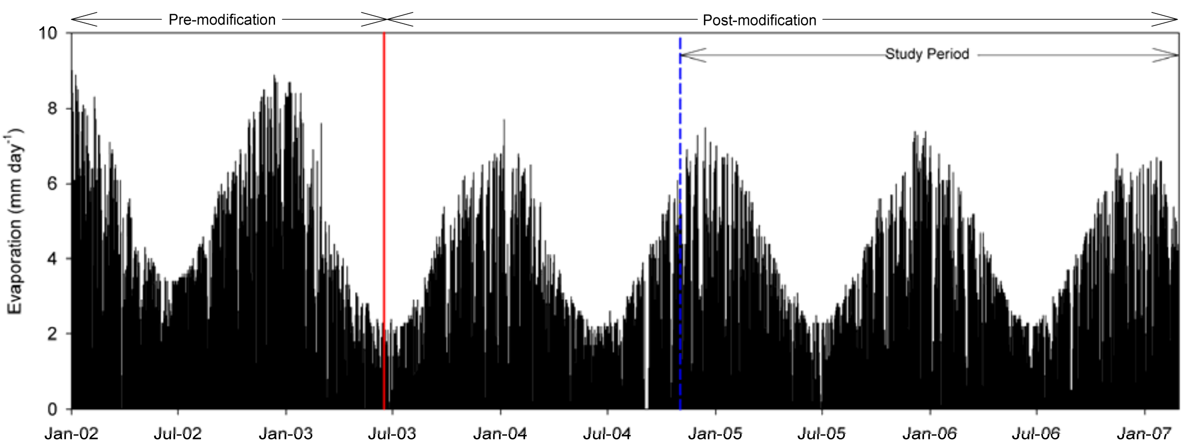

Figure 2.15: Daily evaporation during the pre-modification (January 2002 to mid-July 2003) and post-modification (mid-July 2003 to February 2007) periods. The study period is also indicated (November 2004 to February 2007). Evaporation rates were slightly higher during the pre-modification period.

The seasonal relationship between rainfall and evaporation was generally similar between years during the pre- and post-modification periods, with some exceptions. Total monthly evaporation exceeded total monthly rainfall during winter and spring excluding October 2003, June 2005 and August 2006 (Figure 2.16). In the latter two months total rainfall exceeded evaporation by more than 100 mm. During summer and autumn 2003, 2004, 2006 and 2007 total monthly rainfall often exceeded total monthly evaporation.

The variable climatic conditions during the research period provided an opportunity to study the effect of different tidal exchange management strategies on the hydrology of the wetland. During 2004 and 2005 saline management was trialled, with tidal exchange allowed until the electrical conductivity (EC) of Sportsmans Creek reached approximately 13 dS m^{-1}. The annual rainfall was below average in 2005, with evaporative outputs greatly exceeding rainfall inputs to the wetland with the exception of June (Figure 2.16). In 2006 the management strategy was changed to only allow tidal exchange until EC in the creek reached approximately 3 dS m^{-1}, coinciding with increased annual rainfall and lower annual evaporation. The implications of these changes in the climate and management strategy on surface and groundwater quality and hydrology are discussed in detail in Chapters 3, 4, 5 and 7.

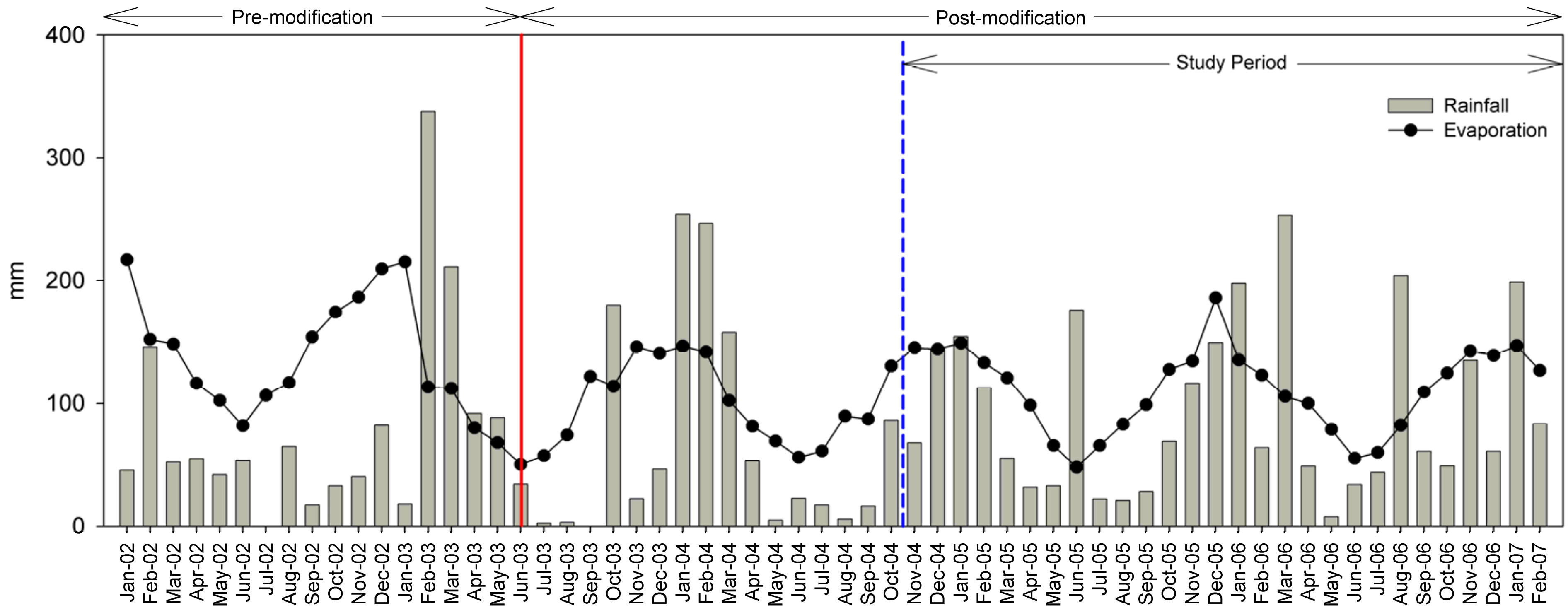

Figure 2.16: Comparison of monthly rainfall and evaporation at the study site pre-modification and post modification. The study period is also indicated.

Chapter 3
A Preliminary Investigation of Water Quality in a Rehabilitated Coastal Wetland

3.1 Introduction

Hydrology is the most important factor in wetland ecosystem functioning as it determines wetland development, vegetation composition and distribution, substrate type and nutrient cycling (Bedford 1996; Bridgham & Richardson 1993; Hughes, Binning & Willgoose 1998). A wetland's hydrology is influenced by micro-topography, tidal regime (in estuarine reaches), seasonal rainfall and evapotranspiration patterns, vegetation, and groundwater flows (Hughes, Binning & Willgoose 1998). The hydrology of many wetlands on the north coast of New South Wales (NSW) has been altered through drainage and prevention of tidal exchange by the construction of levees and floodgates. Many of these wetlands are underlain by acid sulfate soils (ASS) and oxidation of these soils through drainage has resulted in the leaching of heavy metals, acids and salts into the groundwater and surface water (Indraratna, Blunden & Nethery 1999; White et al. 1997).

Rehabilitation of ASS wetlands by restoring tidal exchange and ponding water in the drain and on the floodplain, is common along the east coast of Australia. This prevents further oxidation of ASS and reduces acid discharge (Indraratna, Glamore & Tularam 2002; White et al. 1997), while providing ecological benefits such as increased fish access and encouraging the re-establishment of wetland vegetation (Eertman et al. 2002; Turner & Lewis 1997). Research by Indraratna, Glamore and Tularam (2002) and Johnston, Slavich and Hirst (2005b) has shown that restoring tidal exchange and maintaining high watertables in drained ASS wetlands improves discharge water quality by reducing acid fluxes from groundwater. Maintaining surface water cover further reduces acid fluxes from the system (Johnston, Slavich & Hirst 2004b). While most research has focussed on groundwater and drain water quality, there have been few studies on surface water quality within reflooded ASS wetlands. As a result the characteristics and spatio-temporal variability of surface water quality within rehabilitated backswamp wetlands is poorly understood.

Wetland rehabilitation requires an understanding of water quality, hydrodynamics, interactions between groundwater and surface water, engineering hydraulics, ecology and estuarine dynamics (Glamore & Indraratna 2005). However, the hydrology of sulfidic floodplains is often one of the least understood factors when restoring these ecosystems (White et al. 1997). In Australia there is a lack of environmental data for wetlands, especially relating to their hydrology (Finlayson & Mitchell 1999), making it difficult to predict the ecological response of the system. It is therefore essential to conduct pilot studies to develop an understanding of the underlying temporal and spatial variability of the wetland hydrology (Maher, Cullen & Norris 1994). This information can then be used to modify the management of the wetland and assist in the design of long-term monitoring programs to increase the understanding of the hydrological processes. Environmental monitoring is a management and research tool, and is essential for collecting baseline data and measuring progress towards achieving objectives of a management program (Finlayson & Mitchell 1999). Monitoring programs need to be long-term (i.e. minimum of 5-10 years) to distinguish between natural variation in water quality and changes in response to anthropogenic events. This especially applies to temporal variation, as water quality can vary daily, tidally and seasonally (Anisfeld & Benoit 1997).

Extensive spatial and/or long-term water quality monitoring can result in large datasets, which are often difficult to analyse and interpret (Zhou et al. 2007). Multivariate statistical techniques can be applied to reduce the data set into independent variables and delineate monitoring locations without losing important information (Boyer, Fourqurean & Jones 1997; Zhou et al. 2007). Sometimes termed ‘chemometrics’, techniques such as cluster analysis (CA), principal components analysis (PCA), factor analysis (FA), and discriminant analysis (DA), have been applied to river systems and bays to characterise spatial and temporal differences in water quality, identify sources of pollution, and classify zones of similar influence (e.g. Boyer, Fourqurean & Jones 1997; Singh et al. 2004; Vega et al. 1998; Zhou et al. 2007).

This chapter presents the results of a pilot study to investigate water quality dynamics and characteristics in an estuarine wetland undergoing rehabilitation on the Clarence River floodplain. The study was conducted to develop an understanding of water quality variation to assist the design of a long-term monitoring program. The objectives were to: i) identify the dominant water quality parameters accounting for variation; (ii) assess short-term temporal patterns; and (iii) determine if there was spatial zonation of wetland water quality.

3.2 Materials and methodology

3.2.1 Pre-study wetland condition

The condition of Little Broadwater had deteriorated between May 2004 and the start of the pilot study period in November 2004. This change in condition corresponded with below average winter rainfall and little tidal exchange due to high creek salinity (i.e. tidal exchange was restricted once the salinity of Sportsmans Creek reached 13 dS m^{-1}; see Chapters 2, 4, 5 and 7). A satellite image from May 2004 shows the wetland was covered in a large amount of vegetation (Figure 3.1a) and from observations in the field the majority of this was *Eleocharis equisetina* (spikerush). By November 2004 surface water cover had decreased and much of the vegetation which had been present in May 2004 had disappeared and was decaying in the remaining surface water. The southern region had large areas of exposed sediment with iron and salt crusts on the surface and much of the edge of the wetland was also bare of vegetation (Figure 3.1b).

Figure 3.1: Satellite images of Little Broadwater (a) May 2004 and (b) November 2004, showing the change in condition prior to the start of the study period. The wetland was considerably drier during November 2004 with exposed sediment around the edge.

3.2.2 Field equipment and monitoring

The pilot study was conducted over a four month period from November 2004 to February 2005. The study was designed to provide a rapid assessment of spatial and temporal variation in surface water quality and to determine key water quality descriptors. The first sampling period was during the 2nd and 3rd of November 2004 (Nov04), where 37 samples were collected. The second collection was conducted over four days from the 6th to the 9th of December 2004 (Dec04) with 53 samples collected. The final sample collection was carried out over two days on the 16th and 17th of February 2005 (Feb05), with 56 samples collected. Water samples were collected while wading along a meandering traverse within different habitats or when there was a noticeable change in physical factors such as water temperature or algal cover.

At the time of sample collection, *in situ* measurements of pH, electrical conductivity (EC), and water temperature were taken at each site with a TPS handheld meter. Dissolved oxygen (DO) was also measured *in situ* with a YSI 95 DO meter (model #95D). The probes were placed in a large water sample which was collected at mid-depth in an acid washed one litre beaker, which was triple rinsed with wetland water prior to sample collection. This was done to standardise the method as water depth was often variable (ranging from 0.04 m to 0.77 m) and placing the probe directly into shallow water would disturb the anoxic sediments, thus affecting the water quality reading (i.e. disturbing monosulphidic sediments would reduce the DO; see Chapter 1, Section 1.4). All probes were calibrated before use with standard calibration solutions. Water depth was also recorded at the time of sampling.

At each site one surface water sample was collected from mid-depth in a 125 mL acid washed polyethylene water sample bottle. The bottles and caps were rinsed three times with wetland water prior to sample collection, and were lowered into the water column using a 1.5 m sampling pole. All air was excluded from the bottle before sealing and storing. After collection, samples were stored at 4 ºC and frozen within 24 hours of collection. Samples remained frozen until laboratory analysis.

Laboratory analysis was only conducted on a reduced number of samples which were collected at representative sites – 23 samples from Nov04, 23 samples from Dec04, and 27 samples from Feb05 (one of which was located in Sportsmans Creek) (Figure 3.2). Representative sites were determined through division of the wetland into habitat zones based on observations in the field, and then spatial location within each zone. The samples were analysed for a full range of cations and anions by ion chromatography (IC) and inductive coupled plasma spectrometry with optical emission spectroscopy (ICP-OES) as described below. Each water sample was filtered through 0.45 μm Whatman filter paper in the laboratory to remove particulates. The water was then analysed for soluble bromide (Br^-), chloride (Cl^-), fluoride (F^-), nitrite (NO_2^-), nitrate (NO_3^-), phosphate (PO_4^{3-}) and sulfate (SO_4^{2-}) by IC, and soluble aluminium (Al), arsenic (As), calcium (Ca), cadmium (Cd), cobalt (Co), chromium (Cr), copper (Cu), iron (Fe), potassium (K), magnesium (Mg), manganese (Mn), sodium (Na), nickel (Ni), phosphorus (P), lead (Pb), sulfur (S), selenium (Se) and zinc (Zn) by ICP-OES at the Agricultural and Environmental Analytical Laboratory, University of New England, using standard techniques (Rayment & Higginson 1992).

Continuous monitoring of the *in situ* properties of the drain water was conducted using a Greenspan integrated discharge and water quality monitoring system (Figure 3.2), which was installed and maintained by Greenspan. The data logger was programmed to record at one hour intervals. A DC pump transported water samples from the drain and through the flow chamber where pH, EC, DO and oxygen-reduction potential (ORP) were measured. Additional sensors recording temperature, water level and velocity were located in the drain.

Climatic data was sourced from the Australia Bureau of Meteorology, with rainfall recorded at the Lawrence Post Office approximately 2 km east of the study site. Evaporation rates were measured at the Grafton Agricultural Research Station, approximately 18 km southwest of the study site.

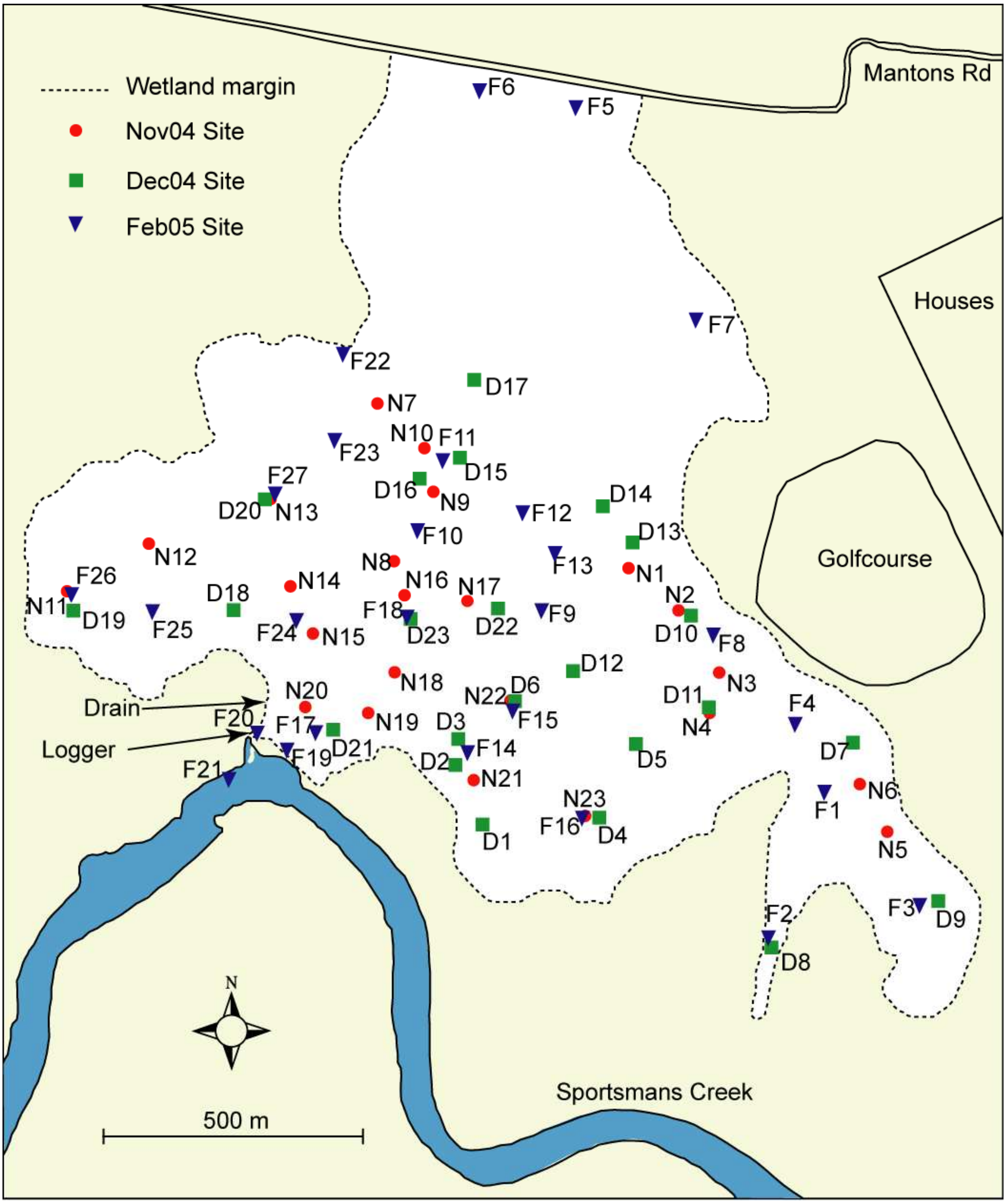

Figure 3.2: Location of analysed sample sites during November 2004 (N1-N23), December 2004 (D1-D23) and February 2005 (F1-F27).

3.2.3 Spatial interpolation of water quality

Surface water Cl^- concentrations and pH were mapped to visualise spatial and temporal variation in the data during the study period. Chloride was used as an indicator of seawater intrusion, as it is naturally present in seawater and is not a product of the dissolution of estuarine clays unlike Ca, K, Mg and Na (Chapter 1, Section 1.5.1). A table containing water quality data and XY coordinates of each sample site, recorded as Eastings and Northings during the monitoring period, was imported into the GIS program ArcMap 9.1. The table was converted into a shapefile with a projected coordinate system (WGS 1984), which geographically depicted the sample sites as points. The wetland boundary was digitised and geographically projected in ArcMap 9.1 using a Quickbird satellite image with projected sample sites as a guide. The surface water boundary was estimated from notes and photos taken in the field.

Chloride concentrations and pH values were interpolated using the Inverse Distance Weighted technique in ArcMap's Spatial Analyst, to predict values at unsampled locations. The wetland boundary, and for Nov04 and Dec04 a surface water boundary line, were used as a barrier to limit the search for input sample points. This resulted in a predicted raster surface of values for Cl^- and pH over the wetland.

3.2.4 Statistical methods

Data were initially tested for normality using the Wilk-Shapiro method, with the majority of datasets being non-normally distributed (skewed or unsymmetrical). This is often a characteristic of water quality data, along with the presence of outliers, seasonal patterns, autocorrelation and dependence on other variables (Helsel & Hirsch 2002). The datasets were unable to be transformed to a normal distribution and as a result nonparametric summary statistics were used in the analysis. Outliers were retained as they reflect the high variability of the environment and nonparametric summary statistics (e.g. median) are not biased by outliers. Temporal variation in water quality was initially tested using Spearman's R coefficient (r_s).

Data were z-score standardised with SPSS 16.0 prior to multivariate analysis. Standardisation makes the data dimensionless by removing the influence of different units of measurement by making the mean of the set of standard scores 1 and the variance 0 (Zar 1999). This avoids misclassification due to the different orders of magnitude of the variance and numerical value of the parameters (Vega et al. 1998). As a result the influence of parameters with a large variance is reduced and the influence of those with a small variance is increased (Singh et al. 2004). The z-score is calculated by subtracting the mean from the value and then dividing by the standard deviation (Zar 1999). Z-score standardisation assumes that the data is normally distributed. However, as previously discussed, the data had a non-normal distribution due to the presence of outliers. Therefore, the application of z-score standardisation to a non-normal distribution was taken into account when interpreting the results.

Multivariate techniques were used to determine spatial and temporal groupings and the factors influencing the clustering. Techniques used were PCA/FA and hierarchical CA. Missing data were replaced with the median of that sample period, as cases were excluded by PCA/FA and CA if there were missing values. Parameters where concentrations were below detectable limits in all the samples had to be excluded from the data set prior to further statistical analysis. Phosphate was excluded from all months and NO_2^-, Cd, Cr, Cu and Pb were excluded from the Feb05 dataset.

Hierarchical agglomerative CA was used to determine both spatial and temporal data clustering. Cluster analysis is an unsupervised pattern recognition technique that clusters similar objects so there is high within-cluster homogeneity and high between-cluster heterogeneity (Shrestha & Kazama 2007). For this study CA was performed with the Ward's method of linkage and the Euclidean distance as the measure of similarity. The Ward's method evaluates the differences between clusters using an analysis of variance approach that attempts to minimise the sum of squares of any two clusters that can be formed at each step (Shrestha & Kazama 2007; Singh et al. 2004). The Euclidean distance is a commonly used distance measure (e.g. Kowalkowski et al. 2006; Shrestha & Kazama 2007; Singh et al. 2004). Cluster analysis was applied to all cases to determine if there was an overall trend for temporal or spatial clustering, and to each sample period to determine spatial zonation with Minitab 13.

Principal components analysis is often used to try and explain the variance of a large number of inter-correlated variables with a much smaller set of independent variables, termed principal components (PCs) (Simeonov et al. 2003; Singh et al. 2004). Principal components are weighted linear combinations of the original variables, with axes in the directions of maximum variance (Shrestha & Kazama 2007). Eigenvalues are the measure of the PCs associated variance, loadings indicate the correlation between PCs and the original variables, and the individual transformed observations are called factor scores (Helena et al. 2000; Vega et al. 1998). The eigenvalue-one criterion is often applied to determine which factors to extract (i.e. only PCs with an eigenvalue greater than one are retained). Principal components identified using this criterion account for the maximum explainable variance (Kowalkowski et al. 2006). Factor analysis follows on from PCA, with the purpose of increasing the contribution of more significant variables and reducing the contribution of less significant variables by rotating the axis defined in PCA (Helena et al. 2000; Shrestha & Kazama 2007; Singh et al. 2004). The new group of variables are called varifactors (VFs) and can include unmeasurable variables (e.g. leakages from sewage systems), whereas PCs are only a linear combination of observable variables (Helena et al. 2000; Shrestha & Kazama 2007; Singh et al. 2004).

Principal components analysis/factor analysis is also a data reduction and grouping tool. The entire data set can be described using only a few PCs/VFs with minimal loss of information (Wunderlin et al. 2001). Varifactors also group the variables based on common features (e.g. source, soluble salts, pollutants) (Kowalkowski et al. 2006; Wunderlin et al. 2001). Principal components analysis of the standardised data was performed on all cases and for each sample period. The PCs were then rotated (varimax rotation) to extract VFs. Varimax rotation minimises the complexity of the VFs by reducing the number of variables with high loadings on each factor (SPSS 2006). Parameters with factor loadings greater than 0.7 were classified as strong and were considered a principal constituent. Principal components analysis/factor analysis was conducted with SPSS 16.0.

The Sportsmans Creek sample from February 2005 (Site F21) was excluded from the descriptive statistics and correlations, as these were specific to analysing within-wetland water quality. However, it was included in the multivariate analysis to identify which sites were similar to the creek water quality.

3.3 Results

3.3.1 Rainfall and evaporation patterns

Annual rainfall in 2004 was consistent with the long-term average, however, nearly 50% of the rainfall occurred in January and February. During the study period, total rainfall was slightly above average and evaporation was below average. Rainfall was highest prior to the Feb05 sample period and lowest before the Dec04 sample period (Chapter 2, Figure 2.13). Evaporation rates varied over the study period, but were generally highest prior to the Dec04 and Feb05 sample periods (Chapter 2, Figure 2.15).

While rainfall over the study period was above average, the antecedent conditions at the site are important in terms of the effects on hydrology. Prior to the start of the pilot study period there was little rainfall with only 11% of the total 2004 rainfall occurring between April and September (71% less than the long-term average over the same period). This six month period was characterised by five consecutive dry phases of 21-25 days, which were interspersed with low rainfall and short dry phases (Chapter 2, Table 2.3). Total monthly rainfall in October was above average, however, it still only accounted for 8% of the annual rainfall.

3.3.2 Temporal variation in water quality

Temporal variation in water quality was initially assessed by comparing the monthly median and range of each parameter (Table 3.1), and determining the r_s. Median water temperature increased over the study period from 24.3 ºC to 31.4 ºC, and was significantly correlated with time ($r_s = 0.42$, $p < 0.001$). Dissolved oxygen ranged from 0.8 mg L^{-1} to 18.6 mg L^{-1} over the study period (Table 3.1) and had a median of 5.9 mg L^{-1}.

Median monthly pH increased between Nov04 and Dec04 (7.46 to 8.29) and then decreased again to 7.55 by Feb05 (Table 3.1). Since pH is a measured as a log scale, a change of one pH unit actually represents a 10-fold change in H^+. The minimum pH during each sample month was below 4 (highly acidic), although there was some temporal variation. Between Nov04 and Dec04 the minimum pH decreased from 2.57 to 2.48, and then increased to 3.08 by Feb05.

The main metals associated with ASS oxidation (Al, Fe and Mn) also had slight temporal variation, although median monthly concentrations were low. Al had the least temporal variation, with monthly medians only ranging between 0.01 and 0.02 mg L^{-1} (Table 3.1). Median monthly Fe increased from 0.09 to 0.39 mg L^{-1} between Nov04 and Dec04, and then decreased to 0.19 mg L^{-1} in Feb-05 (Table 3.1). The maximum concentration of Al increased between Nov04 and Dec04, from 7.58 to 9.06 mg L^{-1}, whereas Fe decreased from 86.05 mg L^{-1} to 72.95 mg L^{-1}. Maximum concentrations of both metals decreased considerably between Dec04 and Feb05, to 2.38 mg Al L^{-1} and 11.33 mg Fe L^{-1}. While the median concentration of Mn increased over the study period and was moderately correlated with time ($r_s = 0.52$, $p < 0.001$), the maximum concentration recorded in each sample period decreased (Table 3.1). The median monthly concentration of other metals (As, Cd, Co, Cr, Cu, Ni, Pb, Sb, Se and Zn) was less than 0.01 mg L^{-1}, with the exception of Se which had a median of 0.017 mg L^{-1} in Dec04 (Table 3.1).

Table 3.1: Median (M) and range of water quality parameters measured at Little Broadwater.

	Nov04 (n = 23)		Dec04 (n = 23)		Feb05 (n = 26)	
Parameter	M	Range	M	Range	M	Range
Water temp. °C	24.3	20.8 – 32.9	26.4	22.7 – 31.3	31.4	22.2 – 42.9
EC dS cm^{-1}	29.1	20.5 – 51.0	21.8	13.0 – 46.5	9.6	2.6 – 19.4
pH	7.46	2.57 – 9.11	8.29	2.48 – 8.72	7.34	3.08 – 8.02
DO mg L^{-1}	5.0	0.8 – 18.6	4.2	2.5 – 8.5	8.5	2.6 – 16.9
F^- mg L^{-1}	1.01	0.00 – 134.32	0.00	0.00 – 35.86	0.00	0.00 – 0.06
Cl^- mg L^{-1}	8288	4203 – 20321	4361	1116 – 18614	2497	481 – 6533
NO_2^- mg L^{-1}	0.000	0.0 – 34.3	0.000	0.0 – 10.6	0.000	0.0 – 0.0
Br^- mg L^{-1}	573	234 – 6948	200	0 – 4394	231	0 – 1476
SO_4^{2-} mg L^{-1}	33.3	18.3 – 62.2	17.6	3.2 – 96.9	12.1	2.1 – 23.0
NO_3^- mg L^{-1}	0.000	0.000 – 0.784	0.238	0.000 – 0.628	0.082	0.000 – 0.578
PO_4^{3-} mg L^{-1}	0.00	0.00 – 0.00	0.00	0.00 – 0.00	0.00	0.00 – 0.00
Al mg L^{-1}	0.007	0.001 – 7.579	0.028	0.011 – 9.056	0.022	0.004 – 2.378
As mg L^{-1}	0.000	0.000 – 0.006	0.003	0.000 – 0.052	0.000	0.000 – 0.040
Ca mg L^{-1}	17.5	4.1 – 453.0	66.3	36.2 – 411.4	59.5	15.8 – 184.0
Cd mg L^{-1}	0.000	0.00 – 0.002	0.000	0.000 – 0.002	0.000	0.000 – 0.000
Co mg L^{-1}	0.001	0.00 – 0.200	0.003	0.000 – 0.254	0.003	0.000 – 0.032
Cr mg L^{-1}	0.000	0.00 – 0.004	0.000	0.000 – 0.003	0.000	0.000 – 0.001
Cu mg L^{-1}	0.000	0.00 – 0.004	0.000	0.000 – 0.004	0.000	0.000 – 0.005
Fe mg L^{-1}	0.087	0.008 – 86.05	0.394	0.114 – 72.65	0.193	0.000 – 11.33
K mg L^{-1}	17.4	3.5 – 273.1	68.4	36.5 – 224.2	36.2	10.7 – 76.5
Mg mg L^{-1}	35.3	9.6 – 340.9	202.9	122.4 – 358.0	153.8	37.9 – 271.2
Mn mg L^{-1}	0.104	0.015 – 44.34	0.358	0.071 – 49.83	1.057	0.343 – 7.82
Na mg L^{-1}	593	107 – 8733	2258	1176 – 8589	1461	279 – 2946
Ni mg L^{-1}	0.002	0.000 – 0.130	0.004	0.000 – 0.116	0.007	0.000 – 0.024
P mg L^{-1}	0.030	0.000 – 2.100	0.122	0.013 – 1.360	0.030	0.000 – 0.122
Pb mg L^{-1}	0.000	0.000 – 0.011	0.000	0.000 – 0.023	0.000	0.000 – 0.004
S mg L^{-1}	14.2	0.0 – 1908.7	95.3	41.3 – 1184.4	83.3	29.7 – 296.8
Sb mg L^{-1}	0.000	0.00 – 0.018	0.004	0.000 – 0.016	0.000	0.000 – 0.022
Se mg L^{-1}	0.002	0.00 – 0.043	0.017	0.000 – 0.081	0.000	0.000 – 0.052
Zn mg L^{-1}	0.000	0.00 – 0.204	0.000	0.000 – 0.164	0.000	0.000 – 0.159

Within-wetland EC decreased from a median of 29.1 dS m^{-1} in Nov04 to 9.6 dS m^{-1} in Feb05 (Table 3.1). Electrical conductivity of water within the drain followed a similar pattern, decreasing from 20 dS m^{-1} in Nov04 to 2 dS m^{-1} in Feb05 (Figure 3.3a). The decrease in wetland and drain EC corresponded with increasing rainfall (Figure 3.3b). Median concentrations of Cl^- and SO_4^{2-} were also highest during Nov04 and lowest during Feb05 (Table 3.1). Electrical conductivity, Cl^- and SO_4^{2-} were strongly correlated with time (r_s = -0.82, -0.72 and -0.75, respectively) and were very significant ($p < 0.001$). Basic cations associated with salinity (Ca, K, Mg, Na and S) did not exhibit a similar pattern to EC, Cl^- or SO_4^{2-}; rather, median concentrations increased between Nov04 and Dec04 and then decreased by Feb05, but were still at least twice the median concentration of Nov04 (Table 3.1). Scatter plots and correlations between EC and salts (Cl^-, SO_4^{2-}, Ca, K, Mg and Na) indicated that some sites during Nov04 had low concentrations of Ca, K, Mg and Na in relation to EC (Figure 3.4).

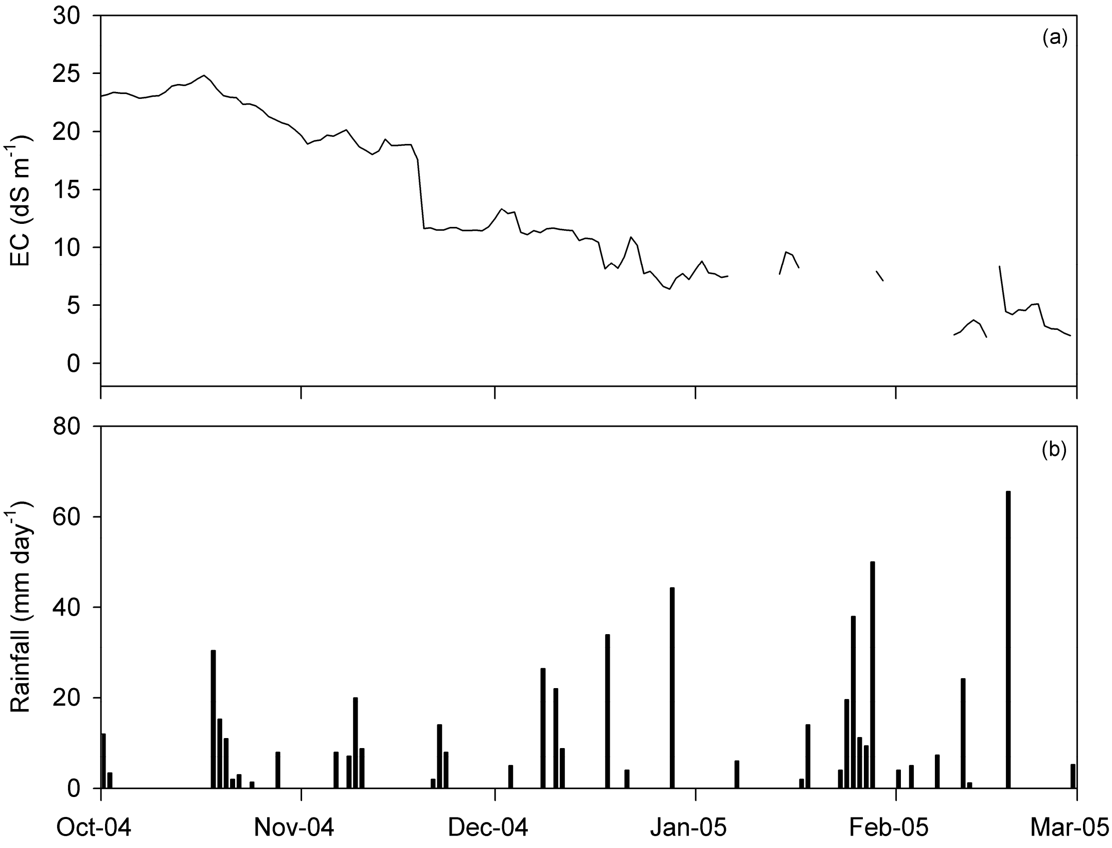

Figure 3.3: (a) Daily mean drain EC decreased over the study period as (b) daily rainfall increased.

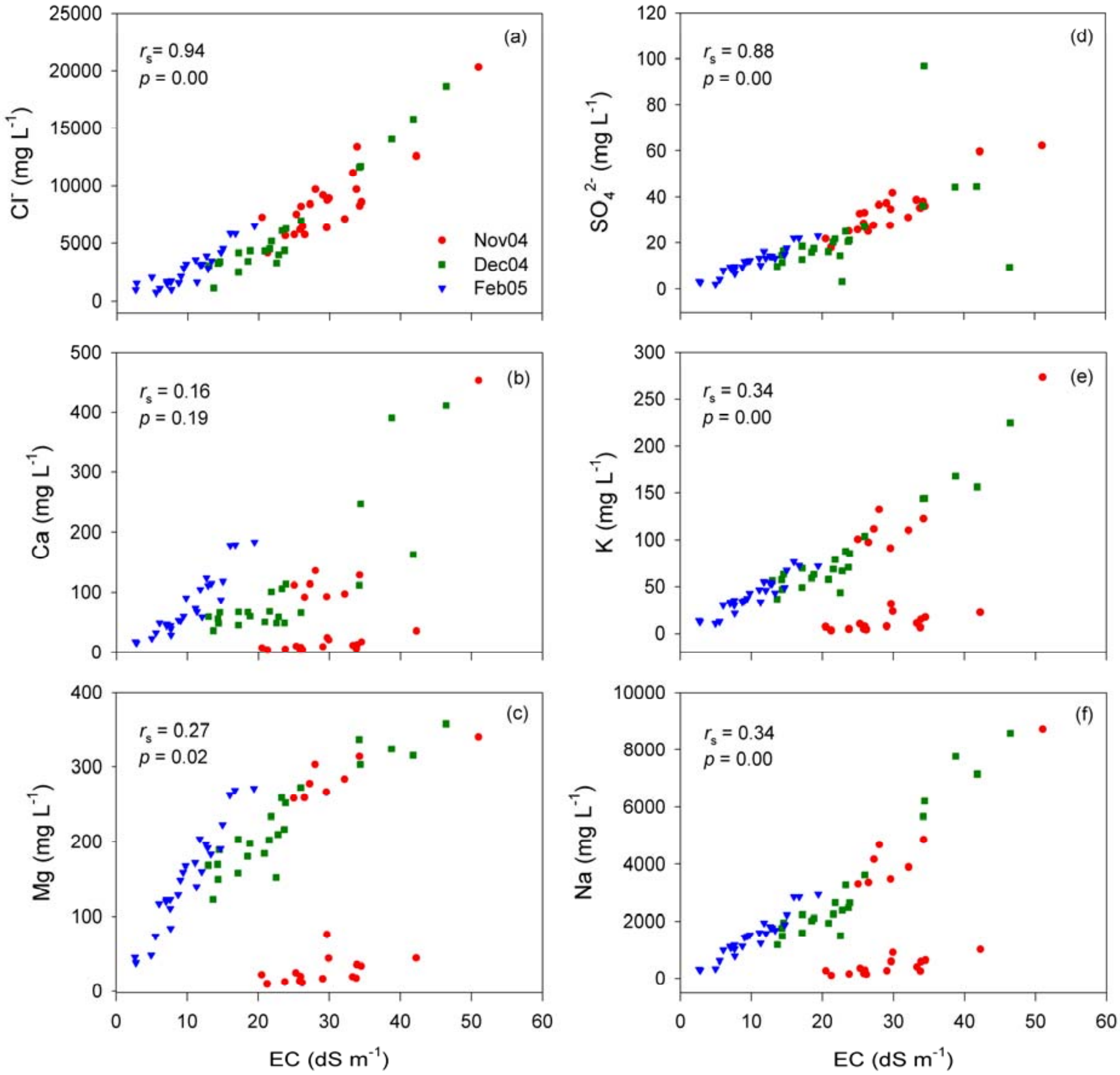

Figure 3.4: The relationship between EC and (a) Cl^-, (b) Ca, (c) Mg, (d) SO_4^{2-}, (e) K and (f) Na. During Nov04 a number of sites had low concentrations of Ca, K, Mg and Na in relation to EC.

Median monthly concentrations of nitrate and phosphorus were highest in Dec04 (Table 3.1). While nitrite had monthly median concentrations of 0.00 mg L^{-1}, monthly maximum concentrations decreased from 34.28 mg L^{-1} to 0.00 mg L^{-1} between Nov04 and Feb05. No phosphate was detected in any of the samples.

3.3.3 Spatial patterns of water quality

Surface water quality was highly variable spatially throughout the study period, especially in terms of water temperature, DO, EC and associated ions, pH and concentrations of Al, Fe and Mn (Table 3.1). Maximum water temperature ranged from 20 °C to 33 °C in Nov04 and

Dec04, and reached a maximum of 42.9 °C in Feb05. Dissolved oxygen concentrations less than 5 mg L^{-1} were regularly recorded in the wetland, although concentrations of up to 18.6 mg L^{-1} were also measured.

Salinity, measured as EC and associated ions (Cl^-, SO_4^{2-}, Ca, K, Mg, and Na), had considerable spatial variation within each sample period. Nov04 and Dec04 had the widest range of EC, varying by over 30 dS cm^{-1} (Table 3.1). Areas of the wetland had EC higher than 40 dS cm^{-1} during both Nov04 and Dec04. Surface water EC in Feb05 ranged between 2.6 dS cm^{-1} and 19.4 dS cm^{-1}. Sulfate, Ca, K, Mg, Na and S also had large ranges in Nov04 and Dec04, with the smallest range in Feb05 (Table 3.1). Concentrations of Ca, K, Mg and Na at sites along the eastern edge of the wetland and in the western and central regions (N2-N16) were low in relation to EC (Figure 3.4). Chloride concentrations were consistently highest in the southern and southeastern regions of Little Broadwater (Figure 3.5). The drain area in the central southern region had low concentration of Cl^- each sample period, as did the northern region during Feb05.

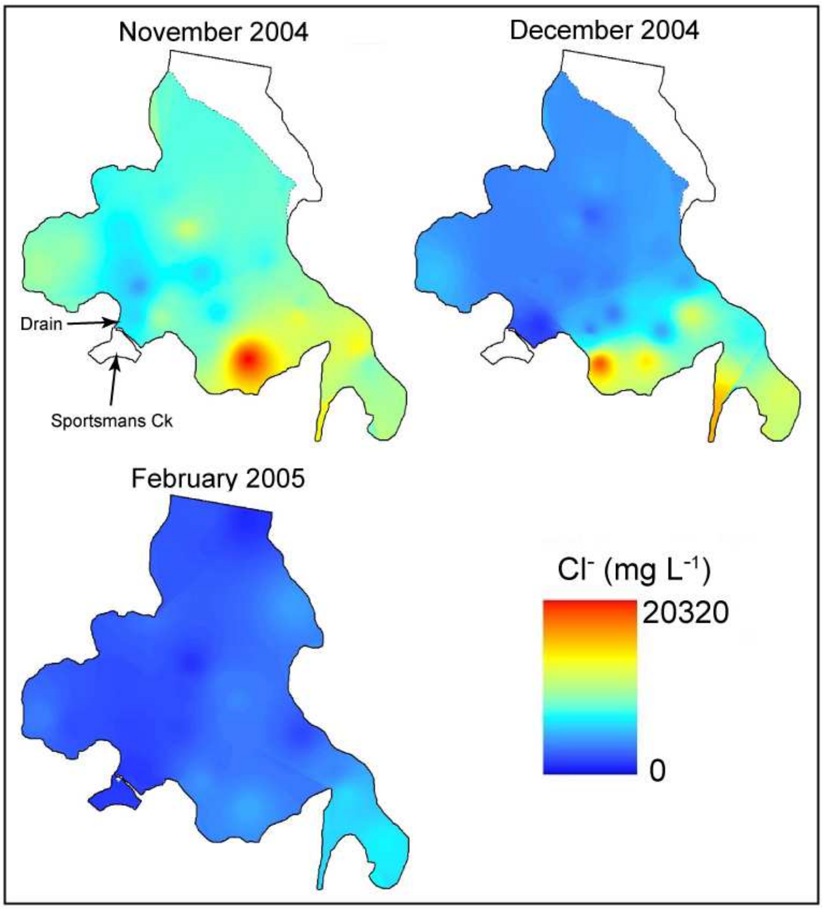

Figure 3.5: Surface water Cl^- concentrations. The southern and southeastern regions were consistently the most saline.

pH had high spatial variability in each sample period, ranging between 2.45 and 9.15 (Table 3.1). The lowest pH was consistently measured in the southern region and during Feb05 acidic surface water was also recorded in the northern area (Figure 3.6). The remainder of the wetland was neutral to alkaline during each sample period. Concentrations of Al, Fe and Mn also exhibited considerable spatial variability, with the highest concentrations corresponding to sites with highly acidic surface water. Other metals such as Co, Cu, Ni and Zn were also highest at these locations, although the overall spatial variability of these parameters and other metals was very low (Table 3.1).

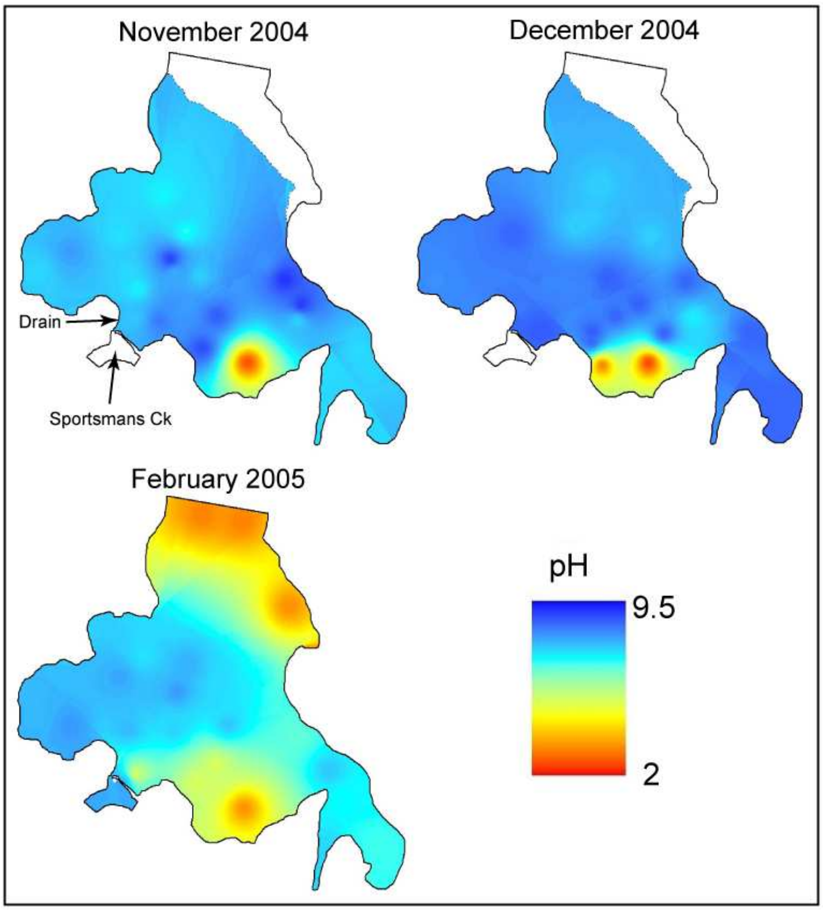

Figure 3.6: Surface water pH at Little Broadwater. The southern region was very acidic each sample period, as was the northern area during Feb05.

Nutrient concentrations were generally highest on the eastern side of the wetland. Although nitrite concentrations had a large range during Nov04 and Dec04 (Table 3.1), nitrite was not detected in the majority of samples and the higher concentrations (> 1 mg L^{-1}) were only detected at five sites – N1, N13, N18, D7 and D12.

3.3.4 Multivariate spatio-temporal patterns of water quality

Cluster analysis of all cases was used to evaluate spatio-temporal patterns during the study period. The resultant dendrogram divided the 73 cases into four clusters at ($D_{\text{link}}/D_{\text{max}}$) x 100 < 35 (Figure 3.7). The three smaller clusters (Groups A, B and C) contained samples from Nov04 and Dec04 only. The sites in Group A were from the southern area of the wetland and had very low pH (< 3.5), high EC and high concentrations of Al, Cr, Fe, Mn, Ni and Zn. Group B consisted of sites from Nov04 which had low ratios of Ca, K, Mg and Na to EC. Group C was characterised by neutral pH and moderate EC, Cl^- and Na, with sites located mainly in the eastern area of the wetland. Group D, the largest cluster, accounted for 44 of the 73 cases, with 61.4% of the group Feb05 samples. The remainder of Group D consisted of samples from Dec04 and one sample from Nov-04 (N20, located near the drain). Group D was characterised by sites with the lowest EC, Cl^- and Na concentrations and pH ranging from 3.08 to 8.72.

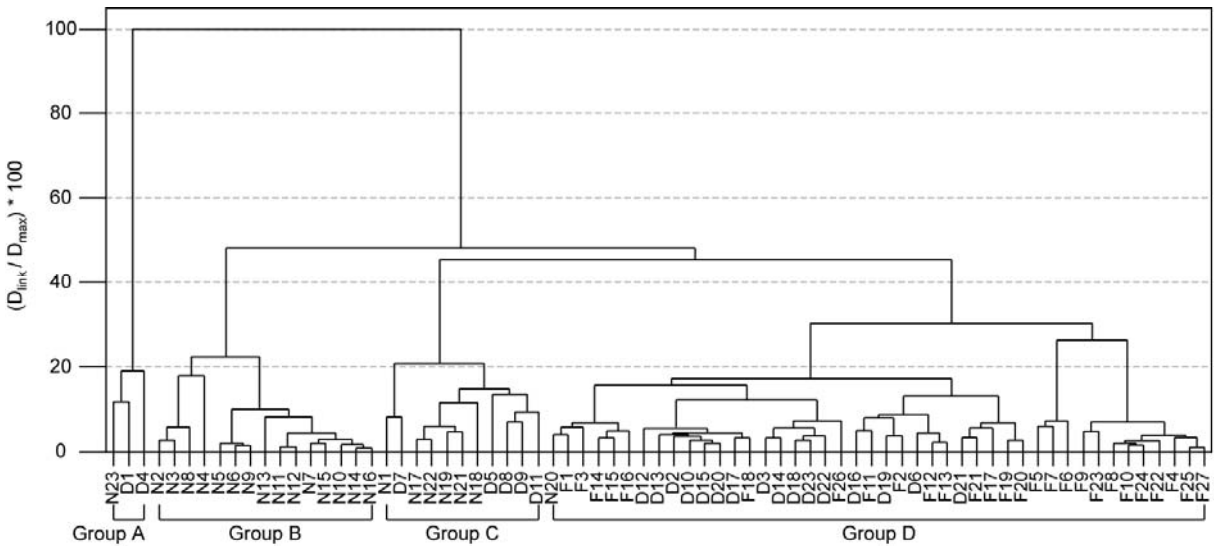

Figure 3.7: Cluster analysis of all cases. Four groups were identified at ($D_{\text{link}}/D_{\text{max}}$) x 100 < 35.

Factor analysis of all cases identified six VFs with eigenvalues greater than 1, which explained 78.6% of the spatio-temporal variance (Table 3.2; Appendix A, Table A1). VF1 had strong positive loadings on Br^-, Al, Ca, Cd, Co, Cr, Fe, K, Mn, Ni, S, and Zn and explained 41.1% of the variance. pH was not strongly represented in VF1, however, it had a moderate negative relationship to the primary constituents. VF2 was related to salinity with strong positive loadings on EC, Cl^-, and SO_4^{2-} (10.6% of the total variance; Table 3.2). VF3, VF4, VF5 and VF6 each accounted for less than 8% of the total variance. VF3 had a strong loading on As, and nutrients (NO_2^- and P) were the principal constituents of VF4. VF5 did not have strong loadings on any parameters and water temperature and DO were the primary contributors to VF6.

Table 3.2: Factor analysis of all cases identified six VFs. The principal components (factor loadings > 0.7) for each VF are listed in order of decreasing contribution.

Varifactor	Principal Parameters	Eigenvalue	% of Variance	Cumulative %
VF1	Fe, Mn, Co, Ni, S, Cd, Br, Cr, Al, Zn, Ca, K	11.92	41.1	41.1
VF2	SO_4^{2-}, EC, Cl^-	3.07	10.6	51.7
VF3	As	2.23	7.7	59.4
VF4	P, NO_2^-	2.04	7.0	66.4
VF5	None > 0.7	1.91	6.6	73.0
VF6	DO, water temp	1.63	5.6	78.6

The parameters which explained the majority of spatial variation within each sample period were compared over time by performing CA and FA on each sample period. Cluster analysis resulted in three groups at $(D_{\text{link}} / D_{\text{max}}) \times 100 < 40$ for each sample period (see Appendix A, Figures A1, A2 and A3), which were then mapped to visualise the spatial patterns (Figure 3.8). For the Nov04 data, Group A (N) consisting of the sites with low ratios of Ca, K, Mg and Na to EC, located in the western and southeastern regions of the wetland. Group B (N) was characterised by sites with normal ratios of basic cations to EC and were located in the central area near the drain. Group C (N) only contained one site located in the south that was very acidic (2.57 pH), saline (51 dS m^{-1}) and had very high concentrations of Al, Fe, and Mn (7.58, 86.05, and 44.34 mg L^{-1}, respectively). Factor analysis of Nov04 identified six VFs that accounted for 92.7% of the total variance (Table 3.3; Appendix A, Table A2). More than 50% of the variance was explained by VF1, which had strong positive loadings on Cl^-, Br, Al, Ca, Cd, Co, Cr, Cu, Fe, K, Mn, Ni, Pb, S, Zn and a negative loading on pH. VF2 (12.8% of the variance) was explained by Mg and Se, and VF3 (9.4%) F^- and SO_4^{2-} (Table 3.3). VF4 was highly participated by nutrients (NO_2^- and P), as was VF5 (NO_3^-) and VF6 only had a strong positive loading on Sb.

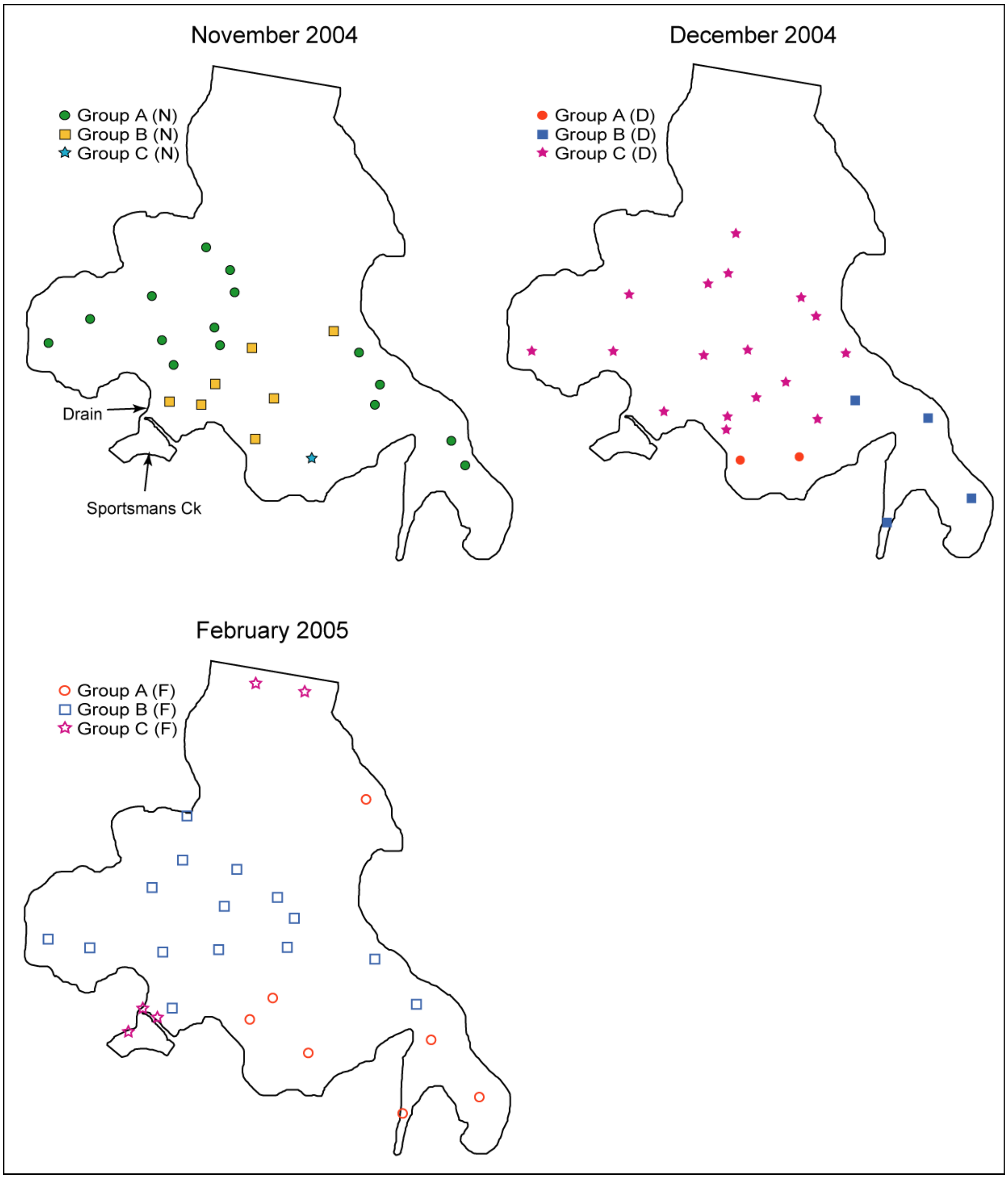

Figure 3.8: Results of cluster analysis of each sample period, indicating spatial clustering within each month. Three groups were identified at (D_{link} / D_{max}) x 100 < 40 for each sample period.

Table 3.3: Results of factor analysis of each sample period, indicating the dominant water quality parameters (in order of loading).

Varifactor	November 2004		December 2004		February 2005	
	Principal Parameters	% of Variance	Principal Parameters	% of Variance	Principal Parameters	% of Variance
VF1	Al, Cd, Fe, Mn, Zn, Co, Cu, Cr, Br, S, Pb, Ni, pH (negative), Ca, Cl^-, K	52.4	Al, F^-, S, Br, pH (negative), Ni, Zn, Ca, Co	29.8	Na, SO_4^{2-}, Mg, K, EC, Cl^-, Ca, S, Mn	39.1
VF2	Se, Mg	12.8	Cl^-, Na, Mg, EC, SO_4^{2-}, K	21.0	pH (negative), Al, Co, Fe, Ni, NO_3^-, water temp. (negative)	18.2
VF3	F^-, SO_4^{2-}	9.4	Cd, Cu, Mn	20.5	F^-, Sb	9.5
VF4	P, NO_2^-	7.7	P, NO_2^-	7.8	P	7.7
VF5	NO_3^-, DO	5.4	Sb (negative), As	5.3	As	6.4
VF6	Sb	5.0	None > 0.7	5.2		5.7

Spatial grouping in Dec04 exhibited a strong salinity gradient, with Group A (D) having the highest EC and Cl^- concentrations and Group C (D) the lowest. Group A (D) consisted of two sample sites in the southern region (Figure 3.8) and was characterised as very acidic (2.48-3.17 pH) with high EC (39-46 dS m^{-1}), very high concentrations of Al (5.01-9.06 mg L^{-1}), Fe (39.34-72.95 mg L^{-1}), and Mn (15.21-49.83 mg L^{-1}), and elevated concentrations of Co, Ni and F^-. Group B (D) consisted of sites in the southeastern corner of the wetland with moderate EC and neutral pH, whilst Group C (D) had the lowest EC and neutral pH and covered the majority of the wetland (Figure 3.8). Six VFs were also identified by FA of Dec04, explaining 89.6% of the total variance (Table 3.3; Appendix A, Table A2). Similar to Nov04, VF1 for Dec04 was related to salinity, metals and acidity. VF1 (29.8% of the variance) had strong positive loadings on F^-, Br, Al, Ca, Co, Ni, S, and Zn, and a strong negative loading on pH. VF2 (21.0% of the variance) was explained by EC, Cl^-, SO_4^{2-}, K, Mg, Na, thus representing salinity. VF3 was explained by Cd, Cu and Mn and accounted for 20.5% of the variance. VF4 (nutrients), VF5 (As and Pb) and VF6 each accounted for less than 10% of the total variance. No parameters had a strong loading on VF6.

Spatial zonation of water quality during Feb05 was different to Nov04 and Dec04, partly due to the presence of surface water in the northern region of the wetland. No groups were characterised by acidic water; rather, groups were determined predominantly by salinity. Group A (F) contained sites in the southern and southeastern regions of the wetland and also a site in the northeast (Figure 3.8). This cluster had the highest EC and variable pH, ranging from 3.27 to 7.40. Group B (F) covered the majority of the wetland and had moderate EC and neutral pH. Sites in the north, drain and creek were clustered in Group C (F) and had the lowest EC with variable pH (3.08-7.82). Eighty-four percent of the spatial variance during Feb05 was explained by six VFs (Table 3.3; Appendix A, Table A2). VF1 accounted for 39.1% of the variance and had strong loadings on salts (EC, Cl^-, SO_4^{2-}, Ca, K, Mg, Mn, Na and S). VF2 (18.2% of the variance) was contributed by Al, Co, Fe, Ni and pH, the latter negatively correlated (Table 3.3). The remaining VFs each accounted for less than 10% of the total variance, with VF3 and VF5 representing nutrients. VF4 was contributed by F^- and Sb and VF6 only had a strong loading on As.

3.4 Discussion

Water quality at Little Broadwater was highly variable over the 4-month study period, with conditions ranging from acidic to alkaline and fresh to saline over both temporal and spatial scales. The poorest quality water was in the southern area of the wetland which was very acidic and often very saline. Dissolved Al concentrations were up to four orders of magnitude in excess of the ANZECC (2000) guidelines of 0.8 µg L^{-1} and soluble Fe exceeded the guidelines by more than two orders of magnitude. Sammut, White and Melville (1996) and Indraratna, Glamore and Tularam (2002) have reported Al concentrations more than three orders of magnitude in excess of the ANZECC (2000) guidelines, and Fe up to an order of magnitude in excess of the guidelines in other drainage channels. The water in the southern area of Little Broadwater was ponded in depressions which may have been the result of sediment shrinkage due to drainage (White et al. 1997). Surrounding acidic and/or saline water flows into the depressions, and in dry periods becomes disconnected from nearby surface water. Evaporation may concentrate salts in these pools of water and readily transports acidic salts upwards through the soil profile due to a shallower watertable in these depressions (Lin et al. 2001), further increasing the acidity and salinity of the surface water.

If dry conditions persist then the area may become scalded due to the acidic and saline surface water and soil (Lin et al. 2001; Rosicky et al. 2006). Salt crusts, caused by high salt concentrations in the surface soil layer, along with iron crusts deposited on the sediment surface can also contribute to the death of vegetation (Rosicky et al. 2006). Prior to the start of the study the wetland condition in the southern region of Little Broadwater was similar to this due to frequent dry periods with low intermittent rainfall. Combined with reduced tidal exchange due to high creek salinity, this contributed to the formation of the large scald and thus the poor water quality observed throughout the study. The extremely high concentrations of Al and Fe in this area were above toxic levels for vegetation (ANZECC 2000) and may have contributed to the poor recovery of vegetation after a flood in 2001 and frosts in 2004.

Other water quality parameters were highly variable, including temperature, DO and nutrients. Variation in water temperature was related to changes in air temperature and solar insulation, both daily and monthly. Water temperature was highest from midday to the early afternoon in all sample periods, and also had the highest median in February 2005 when air temperature is generally the warmest. However, DO concentrations were not correlated with water temperatures. Dissolved oxygen concentrations were frequently below 5 mg L^{-1}, which may cause stress to fish (ANZECC 2000) if maintained for an extended period of time. However, DO is highly variable throughout the day as it is dependent on salinity, water temperature (generally) and biological activity and should therefore be measured over at least one diurnal cycle (ANZECC 2000). Thus, whilst spot measurements of water temperature and

DO give an indication of conditions at that point in time, spatial and temporal variability of these parameters needs to be interpreted carefully due to the high diurnal variation.

Nutrients were generally highest along the eastern side of the wetland where there was reduced tidal flushing. No ANZECC (2000) guidelines have been established for wetland nutrient concentrations, and although many of the sites exceeded the recommended estuary guidelines, Little Broadwater may naturally be a eutrophic wetland and thus the nutrient levels detected may be normal for the system.

3.4.1 Spatio-temporal characteristics of water quality

No overall pattern of spatial zonation within Little Broadwater was identified by CA, although this technique has been successfully applied to a number of studies in lagoons and bays to determine spatial patterns of water quality (e.g. Allen et al. 2007; Boyer, Fourqurean & Jones 1997; Hernández-Romero et al. 2004). Spatial clustering was only evident for cases in the southern region of the wetland during November and December 2004, which were characterised by very acidic and saline water with high concentrations of heavy metals associated with ASS. However, the interpolated maps of pH and Cl^- and examination of the raw data indicated that during February 2005 this region was still acidic and more saline than the rest of the wetland and had high heavy metal loads, indicating that there was some spatial zonation of wetland water quality which was not identified by statistical analysis. The lack of spatial or temporal groupings from the CA exemplifies the high within- and between-sample period variability and thus the complexity of surface water quality characteristics in Little Broadwater.

Spatial zonation within each sample period differed and could be attributed to the changing dominance of water quality parameters and the amount of variance explained by each. Salts, metals and pH had similar dominance during November and December 2004, although they all contributed highly to the first varifactor during November 2004, indicating complex relationships between parameters at this time. In December 2004 the parameters were spread over a number of varifactors suggesting a simplification of relationships. In contrast spatial zonation in February 2005 was due primarily to salts, with ASS-related parameters accounting for a much smaller proportion of the variance. The dominance of salinity during February 2005 resulted in sites in the north of the wetland being clustered with sites in the creek and drains, even though the northern sites were very acidic and had high concentrations of soluble Al and Fe.

Differences in spatial zonation between sample periods were also due to sites in the west and east during November 2004 having low concentrations of Ca, K, Mg and Na in comparison to Cl^- and EC. These sites were grouped together in the individual CA but were spatially separated by a zone of differing water quality. A possible explanation for the low concentrations of these ions is depletion by exchange reactions with clays (Nriagu 1978), although from the data collected it is unclear if this was the reason.

Temporal variation in water quality was predominantly due to decreasing salinity over the study period which was related to rainfall, tidal exchange and micro-topography. Decreasing salinity within the wetland over the study period corresponded to decreasing salinity in the drain, indicating increased freshwater exchange with Sportsmans Creek due to higher rainfall from the upper creek catchment. However, increased direct rainfall between December 2004 and February 2005 led to a 55% decrease in the median salinity within the wetland and only a decrease of 26% in the drain salinity over the same period, indicating the importance of both direct and indirect rainfall and creek water quality. Micro-topography affects temporal patterns of salinity in combination with climatic conditions and tidal flows. For example, if tidal exchange was reduced and rainfall decreased, surface water may pond in depressions which is then evaporated, increasing salinity over time.

Tidal exchange, micro-topography and rainfall also influenced the spatial variation in salinity. Spatial zonation of salinity was consistent across the study period, with the highest EC and concentrations of Cl^- and SO_4^{2-} in the southern and southeastern regions. As previously discussed, shrinkage of sediments due to drainage may have caused depressions in which water pooled and was then subject to evaporation and concentrations of salts. If tidal exchange is allowed when the creek salinity is high, the impact of depressions and evaporation may increase and enhance spatial differences in wetland salinity. High rainfall prior to the February 2005 sampling period appears to have flushed some salts and ASS products from this area of the wetland. This considerably reduced the influence of ASS parameters and increasing the influence of salinity on spatial variation during February 2005.

3.4.2 Long-term monitoring of rehabilitation

The design of a long-term water quality monitoring program needs to consider the following factors: the parameters to be monitored, spatial coverage and sampling frequency (Maher, Cullen & Norris 1994). Spatial and temporal variability, along with logistical and financial limitations, determine the number of sites and frequency of sampling (Maher, Cullen & Norris 1994).

Long-term research at Little Broadwater should focus on salinity and the principal variables associated with ASS (pH, Al, Fe and Mn), as these were identified as the dominant sources of spatio-temporal variation in water quality. This corresponds to other studies on ASS floodplains where concentrations of Al, Fe, Ca, Cl^-, K, Mg, Na and SO_4^{2-} are monitored (e.g. Indraratna, Glamore & Tularam 2002; Johnston, Slavich & Hirst 2005b). Nutrients did not have a considerable contribution to the total variance of water quality during the short study period, and concentrations were generally quite low and did not exhibit consistent spatial or temporal patterns. However, high nutrients may be an indicator of pollution by cattle and/or agricultural runoff and the nutrient status of Little Broadwater is unknown – therefore, nutrients should be included in a long-term monitoring program of the wetland.

Factor analysis did not reduce the dataset substantially and many parameters with small variance and low maximum concentrations were included as primary constituents in explaining the spatio-temporal variation of water quality. This was particularly the case for parameters such as As, Co, Ni, Se and Zn. These ions often contributed highly to the first few varifactors, although this was due to the effect of standardisation of the data which increased the weighting of variables with low variance. However, standardisation was necessary because of the different scales of measurement between parameters (e.g. dS m^{-1}, °C, pH units, mg L^{-1}) and to reduce the influence of parameters with a large variance such as Cl^- and Na. Exclusion of these parameters with low variance from long-term monitoring would considerably reduce the size of future datasets and reduce the amount of redundant information.

The considerable spatial variation in water quality during the study period indicated that a long-term monitoring program needs to have extensive spatial coverage to ensure that differences between regions are detected. The establishment of sample locations also needs to take into account variation in surface water cover, as extended dry periods may result in areas of the wetland drying out. Establishing sites within different habitats may also provide anecdotal information regarding the effect of vegetation cover on water quality and vice-versa.

Monitoring needs to be conducted at least seasonally to account for temporal patterns of rainfall, as this strongly influences both spatial and temporal variation of salinity. Ideally sampling should be performed at least once a month, however, the need for adequate spatial coverage and budget constraints may require sampling to be conducted bi-monthly. Event-based sampling is often incorporated into monitoring programs, although depending on the aim of the long-term research, monitoring the discharge water quality after events may be more appropriate than sampling within the wetland. Access to the site after high rainfall is a further factor which may limit event-based monitoring.

3.5 Conclusion

Salinity was the dominant factor affecting spatio-temporal variation of water quality at Little Broadwater and was determined by a combination of climate, tidal exchange management and micro-topography. Acid sulfate soil oxidation products also contributed to spatial heterogeneity of water quality. The southern region was identified as consistently having different water quality to other areas of the wetland and was saline, very acidic and had high concentrations of metals. These characteristics and observations in the field suggest that there was poor interconnection between the southern region and the rest of the wetland due to depressions, which may have formed as a consequence of drainage. Low rainfall, evaporation and limited exchange with nearby pools of water resulted in a concentration of salts and an upward movement of acidic salts from the shallow groundwater into the surface soil layer.

The establishment of a long-term monitoring program at Little Broadwater needs to consider the extensive spatial area and temporal variation of water quality within the wetland. The objectives of the rehabilitation project will determine the physico-chemical parameters monitored, spatial coverage and frequency of sampling. The high spatial variability of water quality at Little Broadwater requires that sample sites be established throughout the wetland. It is recommended that monitoring of ASS oxidation products such as Al, Fe and Mn, along with pH and salts (EC, Ca, Cl^-, K, Mg, Na and SO_4^{2-}) should be conducted at least seasonally. Event-based sampling may also be incorporated into long-term monitoring of Little Broadwater. However, it may be more appropriate to monitor this through discharge water quality due to high water retention time of the wetland and poor access to the site during wet conditions.

Chapter 4
Water Quality Dynamics of a Rehabilitated Floodplain Wetland in Northern New South Wales

4.1 Introduction

The coastal floodplains of eastern Australia have been severely degraded through the drainage of wetlands and prevention of tidal exchange (Johnston, Slavich & Hirst 2005b). However, drained wetlands are increasingly being subjected to rehabilitation trials to improve water quality and habitat, while maintaining current land use practices and providing flood protection (Haskins 2000). This may involve reflooding the wetland either seasonally or permanently, which encourages the regrowth of water-tolerant vegetation and provides habitat for waterbirds, fish and other aquatic animals (Eertman et al. 2002; Warren et al. 2002).

Rehabilitation of wetlands underlain by acid sulfate soils (ASS) is of particular importance, as these degraded environments can discharge very acidic water with high loads of aluminium (Al) and iron (Fe) into estuaries. These soils are stable when maintained in an anoxic state (Indraratna, Blunden & Nethery 1999) and consequently restoring wetlands may be a long-term solution to reducing and preventing further impacts. Two common techniques for rehabilitation of ASS wetlands are freshwater ponding and restoring tidal exchange. These techniques are designed to reduce acid production and discharge and improve habitat whilst maintaining the agricultural viability of the land, since many floodplain wetlands are located on private land.

Freshwater ponding is beneficial for drained floodplain areas which are used for grazing as it provides a means to remediate ASS-affected wetlands while increasing the grazing potential (Clay et al. 2007). Ponding maintains a high watertable and water is lost primarily through evapotranspiration (Tulau & Henderson 2001). Water is ponded through the use of in-drain water retention structures such as weirs to maintain high drain water and groundwater levels and hold water on the floodplain. Benefits of freshwater ponding include: reduction of acid generation, discharge, and groundwater movement to drains; increased vegetation and organic matter cover in the backswamp, thereby increasing backswamp grazing and habitat values; and, reduced occurrences of low dissolved oxygen (DO) due to an increase in vegetation cover (Tulau & Henderson 2001). However, there are a number of problems associated with freshwater ponding. Firstly, acidic water is essentially contained on the floodplain and during high rainfall may be flushed into the river system (White et al. 1997). A second problem with freshwater ponding is those wetlands which are located along the estuarine reach of a river usually have limited availability of freshwater during the dry season (White et al. 1997). The risk of the wetland drying out is increased, which may then lead to re-oxidation of the ASS and a large flush of acidic products into the river system when the wet season begins (Henderson & Tulau 2001).

Restoring tidal exchange is a second option for rehabilitation of ASS wetlands. Bicarbonates that are naturally present in the brackish water neutralise the acidic water. This is a better option than freshwater ponding as tidal exchange can continue during dry periods, reducing the risk of the wetland desiccation. However, this form of rehabilitation is only applicable to wetlands along the lower estuary where the tidal water is saline to brackish throughout the year. There are often concerns of salinisation of groundwater and the surrounding land (White et al. 1997) when reflooding with brackish or saline water, highlighting the importance of understanding the hydrology of the wetland prior to rehabilitation efforts to ensure excessive salinisation is reduced. A further problem associated with restoring tidal exchange is that often there is a severe die-back of vegetation due to the lack of salt-tolerant species, resulting in salt scalding (Tulau 2007). Therefore, the immediate benefits to grazing are limited due to the time lag for salt-tolerant vegetation to establish.

Monitoring of water quality during the rehabilitation process is essential to distinguish between natural variability and ecosystem response to changing management strategies (Finlayson & Mitchell 1999). Reference sites are often used to determine the progress of the project, however, reference sites should be similar in size, tidal range, position in the landscape, adjacent land use and geomorphology (Neckles et al. 2002). If no suitable reference sites can be found, then it is even more important to establish long-term monitoring to understand the temporal and spatial variation of the restoration site for directing management efforts (Boyer, Fourqurean & Jones 1997).

This chapter presents the results of a 2-year study designed to investigate the surface water hydrology of an estuarine ASS wetland undergoing a rehabilitation trial. The specific objectives were to: i) determine if restored tidal exchange led to an improvement in discharge water quality; ii) investigate spatio-temporal characteristics of surface water quality; and

iii) quantify the impact of tidal exchange management and climate on within-wetland surface water quality.

4.2 Materials and methodology

4.2.1 Field equipment and monitoring

The field-based measurements were designed to investigate the seasonal and spatial variation of surface water quality and the effect of tidal gate manipulation. Drain water quality was measured continuously from March 2002 to February 2007 (60 months). A pilot study (Chapter 3) led to the establishment of 45 surface water sampling sites and bi-monthly monitoring commenced in April 2005 and continued to February 2007 (sample periods are referred to as Apr05, May05, Jun05, etc.).

Continuous monitoring of the *in situ* properties of the drain water at Little Broadwater was conducted using a Greenspan integrated discharge and water quality monitoring system, installed and maintained by Greenspan and Clarence Valley Council (CVC). A second monitoring system was located in Reedy Creek behind the floodgates, which discharges from the degraded Everlasting Swamp upstream of Little Broadwater (Figure 4.1). Monitoring at Reedy Creek ceased in mid-July 2005. The data loggers were programmed to record at one hour intervals. A DC pump transported water samples from the drain and through the flow chamber where pH, electrical conductivity (EC) and DO were measured (Figure 4.2). Additional sensors recording temperature, water level and velocity were located in the drain. Occasional clogging of the DO probe reduced data accuracy during some periods. The elevation of the monitoring system was surveyed and the water height data was corrected using m AHD.

Surface water sampling sites were selected by overlaying a map of the wetland with a 100 m grid that covered the expected permanent water area. Sites were then selected based on the general spatial zonation identified by the pilot study and representation of different habitats. A total of 44 sites were selected within the wetland, and another was located in Sportsmans Creek between the Sportsmans Creek weir and the Little Broadwater mouth (Site 20; Figure 4.3).

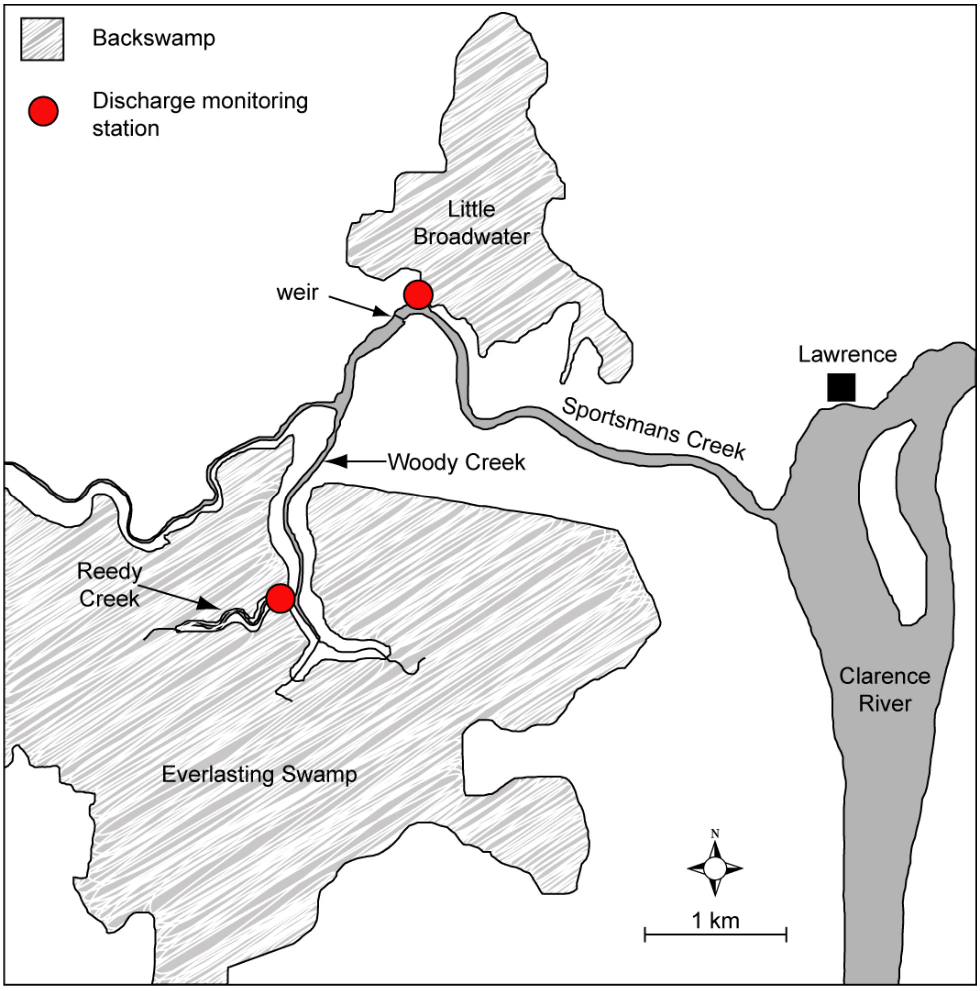

Figure 4.1: Location of discharge water quality monitoring stations at Little Broadwater and Reedy Creek, which drains the degraded Everlasting Swamp.

Figure 4.2: The flow-through chamber and sensors of the Greenspan integrated discharge and water quality monitoring system (courtesy CVC).

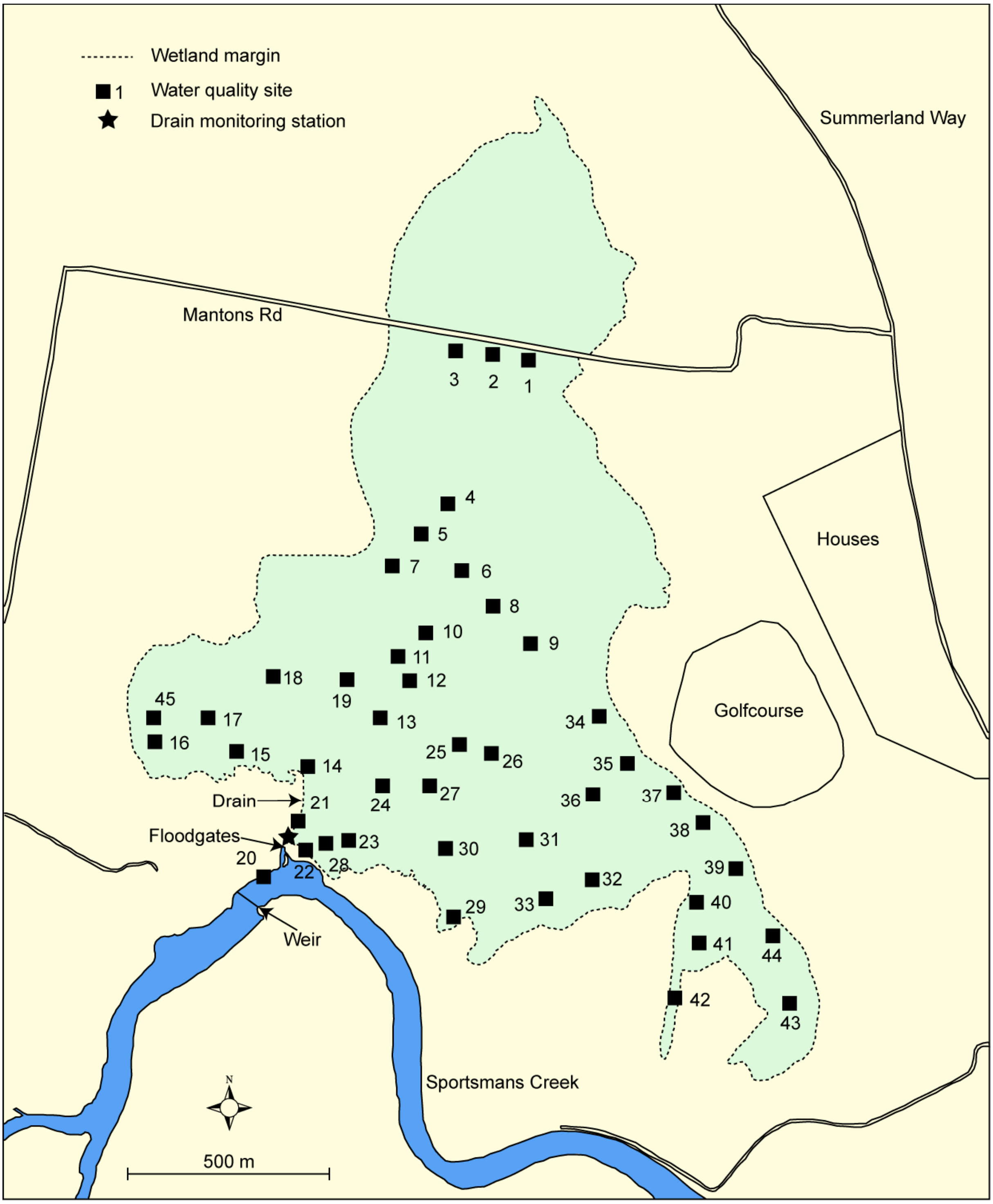

Figure 4.3: Surface water monitoring sites in Little Broadwater. Forty four sites were established in the wetland and another site was located in Sportsmans Creek (Site 20).

At each site pH, EC and water temperature were measured with a Horiba U-10 or a TPS WP-81 handheld probe. The probes were calibrated before use with standard calibration solutions. Dissolved oxygen was measured with a Hach LDO^{TM} probe. *In situ* measurements of water quality were taken with the same method as described in Chapter 2, Section 3.2.2 due to the variable depth of water. Water depth was also measured at each site with a tape measure. Two surface water samples were collected from mid-depth at each site in 125 mL acid washed polyethylene water sample bottles. The bottles and caps were rinsed three times with wetland water and lowered into the water column using a 1.5 m sampling pole. All air was excluded from the bottles before sealing and storing. After collection, samples were stored at 4 °C as soon as possible, frozen within 48 hours of collection and remained frozen until laboratory analysis. The chemical parameters analysed were determined by the results of the pilot study, which indicated that spatial and temporal variability of water quality was due primarily to variation in chloride (Cl^-), calcium (Ca), sodium (Na), potassium (K), magnesium (Mg), sulfate (SO_4^{2-}), Al, Fe and manganese (Mn). Total nutrient (nitrogen (TN) and phosphorus (TP)) concentrations were also determined.

Of the two water samples collected, one was filtered through 0.45 µm Whatman filter paper and the other remained unfiltered. The filtered sample was analysed for cations (Al, Ca, Fe, K, Mg, Mn and Na) using the standard atomic absorption spectrometry (AAS) procedure, SO_4^{2-} via the turbidimetric method, and Cl^- using the automated colour method (Table 4.1). Total nutrients were determined from the unfiltered sample by persulfate digestion (Hosomi & Sudo 1986). Water quality analysis was performed in the Water Resources Laboratory and the Agricultural and Environmental Analytical Laboratory at the University of New England, Armidale, and the National Marine Science Centre, Coffs Harbour.

Table 4.1: Methods for the measurement and analysis of water quality variables.

Parameter	Method	Reference
Water temperature, EC and pH	U-10 Water Quality Checker/TPS WP-81	Horiba (1991); TPS Pty Ltd (1997)
DO	Hach LDO^{TM} sensor	Hach Environmental (2006)
Al, Ca, Fe, K, Mg, Mn and Na	AAS	Varian-Techtron (1979)
Cl^-	Automated colour	Rayment and Higginson (1992)
SO_4^{2-}	Turbidimetric	Rayment and Higginson (1992)
Nutrients (TN, TP)	Persulfate digestion	Hosomi and Sudo (1986)

Climatic data were sourced from the Australia Bureau of Meteorology, with rainfall recorded at the Lawrence Post Office approximately 2 km east of the study site. Evaporation rates were measured at the Grafton Agricultural Research Station, approximately 18 km southwest of the study site.

4.2.2 Statistical methods

Data were initially tested for normality using the Wilk-Shapiro method, with the majority of datasets being non-normally distributed (skewed or unsymmetrical). This is often a characteristic of water quality data, along with the presence of outliers, seasonal patterns, autocorrelation and dependence on other variables (Helsel & Hirsch 2002). The datasets were unable to be transformed to a normal distribution, and as a result nonparametric summary statistics were used in the analysis. Outliers were retained as they reflect the high variability of the environment and nonparametric summary statistics (e.g. median) are not biased by outliers. Temporal variation in water quality was initially tested using the Spearman's R coefficient (r_s) with SPSS 16.0. Correlations were classified as strong ($r_s > 0.7$), moderate (r_s 0.4-0.7) or weak ($r_s < 0.4$).

Multivariate techniques (cluster analysis (CA) and factor analysis (FA)) were used to determine spatial and temporal groupings and the primary water quality parameters influencing the clustering. Refer to Chapter 3 for specific details of the statistical analyses. Cluster analysis was initially applied to all cases to determine if there was a dominant pattern of spatial or temporal grouping. Temporal CA and spatial CA were then performed on the median parameters for each sample period and for each sample site, respectively. Factor analysis was performed on all cases. Parameters with factor loadings greater than 0.7 were classified as strong and considered a principal constituent. Multivariate statistics were calculated with Minitab 13 and SPSS 16.0.

Significant differences between spatial clusters and temporal clusters were tested using the Mann-Whitney test for independent variables (spatial groups) and the Wilcoxon Signed Ranks test for related samples (temporal groups). Factor scores for each varifactor (VF) identified by FA were used rather than raw data, as these are a combination of parameters and simplify the discussion of variability and influencing parameters. A significance level of $p < 0.05$ was applied.

4.2.3 Spatial interpolation of water quality

Surface water Cl^- concentrations and pH were mapped to visualise spatial and temporal variation in the data during the study period. Chloride was used as an indicator of seawater intrusion, as it is naturally present in seawater and is not a product of the dissolution of estuarine clays unlike Ca, K, Mg and Na (Sammut, White & Melville 1996). A table containing water quality data and XY coordinates of each sample site, recorded as Eastings and Northings during the monitoring period, was imported into the GIS program ArcMap 9.1. The table was converted into a shapefile with a projected coordinate system (WGS 1984), which geographically depicted the sample sites as vector points. The wetland boundary was digitised and geographically projected in ArcMap 9.1 using a Quickbird satellite image with projected sample sites as a guide. The surface water boundary was estimated from notes and photos taken in the field.

Chloride concentrations and pH values were interpolated using the Inverse Distance Weighted technique in ArcMap's Spatial Analyst, to predict values at unsampled locations. The wetland boundary (and for October 2005 a wet boundary line) was used as a barrier to limit the search for input sample points. This resulted in a predicted raster surface of values over the wetland for Cl^- and pH.

The factor score mean and standard deviation for each VF identified in FA of all cases were also mapped using the technique described above. Mapping of factor score means indicated the average spatial variability of water quality, whereas the standard deviation indicated the temporal variability of areas within the wetland (Boyer, Fourqurean & Jones 1997).

4.3 Results

4.3.1 Tidal exchange management

Rainfall and tidal exchange management have an effect on both drain and within-wetland water quality. However, adjustments to tidal exchange structures were often in response to rainfall events, making it difficult to determine the impact of tidal exchange management alone. In addition, records of adjustments to structures were only kept from April 2005, and even then not all changes were noted, further exacerbating the problem of separating the effect of management from rainfall.

The discharge pattern for eleven recorded tidal exchange adjustment periods during 2005 and 2006 were examined (Figure 4.4a), to aid in identification of possible exchange adjustments prior to April 2005. Adjustments to structures were categorised according to their purpose – exchange, drain or restrict. Periods of exchange (E1-E4) were characterised by increased inflow when the top dropboards were removed and the tidal gates were open longer on the rising tide. Restriction periods were during winter months when creek salinity was increasing, and were characterised by low inflow (Figure 4.4a). The first restriction period occurred in August 2005 (R1; Figure 4.4a) when one tidal gate was closed completely and the other was left to partially operate. The second restriction period (R2) was from mid-June 2006 to late-August 2006 when there was low rainfall (Figure 4.4b). The remaining adjustment periods (D1-D5) were to drain water off the wetland and followed large rainfall events (Figure 4.4b) which had caused minor flooding around the wetland. Water was drained from the wetland by removing the fish-flap but leaving dropboards in and the tidal gates closed to reduce inflow. As a result, these periods were characterised by increased outflow and decreased inflow (Figure 4.4a). Although management during 2006 was a combination of drainage and exchange throughout the year, the drain water level did not show evidence of large tidal fluctuations in contrast to previous years (Figure 4.5a). Changes in drain water level were instead mainly due to rainfall events.

Using the characteristics of inflow and outflow identified from known tidal exchange adjustment periods (Figure 4.4a), discharge prior to April 2005 was examined to determine when adjustments were made during the early period of tidal exchange. Hourly discharge was compared to hourly drain water level and daily rainfall to aid in the analysis (Figure 4.5a, b, c, respectively), as fluctuations in drain level indicated tidal variation and therefore exchange. During the last half of 2003 and the first half of 2004 there were large fluctuations in drain water level which corresponded with increased inflow on the discharge graph, indicating a period of exchange. Exchange was then restricted until late November 2004, evident from the minimal fluctuations in drain water level and decreased inflow. After this period exchange was increased again until mid-2005, corresponding with increased fluctuations in drain water level.

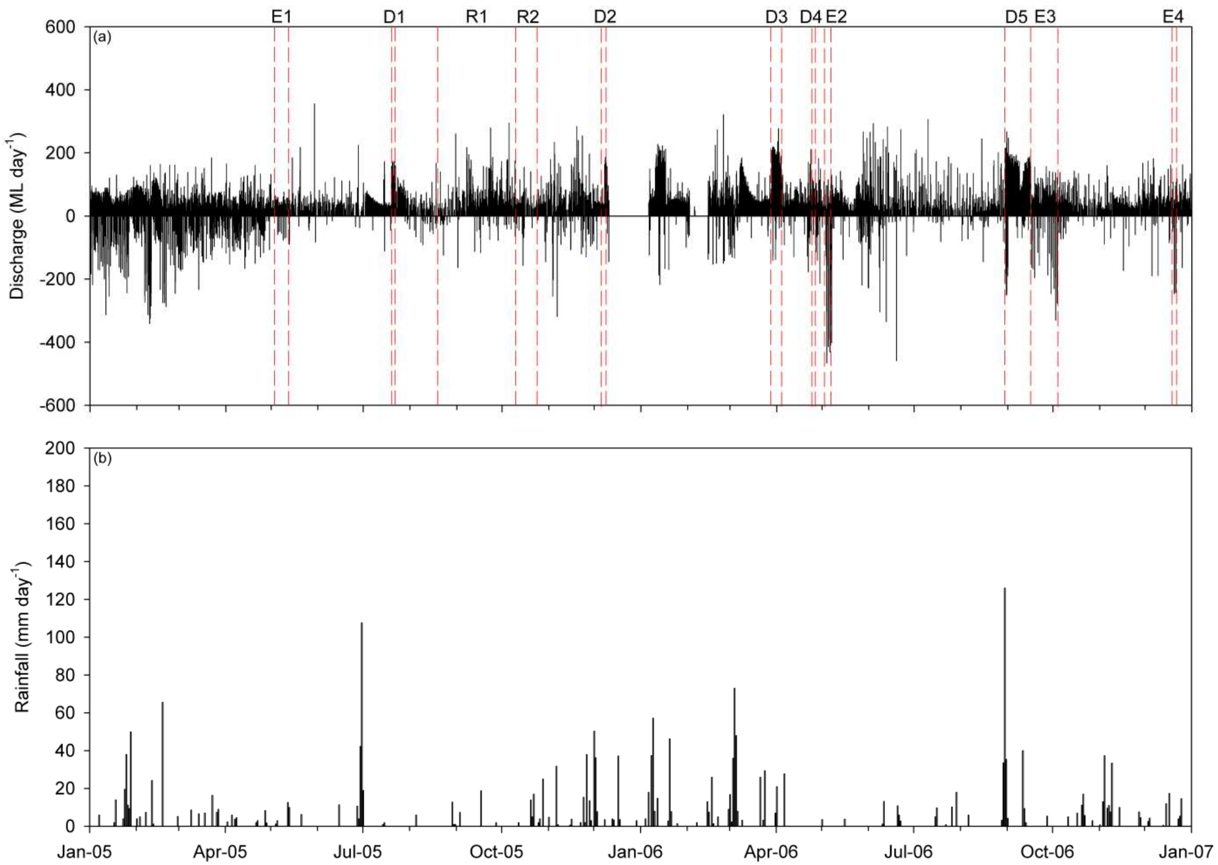

Figure 4.4: (a) Hourly discharge from Little Broadwater 2005-2006 and (b) daily rainfall (data provided by the Australian Bureau of Meteorology). Periods of adjustments to tidal exchange structures are indicated as: exchange (E1-E4); restriction (R1 and R2); and drainage (D1-D5). Positive discharge values indicate outflow and negative values indicate inflow.

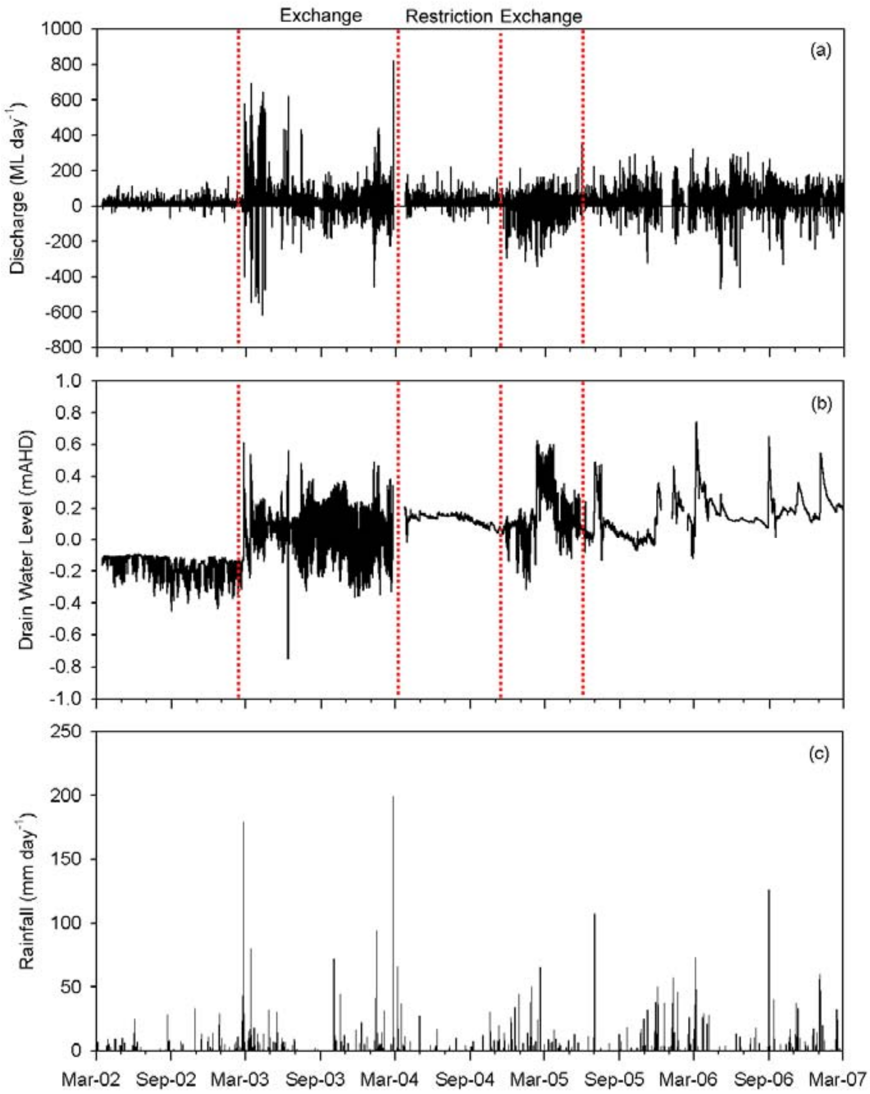

Figure 4.5: (a) Hourly discharge, (b) hourly drain level and (c) daily rainfall at Little Broadwater. Periods of exchange or restriction are indicated by the dashed red lines. Positive values of discharge indicate outflow and negative values indicate inflow.

4.3.2 Discharge water quality

The electrical conductivity of the discharge water exhibited a strong seasonal pattern at both Little Broadwater and Reedy Creek (Figure 4.6a). Electrical conductivity at Little Broadwater generally increased during winter and spring, and rapidly decreased in late summer and early autumn in response to increasing rainfall. However, during the winter of 2005 and 2006 there were unseasonably large rainfall events (Figure 4.6d), which resulted in winter fresh periods. Electrical conductivity remained low after the late-winter 2006 rainfall event. Reedy Creek EC was similar to that in the Little Broadwater drain.

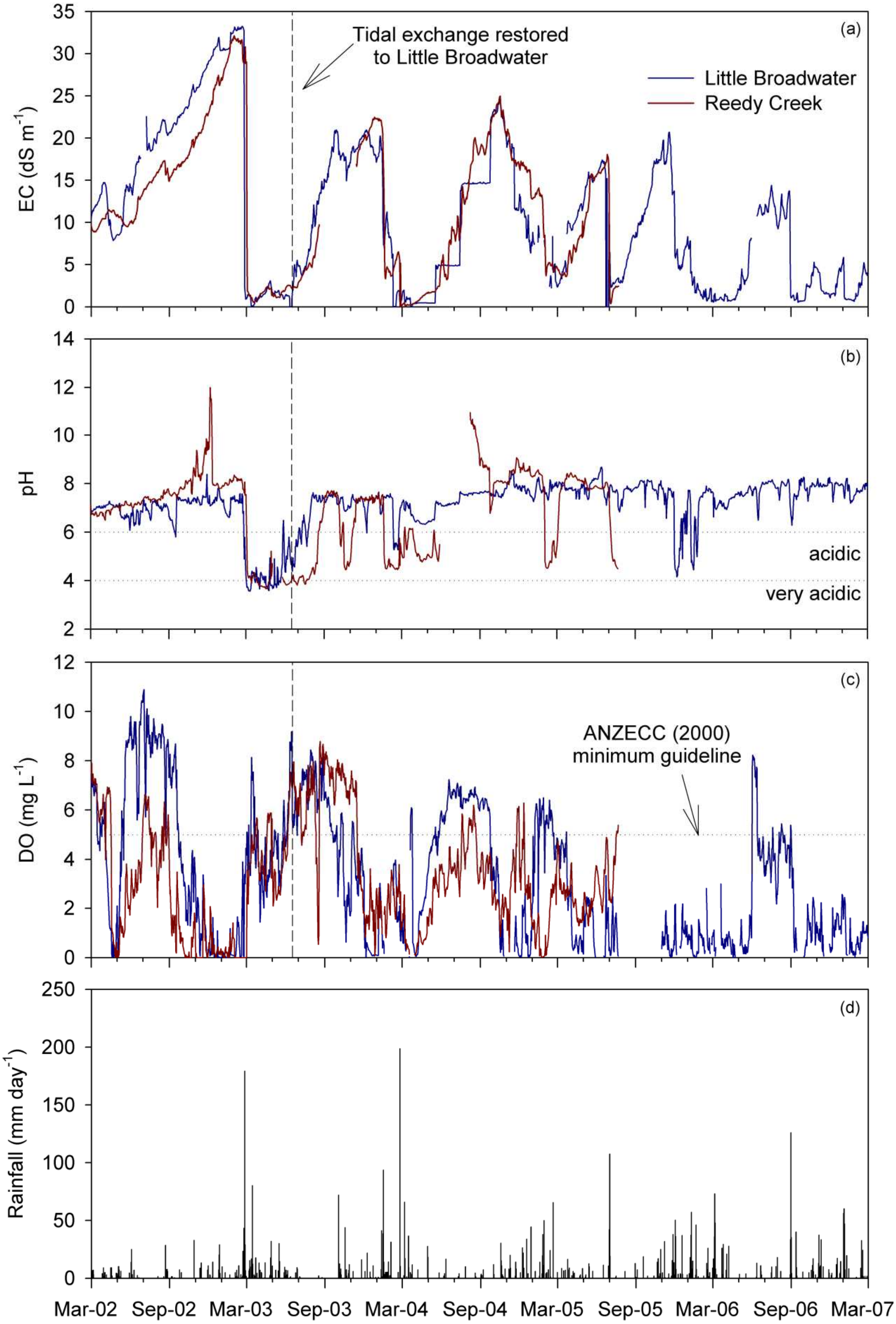

Figure 4.6: (a) Mean daily drain EC, (b) pH and (c) DO from Little Broadwater and Reedy Creek, March 2002 to March 2007. (d) Daily rainfall recorded at Lawrence. The date of restored tidal exchange at Little Broadwater is indicated.

The frequency, duration and severity of acid discharge events from Little Broadwater were reduced after restoring tidal exchange in June 2003 (Figure 4.6b). Prior to restoring tidal exchange (pre-modification) at Little Broadwater, discharge water pH at Reedy Creek and Little Broadwater was similar. Both sites had very acidic discharge (average pH of 4) in response to a large rainfall event in late February 2003. Drain water pH began to increase rapidly at Little Broadwater after tidal exchange was restored (post-modification) and had returned to near-neutral by July 2003. Discharge water pH at Reedy Creek did not return to near-neutral until mid-August 2003, 6 weeks later than Little Broadwater (Figure 4.6b). After this time there were four acid discharge events recorded at Reedy Creek, however, Little Broadwater only had two. The first acid discharge event from Little Broadwater post-modification was in February 2004, although drain water pH remained between 5 and 6 and only lasted for 2 weeks. In comparison, Reedy Creek had a pH of less than 6 (minimum 4.5) for at least 4 months (the logger failed for a few months after this time). The second acid discharge event at Little Broadwater consisted of two small events that quickly succeeded each other. The first small acid event occurred in December 2005 and lasted for 2 weeks and the second small acid event occurred in January 2006, lasting for only a week. The pH remained above 4 in both events.

Dissolved oxygen was similar between sites and exhibited a seasonal pattern (Figure 4.6c). Generally, DO was highest during winter and lowest during summer, however, it was predominantly less than 5-6 mg L^{-1}. There also appeared to be a trend of decreasing DO at both locations from 2002 to 2007.

4.3.3 Wetland surface water quality

The physico-chemical characteristics of water quality within Little Broadwater were highly variable over the study period (Table 4.2). Low 3rd quartile values, in comparison to the maximum values, indicated that the majority of samples had low concentrations of salts, metals and nutrients and only a small number had high concentrations.

Surface water EC within the wetland had high spatio-temporal variation, ranging between 0.2 dS m^{-1} and 53.4 dS m^{-1} (Table 4.2). Basic cations (Ca, K, Mg and Na) were strongly correlated with EC, as was Cl^- and SO_4^{2-} ($p < 0.01$; Figure 4.7). Major soluble salt ion ratios were compared to sea water and average world river water to determine if there was evidence of clay dissolution (*sensu* Sammut, White & Melville 1996) (Table 4.3). Ratios were generally similar to that of sea water, with the exception of Jun05 when there was a depletion of Na relative to Ca, K and Mg. Concentrations of Ca were also low relative to K in Jun05 and Aug05. However, in Aug06 there was a depletion of Ca, K and Mg relative to Na and similarly in Dec06 K concentrations were low relative to Na.

Table 4.2: Statistical descriptives of water quality parameters for Little Broadwater, April 2005 to February 2007.

Parameter	Median	Minimum	Maximum	1st Quartile	3rd Quartile
pH	6.80	2.76	11.43	5.00	7.74
Temp (^{o}C)	22.2	8.5	36.0	18.4	26.5
EC (dS m^{-1})	4.7	0.2	53.4	2.1	8.5
DO (mg L^{-1})	8.2	0.2	20.0	4.7	11.2
Al (mg L^{-1})	0.00	0.00	5.42	0.00	0.31
Ca (mg L^{-1})	15.2	0.0	348.5	7.7	40.6
Cl^- (mg L^{-1})	663	26	13187	306	1712
Fe (mg L^{-1})	0.22	0.00	21.65	0.07	0.64
K (mg L^{-1})	14.0	0.2	230.7	7.9	30.3
Mg (mg L^{-1})	48.5	2.1	880.1	25.9	102.8
Mn (mg L^{-1})	0.47	0.00	9.32	0.17	1.03
Na (mg L^{-1})	373	21	9529	204	706
SO_4^{2-} (mg L^{-1})	35.9	1.3	488.0	14.1	99.2
TN (mg L^{-1})	3.28	0.00	36.59	1.88	5.66
TP (mg L^{-1})	0.101	0.000	3.243	0.049	0.184

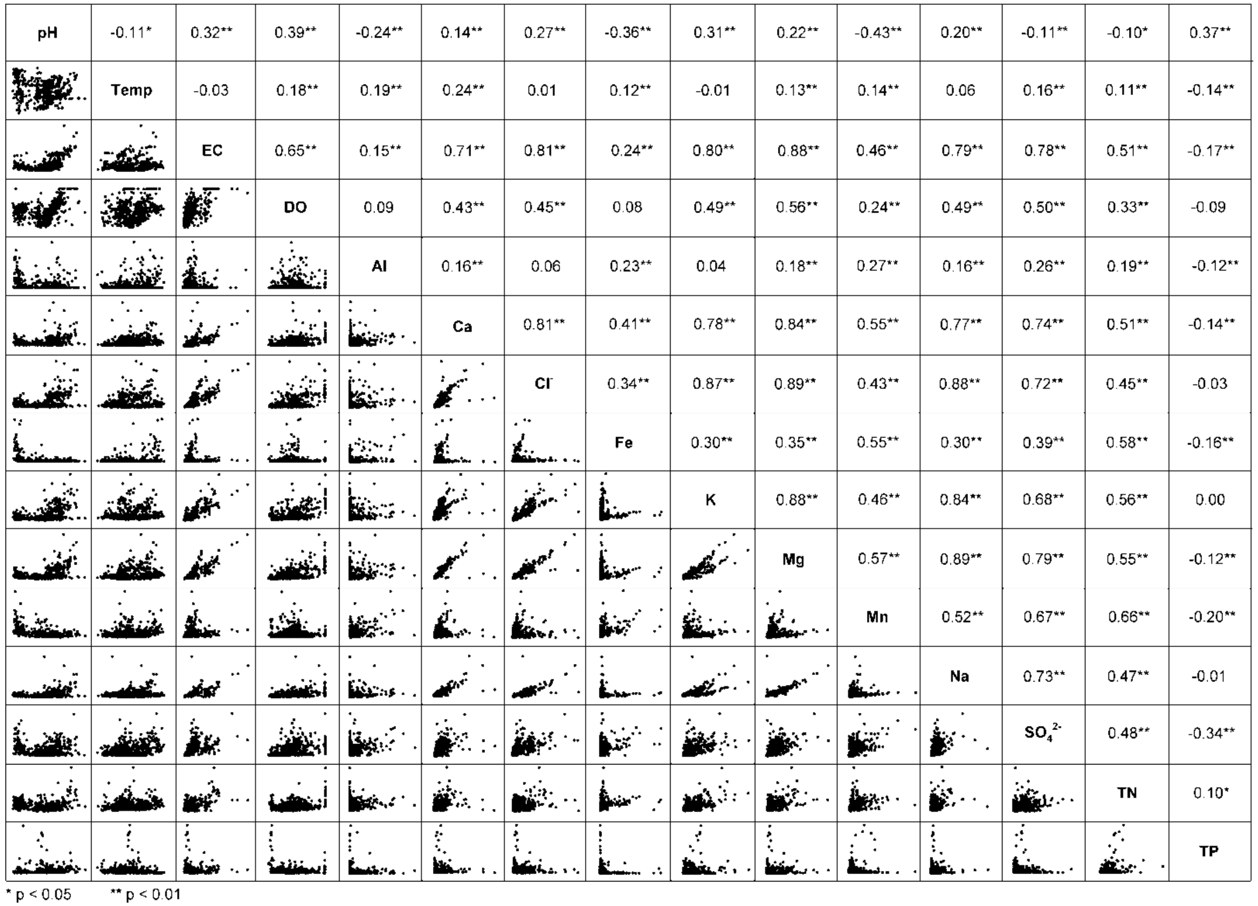

Figure 4.7: Correlation matrix indicating relationships between surface water quality variables. The Spearman's R coefficient is indicated, along with significance.

Table 4.3: Ratios of median monthly soluble ions at Little Broadwater.

	Cl^-:Na	Na: K	Na:Mg	Na:Ca	Mg:Ca	Ca:K
Apr05	3.1	23.3	6.1	18.2	3.0	1.3
Jun05	4.4	9.3	4.9	16.2	3.3	0.6
Aug05	1.6	19.9	6.7	36.5	5.5	0.5
Oct05	2.1	27.0	7.0	22.5	3.2	1.2
Dec05	2.5	24.8	5.1	12.9	2.5	1.9
Feb06	1.2	31.5	7.8	22.7	2.9	1.4
Apr06	1.0	30.2	8.1	23.0	2.8	1.3
Jun06	1.7	32.5	9.5	39.2	4.1	0.8
Aug06	1.7	34.7	11.4	42.1	3.7	0.8
Oct06	1.8	20.6	7.4	18.1	2.4	1.1
Dec06	1.7	36.5	8.6	29.2	3.4	1.3
Feb07	1.8	20.8	7.7	18.2	2.4	1.1
Sea water*	1.8	26.9	7.8	25.6	3.3	1.1
Mean river water*	1.2	4.0	1.5	0.4	0.3	8.4

* Source: Hem (1985)

There was no seasonal trend of Cl^- variation observed during the study (Figure 4.8). Rather, 2005 was more saline than 2006, with median Cl^- concentrations of 4048 mg L^{-1}, 3711 mg L^{-1} and 3679 mg L^{-1} in Apr05, Jun05 and Oct05, respectively. The lowest median concentrations of Cl^- were in Feb06, Apr06 and Feb07 (234 mg L^{-1}, 109 mg L^{-1} and 327 mg L^{-1}, respectively). Rainfall was more consistent throughout 2006 than 2005 (Figure 4.4b) and coincided with a change in management strategies from brackish/saline exchange in 2005 to freshwater exchange in 2006. Temporal variability of Cl^- and EC within each year corresponded to rainfall patterns with high rainfall in late July 2005 and August 2006 resulting in decreased Cl^- concentrations throughout the wetland in the following sample periods.

There was considerable spatial variation in Cl^-, which was most noticeable during the more saline months (Figure 4.8). The southeastern area of Little Broadwater had higher concentrations of Cl^- in the majority of sample periods, with the exception of Jun06 and Aug06 when the area around the drain had the highest concentrations. Sportsmans Creek had increased Cl^- concentrations during these months and also in Oct05. Surface water cover was reduced considerably in Oct05 when there was very low rainfall and exchange was restricted (Figure 4.4). Isolated pools of water had very high Cl^- concentrations during this period.

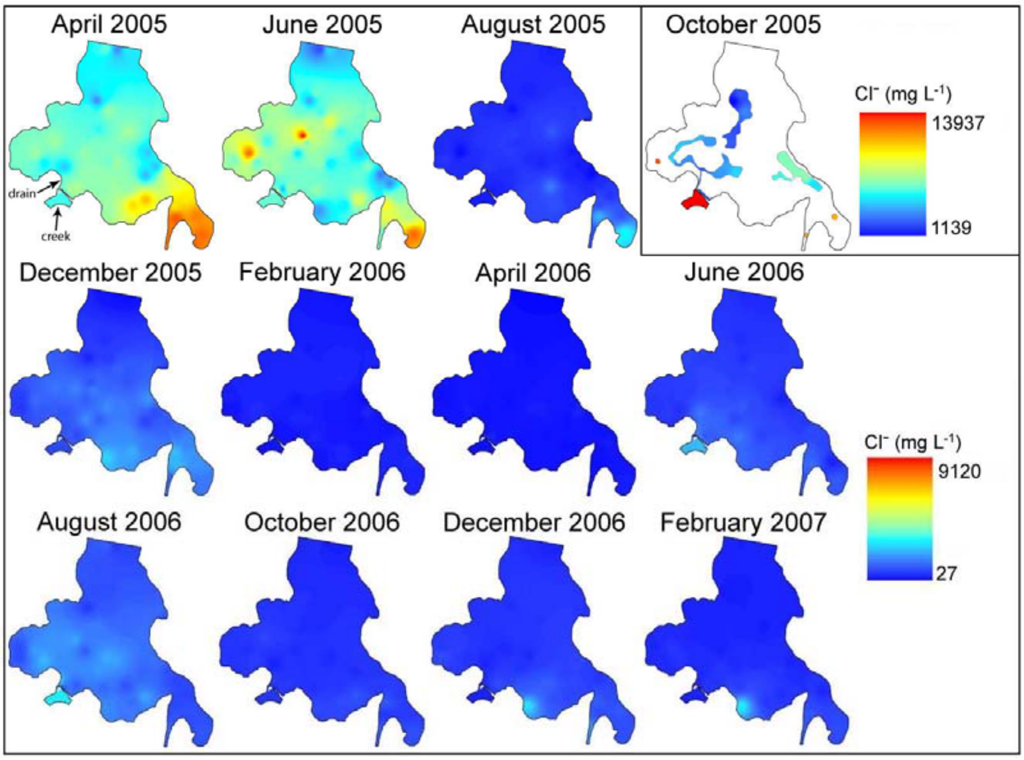

Figure 4.8: Chloride concentrations in Little Broadwater. Note the different scale for October05 due to very high Cl^- concentrations.

Surface water pH was highly variable both spatially and temporally (Figure 4.9), ranging from 2.76 (Site 3 Aug06) to 11.43 (Site 34 Oct06) (see Figure 4.3 for site locations). While the median pH was 6.84 (near-neutral) over the study period, at least 25% of the samples were acidic (pH ≤ 5.12; Table 4.2). Prior to the desiccation in Oct05, the majority of the wetland had neutral water with the exception of the sites in the north (Sites 1-3) and a small area in the south (Site 29). In Dec05 the entire wetland was acidic (pH < 6), with a pH of less than 4 measured at most sites (Figure 4.9). This acid event coincided with increased rainfall and acidic discharge from the wetland (Figure 4.6b, d). pH in the central area of the wetland improved to near-neutral by Feb06, however, all other areas remained acidic for at least 8 months (Figure 4.9). There was some exchange during this time, although tidal gate adjustments were predominantly to drain water from the wetland (Figure 4.4a). In Oct06 the southeastern and northern areas had become alkaline but by Dec06 were acidic again (Figure 4.9). Surface water pH was only weakly correlated ($p < 0.01$) to EC and basic cations (Figure 4.7).

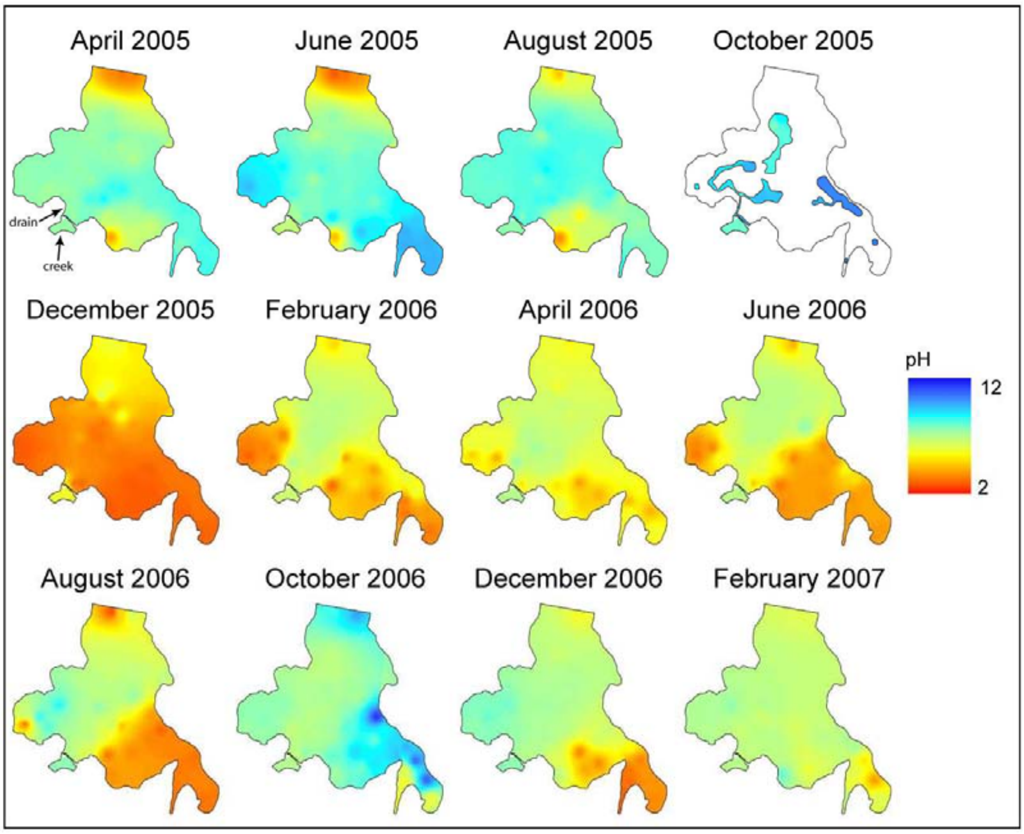

Figure 4.9: pH of Little Broadwater over long-term monitoring. The severe acidification in Dec05 was due to oxidation of ASS in Oct05.

Metal concentrations (Al, Fe and Mn) were also highly variable throughout the study (Figure 4.10a, b). The maximum concentrations measured were 5.42 mg Al L^{-1}, 21.65 mg Fe L^{-1} and 9.32 mg Mn L^{-1} (Table 4.2). High concentrations of Al, Fe and Mn corresponded with acidic water, predominantly at a pH of less than 4 (Figure 4.11). Accordingly, Dec05 had the highest median concentration of Al, Fe, and Mn, as did the acidic sample sites in the southeast corner of Little Broadwater. However, high concentrations of Al and Mn were also measured when the surface water was near-neutral (pH of 6-9; Figure 4.11a, c).

Total nutrient concentrations were generally high with a median concentration of 3.21 mg TN L^{-1} and 0.098 mg TP L^{-1} (Table 4.2). Total nitrogen was moderately correlated ($p < 0.01$) with EC, basic cations, Fe and Mn, however, TP was only weakly correlated ($p < 0.01$) with these parameters (Figure 4.7).

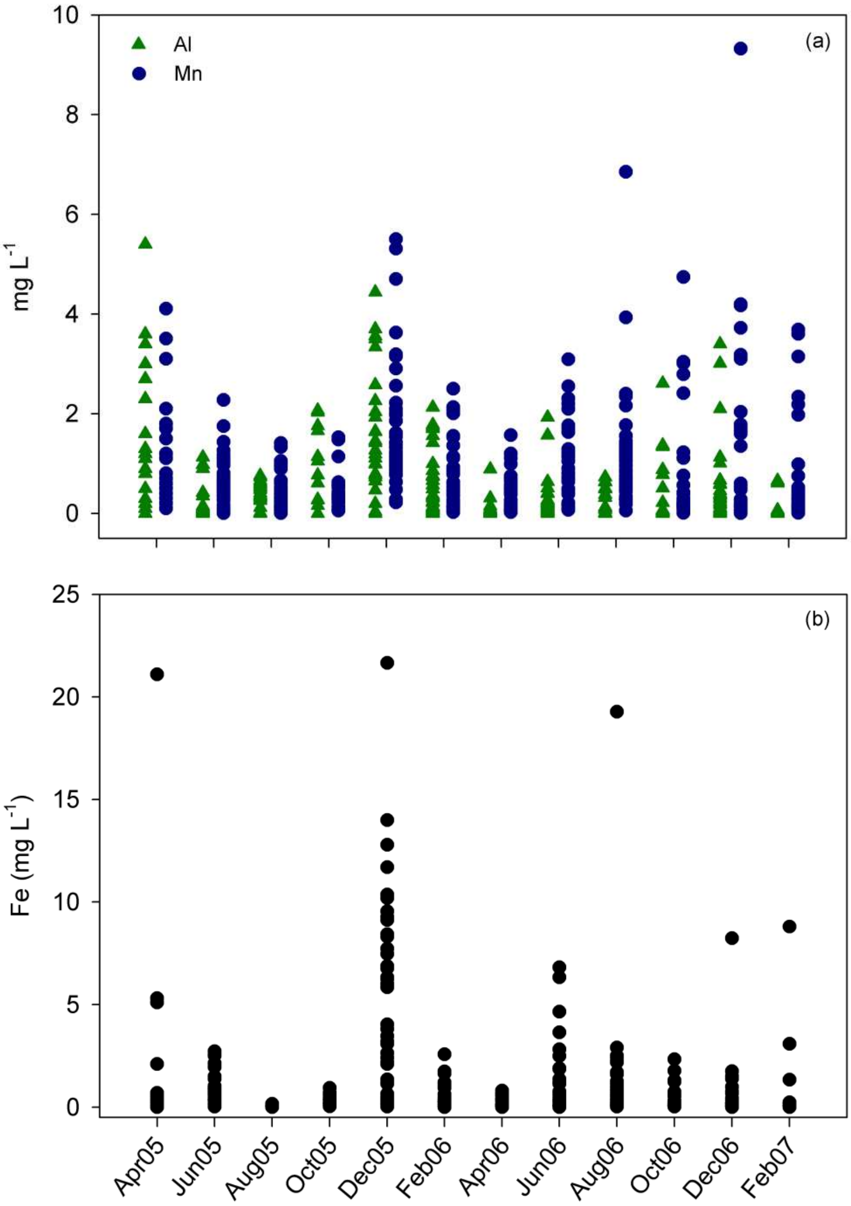

Figure 4.10: Surface water concentrations of (a) Al and Mn, and (b) Fe, Apr05 to Feb07.

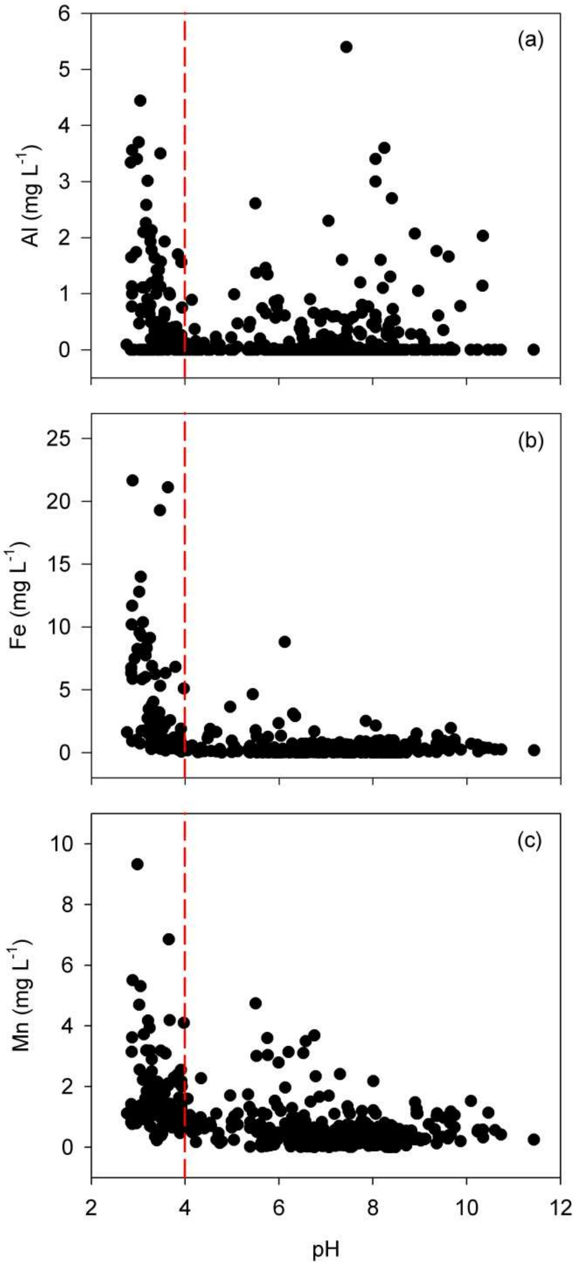

Figure 4.11: Relationship between concentrations of (a) Al, (b) Fe and (c) Mn, and pH. Concentrations of metals were generally highest when the pH was less than 4.

4.3.4 Multivariate characteristics of water quality

Longer-term patterns of spatial and temporal clustering was initially analysed through CA of all cases (Figure 4.12). The initial clustering at (D_{link}/D_{max}) x 100 < 70 was based on salinity, with 21% of all cases classified as Saline and 79% as less saline. The less saline cluster could then be further divided into two smaller clusters at (D_{link}/D_{max}) x 100 < 30, labelled Acidic and Fresh (Figure 4.12). The Saline group consisted primarily of samples from Apr05, Jun05 and Oct05 (Table 4.4). The majority of Dec05 samples were grouped in the Acidic cluster and also approximately half of the samples from Feb06, Jun06 and Aug06 (Table 4.4). The Fresh cluster accounted for the majority of samples and consisted mainly of the Aug05, Apr06, Oct06, Dec06 and Feb07 sample periods (Table 4.4).

Factor analysis of all cases yielded three varifactors (VFs) explaining 68.8% of the total spatio-temporal variance (Table 4.5). VF1 explained 41.0% of the total variance and was strongly contributed to by EC, Ca, Cl^-, K, Mg and Na – thus VF1 represented salts. VF2 (19.2% of the variance) was highly influenced by Fe, Mn and pH (negative loading) and had a moderate loading on Al (Appendix C, Table C1) and therefore represented ASS. Total nutrients were moderately to strongly represented in VF3 and only accounted for 8.6% of the total variance (Table 4.5; Appendix C, Table C1).

Temporal CA was performed to simplify the temporal pattern observed in the CA of all cases. Three clusters were identified at (D_{link}/D_{max}) x 100 < 35 and were termed Saline-phase, Acidic-phase and Fresh-phase (Figure 4.13). Saline-phase represented Apr05, Jun05 and Oct05 sample periods. Acidic-phase only consisted of Dec05. All other sample periods were clustered into Fresh-phase. In contrast to CA of all cases (Figure 4.12) Dec05 was initially clustered with Saline-phase rather than Fresh-phase (Figure 4.13).

Temporal phases were compared by testing the significant difference between the components, or VFs, identified in Table 4.5. Saline-phase and Fresh-phase were significantly different ($p < 0.05$) to Acidic-phase for all VFs (Table 4.6). Salts and ASS components were significantly different between Saline-phase and Fresh-phase ($p < 0.01$).

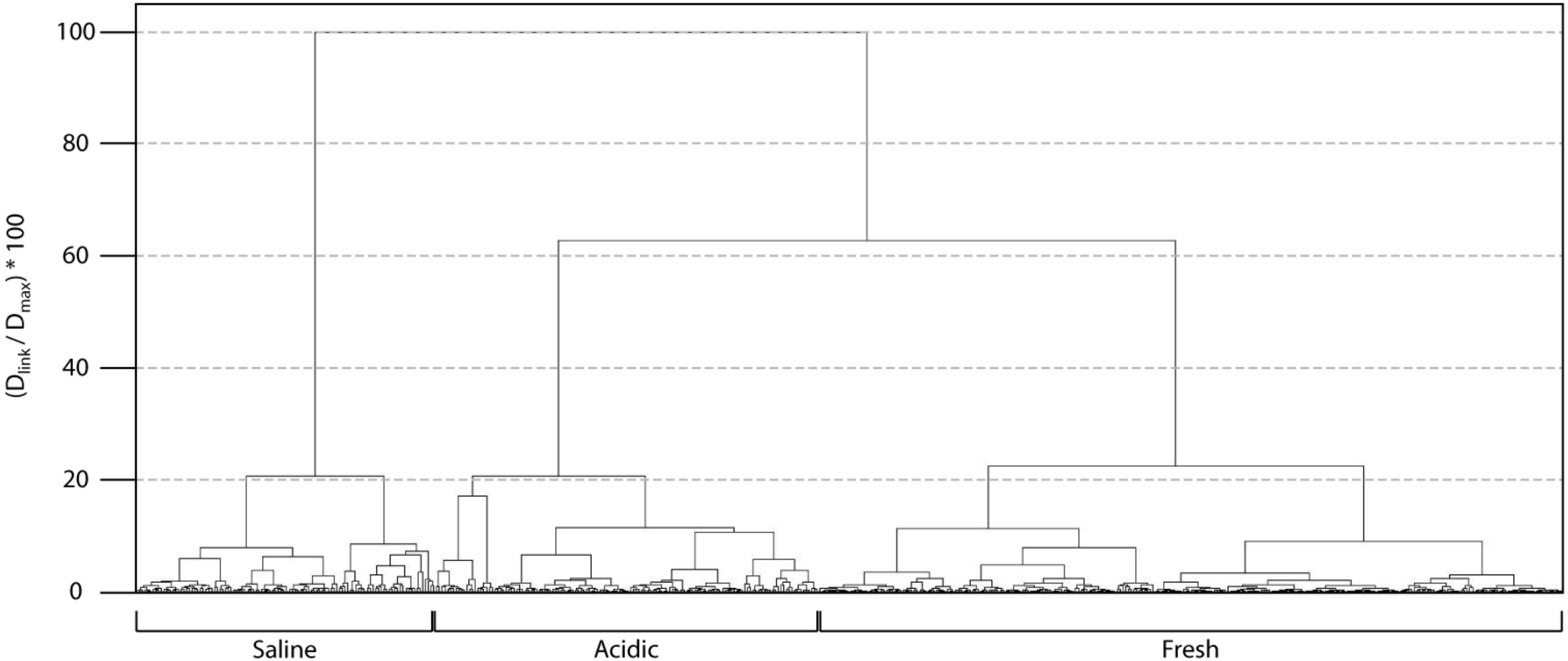

Figure 4.12: Cluster analysis of all cases (also see Appendix B). Samples were divided into three groups at (D_{link}/D_{max}) x 100 < 30: Saline; Acidic and; Fresh. Saline consisted mainly of Apr05, Jun05 and Oct05 sample periods. The acidic cluster was dominated by samples from Dec05.

Table 4.4: Percentage of samples for each month and overall which fall into each group for overall CA.

Month	Percentage of samples in Saline	Percentage of samples in Acidic	Percentage of samples in Fresh
Apr05	88.6	11.4	0.0
Jun05	90.7	9.3	0.0
Aug05	4.4	4.4	91.1
Oct05	96.0	0.0	4.0
Dec05	0.0	93.3	6.7
Feb06	0.0	40.0	60.0
Apr06	0.0	15.6	84.4
Jun06	0.0	50.0	50.0
Aug06	6.7	42.2	51.1
Oct06	0.0	8.9	91.1
Dec06	2.3	20.9	76.7
Feb07	0.0	13.3	86.7

Table 4.5: Factor analysis of all cases identified three VFs. The principal constituents (factor loadings > 0.7) for each VF are listed in order of decreasing contribution.

Varifactor	Principal Parameters	Eigenvalue	% of Variance	Cumulative %
VF1 - Salts	Mg, Cl^-, EC, K, Na, Ca	6.15	41.0	41.0
VF2 – ASS	Mn, Fe, pH (negative)	2.88	19.2	60.2
VF3 - Nutrients	TP	12.9	8.6	68.8

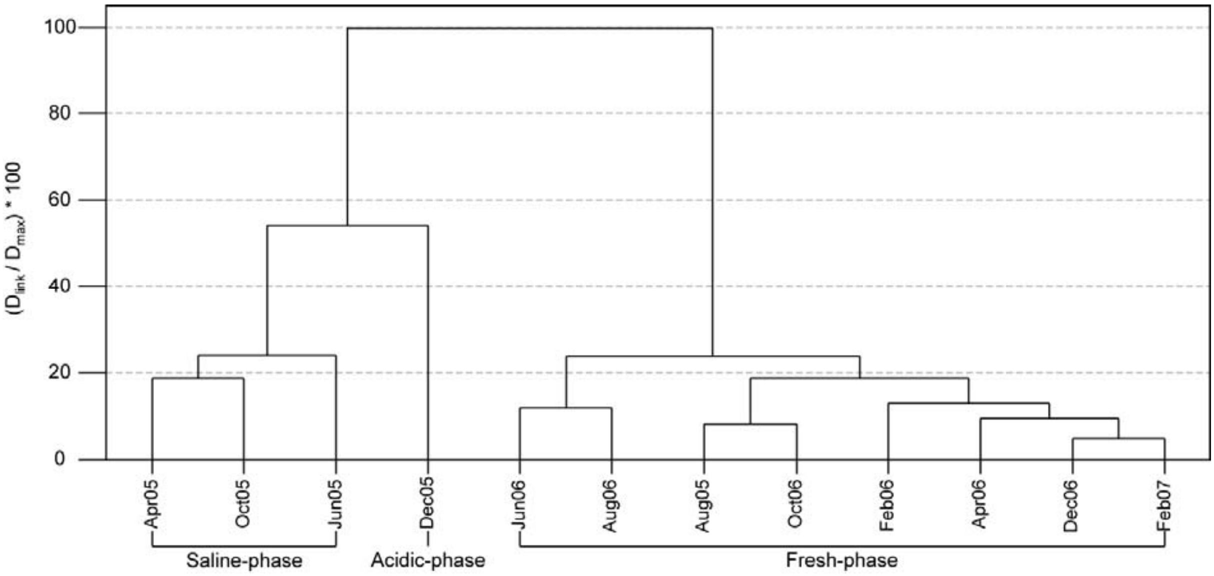

Figure 4.13: Temporal cluster analysis divided the sampling periods into three phases at (D_{link}/D_{max}) x 100 < 35: Saline, Acidic and Fresh.

Table 4.6: Significant differences between temporal phases (see Figure 4.13) and spatial zones (see Figure 4.14). Refer to Table 4.4 for VFs.

Group 1	Group 2	VF1 - Salts	VF2 - ASS	VF3 - Nutrients
Saline-phase	Fresh-phase	**	**	-
Saline-phase	Acidic-phase	**	**	*
Fresh-phase	Acidic-phase	*	**	**
ULB	MLB	**	*	**
ULB	LLB	**	**	-
MLB	LLB	**	**	**

* p < 0.05 ** p < 0.01 - >0.05

Spatial CA identified three zones of similar water quality characteristics within Little Broadwater (Figure 4.14; Appendix C, Figure C1). Cluster 1 covered mainly the northern area (Sites 1-13, 19 and 45) and was called Upper Little Broadwater (ULB). Cluster 2 comprised of sites in the west, drain and eastern regions (Sites 14-18, 20-28 and 34-38) and was labelled Middle Little Broadwater (MLB). Cluster 3 contained the southern and southeastern sites (Sites 29-33 and 39-44) and was named Lower Little Broadwater (LLB). ULB and MLB were significantly different for all VFs, as were MLB and LLB ($p < 0.05$; Table 4.6). ULB and LLB were significantly different for the salts and ASS components ($p < 0.01$).

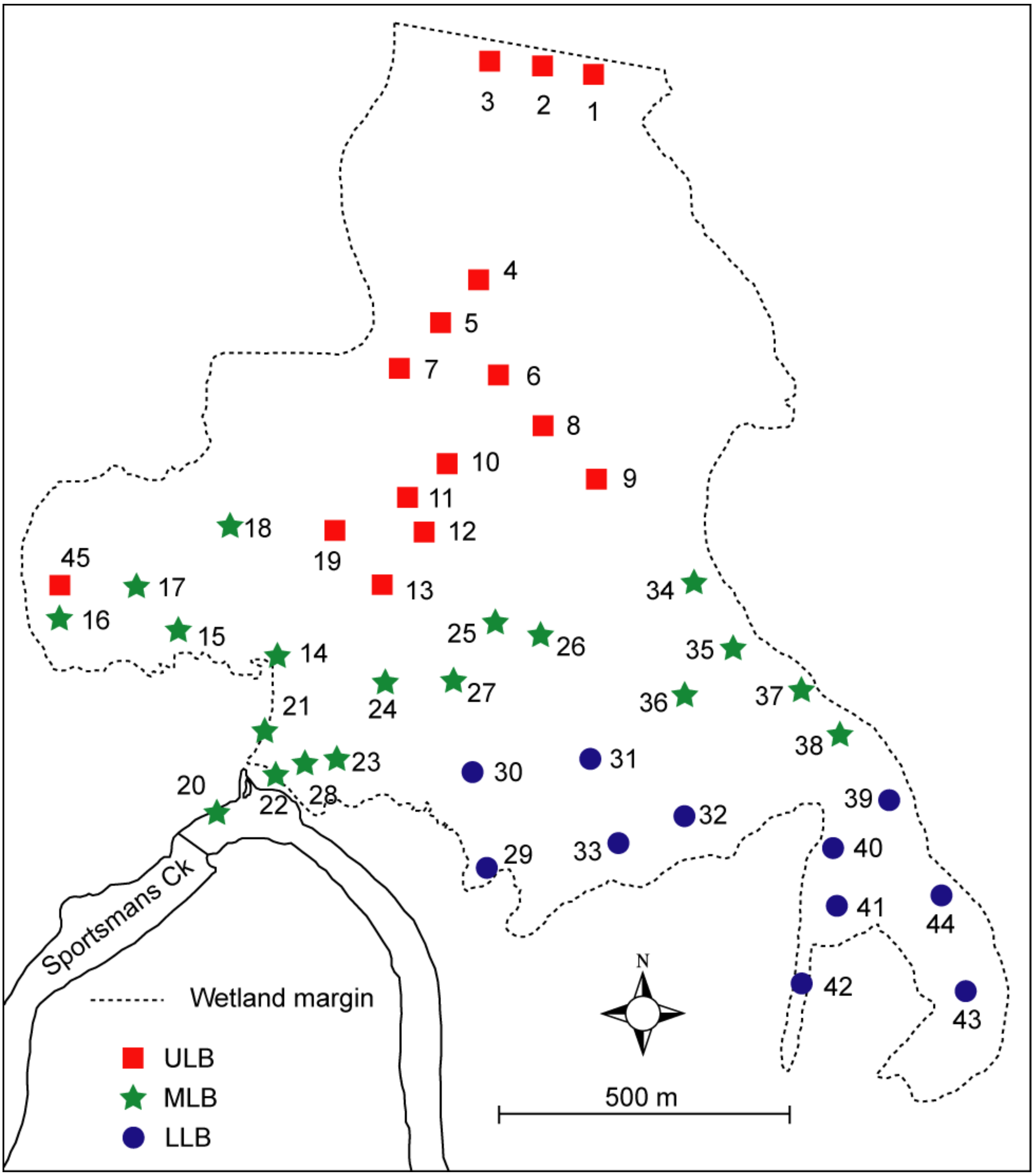

Figure 4.14: Spatial groups identified in spatial cluster analysis. Little Broadwater can be divided into three zones of different water quality characteristics: ULB, MLB, LLB.

Time series plots of the average factor scores for the three VFs indicated spatio-temporal patterns of water quality at Little Broadwater (Figure 4.15). Temporal variation of the salts and ASS components was similar between all zones (Figure 4.15a, b), with the salts component highest during Apr05, Jun05 and Oct05 (Saline-phase) for all areas. LLB had the highest average factor score for the salts component over the study period, with the exception of Aug06 when MLB was the highest. This corresponded with increased Cl^- concentration in the drain area (Figure 4.8). ULB consistently had the lowest average factor scores for the salts component (Figure 4.15a). LLB also had the highest average factor scores for the ASS component over the study period, although during Apr05 and Jun05 ULB was similar to LLB (Figure 4.15b). The ASS component was highest during Dec05 (Acidic-phase) for all spatial zones, due to acidification following the desiccation in Oct05. There was no consistent pattern of average factor scores for the nutrient component over the study period, however, LLB and/or ULB were generally the highest (Figure 4.15c). In Feb07 there was a considerable increase of nutrients in the LLB zone.

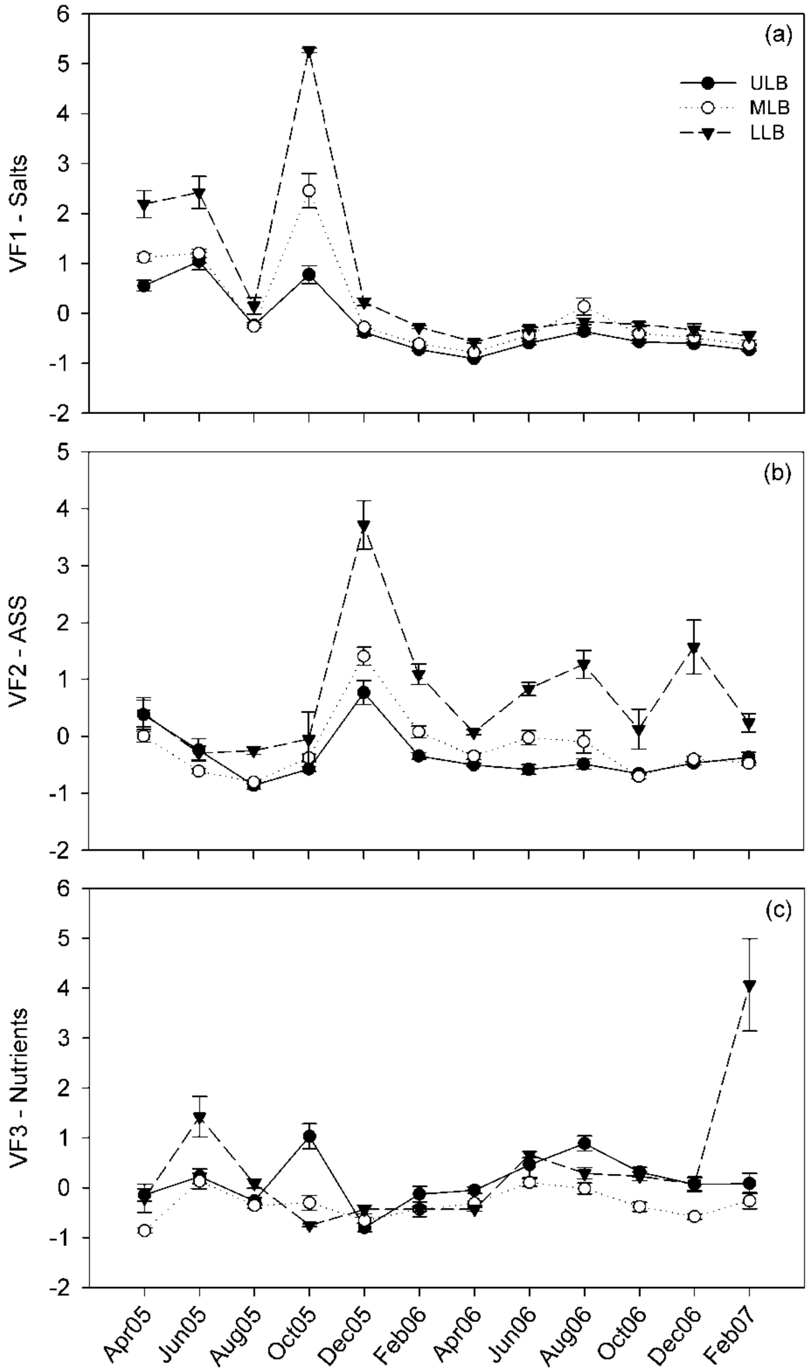

Figure 4.15: Time series plots of spatial zones identified at Little Broadwater (mean factor score +/- 1 s.e.) for each VF: (a) Salts; (b) ASS; and (c) Nutrients.

Factor score means and standard deviation were mapped to summarise and visualise spatial and temporal variation of water quality over the study period. On average, the salts component was highest in the southeastern corner (LLB) and lowest in the north (ULB) (Figure 4.16). The ASS component was also highest in LLB but was low throughout the majority of the wetland. The nutrient component was lowest in Sportsmans Creek and around the drain area, and highest in LLB and the northern region of ULB. Temporal variability of all components was greatest in the southeastern corner (LLB), with high variation of the ASS component also in the south (Figure 4.17). The salinity component was also highly variable in the western region around Site 16 (see Figure 4.3). The remainder of the wetland had low to moderate temporal variation of all components (Figure 4.17).

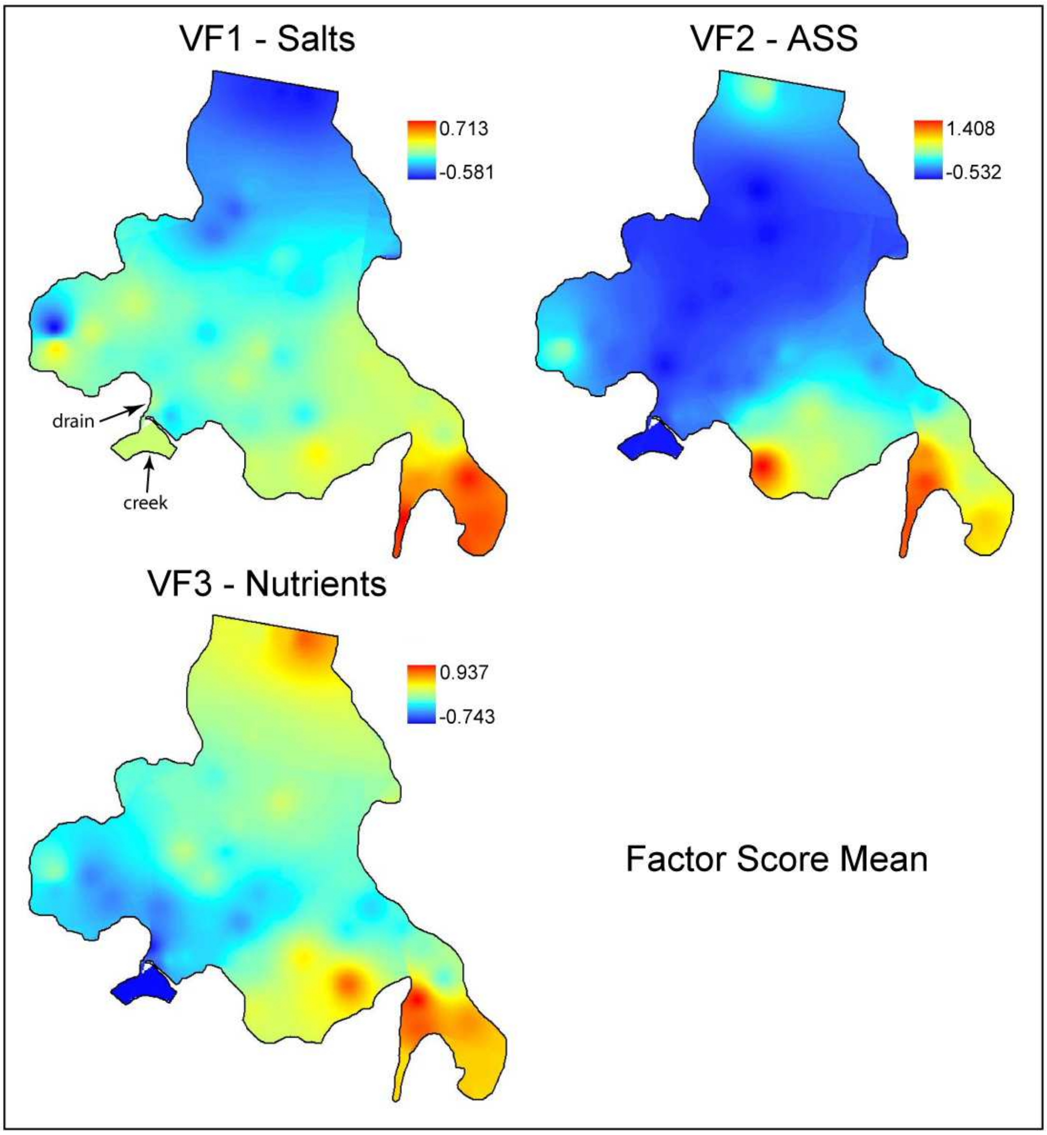

Figure 4.16: Spatial variation of mean factor scores for the three varifactors identified in factor analysis. Salts, ASS and nutrient components were all high in the southeastern corner of the wetland.

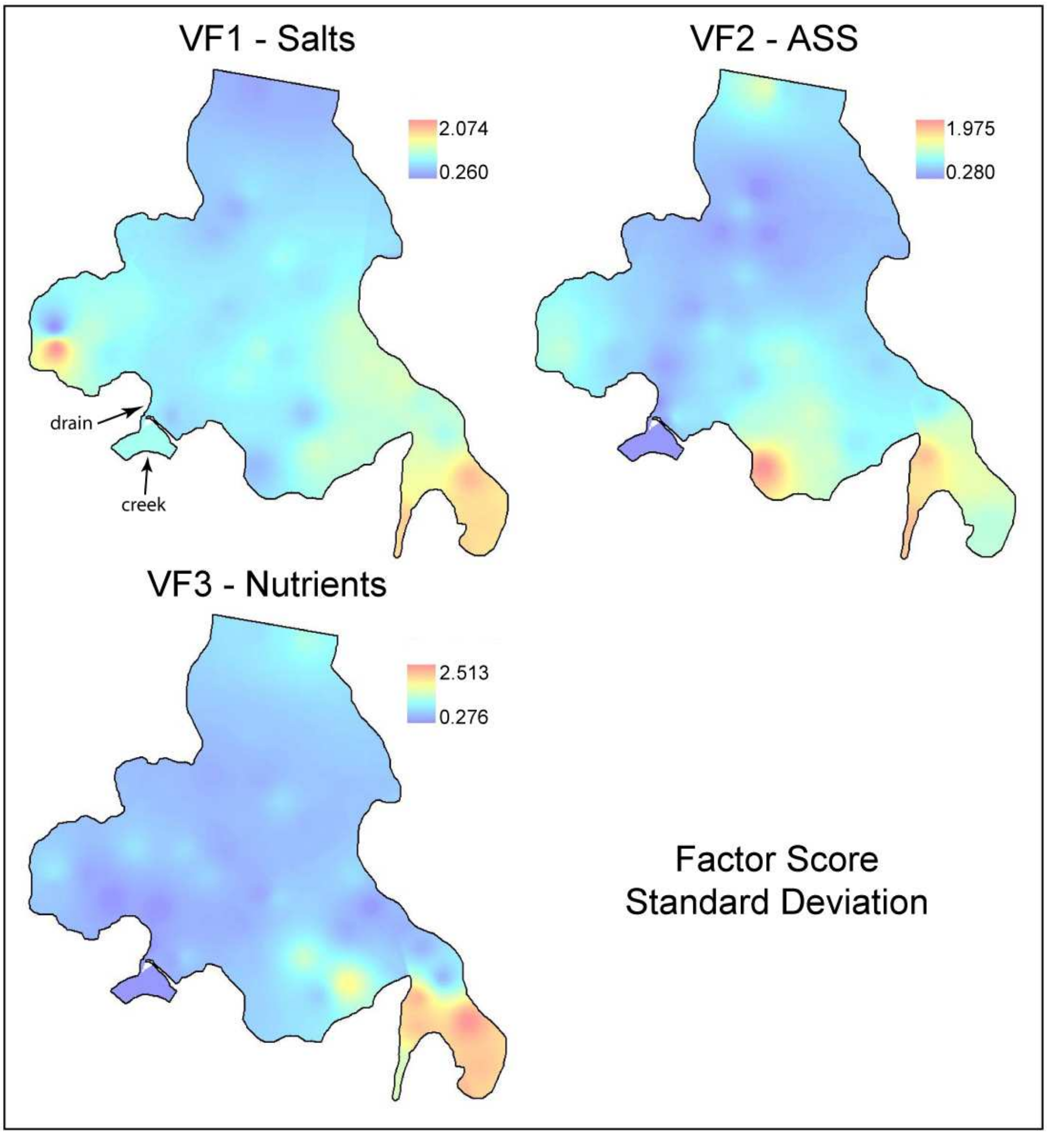

Figure 4.17: Temporal variability of the three retained varifactors (salts, ASS, nutrients), as indicated by the standard deviation of the factor scores for each sample site. The southeastern corner had high temporal variability for all three components.

4.4 Discussion

4.4.1 Discharge and within-wetland water quality

Quantitative analysis of drain water quality at Little Broadwater showed that the frequency, duration and severity of acid discharge reduced after rehabilitation strategies were implemented in mid-2003. However, the improvement did not appear to be related to buffering by saline water, as acidic events frequently occurred at Reedy Creek but not at Little Broadwater even though EC was similar between the two sites. Decreased acidic discharge from Little Broadwater could instead be explained by dilution of drain water by increased exchange and maintaining surface water cover within the wetland, rather than saline buffering. Periods of maximised tidal exchange at Little Broadwater often corresponded with the acidic discharge from Reedy Creek, indicating that dilution was important in reducing acid discharge from Little Broadwater. However, in July 2005 when there was an acid discharge event at Reedy Creek, exchange was reduced at Little Broadwater. Hence, the lack of acid discharge from Little Broadwater at this time could be attributed to maintaining surface water cover within the wetland, which prevented oxidation of the ASS and thus acid production.

Although acidic discharge was reduced overall at Little Broadwater, an acid event occurred during the 2005/2006 summer as a result of tidal exchange management and dry conditions during the previous spring. This resulted in desiccation of the wetland and subsequent oxidation of ASS, the products of which were mobilised by increased rainfall over the summer period. The management strategy during 2004 and 2005 was to allow tidal exchange until the creek EC reached approximately 13 dS m^{-1}. Reduced exchange, in combination with low rainfall and high evaporation, dramatically reduced surface water cover within the wetland and resulted in the oxidation of ASS. Increased rainfall in November 2005 mobilised large amounts of acid and heavy metals into the surface water which was then discharged from the wetland. Evapotranspiration of surface water and oxidation of ASS during dry periods have been identified as limitations to the success of ponding (Henderson & Tulau 2001), highlighting the need for adaptive management in brackish systems to maintain surface water cover. Groundwater may have also been an extra source of acidic water during this period, although the large amount of acidic surface water observed at the time suggests that the surface water was the main source of acidic water within the wetland.

Even though discharge water quality improved in terms of acidity, the concentration of DO in the drain was often below the ANZECC (2000) guideline of 5 – 6 mg L^{-1}. Dissolved oxygen is dependent on salinity, water temperature and biological activity (ANZECC 2000), and in ASS landscapes DO may also be depleted by the mobilisation of monosulfides which are commonly found in drains (Bush et al. 2004b). Dissolved oxygen was lowest over the summer months at both Little Broadwater and Reedy Creek, thus indicating that water temperature was having a strong influence. While there was a trend of decreasing DO at both sites over the study period, this may have been due partially to the unreliability of the DO probes. Within-wetland DO was highly variable, however, spot measurements provide limited information because of the diurnal fluctuations in concentrations which occur due to the dependency on temperature and biological activity.

While discharge water quality was improved by restoring tidal exchange and reflooding the wetland, poor quality water was measured within the wetland. This was attributed to poor circulation and flushing of outer areas. Mapping of surface water pH and Cl^- concentrations over the monitoring period provided a preliminary indication of which areas of Little Broadwater were well flushed and those that were not. The central area appeared to be regularly flushed, returning to near-neutral within two months of the acidification event while the rest of the wetland remained acidic for up to 8 months. The southeastern area suddenly became alkaline between August and October 2006, but it is unclear why this occurred. There was a decrease in salinity and rainfall over this period, so there would have been no increased buffering or dilution of the acidic water. Between October and December 2006 this area became acidic again, however, this could be attributed to the drier conditions experienced over the spring which may have led to oxidation of ASS around the edges of the wetland. The southern and southeastern regions were generally the most saline areas of the wetland and were also acidic for most of the two-year study period. Poor flushing of these regions may have contributed to or been the reason for the poor water quality. From observations in the field, exchange with Sportsmans Creek was limited during times of high drain water level due to increased discharge from the wetland. Therefore, only areas adjacent to and at the end of the drains were flushed and other areas were dependent on rainfall to flush or dilute acidic and/or saline water. The poor water quality of the southern and southeastern regions may have been due to groundwater inputs, and/or mobilisation of acids and metals from surface sediment oxidation. These processes are discussed in detail in Chapter 5.

Although the EC level for restriction of exchange under saline management was relatively low, excessive salinisation of areas within the wetland could still occur due to evapotranspiration processes. Monitoring in the southern and southeastern region showed that surface water EC could be up to 7 dS m^{-1} higher than in the creek. During the desiccation period in October 2005, individual pools had an EC nearly 2.5 times greater than the creek due to the concentration of salts by evapotranspiration. To try and reduce the excessive salinisation of the southeastern area, the EC level for tidal exchange was revised to 3 dS m^{-1} in 2006. During 2006 there was also consistent rainfall throughout winter in the Sportsmans Creek catchment, maintaining lower creek EC. This meant that exchange could continue throughout the drier winter and spring periods and prevent desiccation and oxidation of ASS. As a result, surface water salinity within Little Broadwater was considerably fresher during 2006 than in 2005. The variability of water quality and high dependence on rainfall for reducing salinity, thereby promoting tidal exchange, presents a number of implications for management, which are discussed in detail in Chapter 7.

Ratios of salts were generally similar to that of sea water, indicating that the high salinity observed in poorly flushed areas was due to a concentration of the salts in the water rather than an alternative source (such as sediments). The exception to this occurred during June 2005 when there was a depletion of Na relative to Ca, K and Mg. Under acidic conditions, hydrolysis of estuarine clays can release Ca, K, Mg and Na into the soil, which are then transported by rainfall or groundwater into the surface water (Nriagu 1978; Sammut, White & Melville 1996). However, there should not then be a deficiency in Na relative to the other dissolved salt species. Sammut, White and Melville (1996) suggested that other clay dissolution products or the acidic soils may absorb the soluble Na, accounting for the low ratio. In contrast, water during August 2006 was deficient of Ca, K and Mg relative to Na, indicating an additional source of Na. Again, this may have been due to preferential sorption of the other species over Na.

Metal concentrations often exceeded the ANZECC (2000) guidelines, and had varying dependencies on pH. The concentrations of Fe and Mn were logarithmically proportional to pH, as was Al to a lesser extent. Concentrations of metals increased considerably as the pH fell below 4. This relationship has been observed in groundwater by Sammut, White and Melville (1996) on the Richmond River floodplain, north coast of NSW and Indraratna, Blunden and Nethery (1999) on the Shoalhaven River floodplain, south coast of NSW. Metals precipitate out as the pH increases, forming flocs of iron hydroxide and aluminium hydroxide (Sammut, White & Melville 1996). This accounted for the sudden decline in metals at Little Broadwater when pH was greater than 4. While Al and Mn had high concentrations at pH less than 4, high concentrations were also measured when the surface water was near-neutral. These high concentrations may have been due to resuspension and dissolution of aluminium hydroxides. The maximum Al concentration was more than three orders of magnitude greater than the ANZECC (2000) guidelines, Fe was more than 70 times the recommended concentration and Mn was between 6 and 117 times the ANZECC (2000) guideline (freshwater and marine water, respectively). The threshold for concentrations of metals which may have adverse effects on aquatic animals is still unknown for estuarine systems (Sammut, White & Melville 1996), so while metal concentrations in Little Broadwater at times greatly exceeded the ANZECC (2000) guidelines, the direct impact this had on habitat suitability is unclear. A study by Glamore and Indraratna (2005) found that drain water quality could not meet ANZECC guidelines with tidal restoration, but there was still a significant improvement in pH, Fe and Al. Therefore, whilst metal concentrations may have been very high at times in the Little Broadwater during the study period, there may nevertheless have been a considerably improvement on loads prior to restoring tidal exchange.

Concentrations of total nutrients in Little Broadwater over the study period classify the wetland as a eutrophic to hypereutrophic system (Salas & Martino 1991). It is uncertain if nutrient loads are naturally high in this wetland system due to the lack of undisturbed reference sites to which it can be compared. Wetland nutrient concentrations are a function of water volume, water depth, land use, runoff, macrophyte biomass, phytoplankton biomass and sediment nutrient loads (Mitsch & Reeder 1991). The long-term agricultural use of Little Broadwater and the catchment, along with poor flushing, have most likely contributed to the high nutrient concentrations observed in the wetland. Reflooding the wetland may have also contributed to increased nutrient loads, as research by Portnoy (1999) has shown that saline restoration of wetlands can result in nutrient mobilisation and increased organic decomposition.

4.4.2 Spatio-temporal characteristics of water quality

Overall, CA and FA proved to be useful techniques to characterise the spatio-temporal variation in water quality at Little Broadwater. While FA did not reduce the number of parameters needed to define spatio-temporal variation at Little Broadwater, the resultant VFs grouped parameters based on common features, simplifying the examination and discussion of the influence of components on the total variation of water quality (Kowalkowski et al. 2006; Wunderlin et al. 2001). Initial CA of all cases indicated high temporal and spatial variability of within-wetland water quality, with salts accounting for most of the variance (41%). Water quality parameters associated with ASS oxidation (pH, Al, Fe and Mn) accounted for a further 19% of the variance. No seasonal patterns were identified in both the initial or temporal CA due to differences in rainfall patterns between years, changes in tidal exchange management, and the oxidation of ASS and subsequent acidification of the wetland in 2005.

General patterns of spatial and temporal variation were examined using sample month and site medians, which divided the study period into three phases – Saline, Fresh and Acidic – and the wetland into three spatial zones correlating with the preliminary zones identified from the pH and Cl^- maps, and the pilot study (Chapter 3). While using the median indicated average spatial and temporal patterns, the within-cluster variability was masked (Johnston 1993) and hence there was a loss of information about the complexity of water quality variability. This was particularly evident for Acidic-phase which only represented December 2005, although 40-50% of samples from February, June and August 2006 were included in the acidic cluster for CA of all cases. Using median values may have also given more weighting to individual parameters which then influenced clustering. For example, December 2005 samples were initially clustered into the fresh group for CA of all cases, however, temporal CA initially grouped December 2005 with the Saline-phase sample periods. Examination of the raw data showed that while median EC was lower than that for August 2005 (which was grouped into Fresh-phase), median concentrations of Ca, Mg and SO_4^{2-} were higher and thus appear to have influenced the clustering. Nonetheless, the salts component was significantly different between Saline- and Acidic-phase, as was the ASS component between Fresh- and Acidic-phase, further exemplifying the loss of within-cluster variability when using averages.

There was considerable variation between spatial zones in relation to average water quality, however, temporal variability within each zone was similar throughout the wetland with the exception of the southeastern region (LLB). The ULB zone, which covered the northern section of the wetland and a site along the western boundary, was the freshest zone and had little temporal variation of components. ULB could be divided into two sub-zones, with the far northern area characterised as more acidic, with higher metal concentrations and a very high nutrient load in comparison to the lower area. The near-neutral conditions in the lower region of ULB indicated that this area received more regular flushing by tidal exchange and/or had more consistent water cover to prevent ASS oxidation. However, the location of ULB in the upper wetland catchment and the consistent fresh conditions suggest that surface runoff after rainfall was the dominant mechanism for flushing. The MLB zone received the most tidal exchange, due to close proximity to the drain, and also had the deepest water which reduced the frequency of sediment desiccation and subsequent ASS oxidation. Therefore, MLB had the best water quality in terms of acidity, metals and nutrients with the least temporal variation.

In contrast, LLB was characterised as saline and acidic with high concentrations of metals and nutrients. The moderate correlation of TN and EC observed in the data was most likely due to this pooling of nutrient-rich and saline water in this region. LLB appeared to be a ‘sink’ area where water pooled and had little exchange with other areas of the wetland. Due to the poor exchange capacity of the zone, salts were concentrated by evapotranspiration and acidic water pooled for long periods of time. The high salt concentration and low pH may also be a function of the sediment in the area. Acid sulfate soils are often quite saline due to their location in the landscape, and can therefore contribute large amounts of Cl^- and other salts to the surface water through groundwater-surface water interactions. Another mechanism by which sediments and surface water interacted may have been through oxidation of ASS during reduced surface water cover. This may also lead to acidic salts being drawn to the sediment surface by evapotranspiration (Rosicky et al. 2006; Walker 1972; see Chapter 5). LLB relied on rainfall and surface runoff to flush salts and acidic water from this region. As a result of this cycle of evapotranspiration and then flushing due to rainfall, LLB had high temporal variability of all components. Nutrient concentrations increased considerably between December 2006 and February 2007, however, it is unclear why this occurred as there was high rainfall over this period which would have flushed nutrients from this area. Thus, while some processes of wetland functioning and flushing could be determined from this study, other processes were unclear from the data available and indicate complex functioning.

4.5 Conclusion

The considerable spatio-temporal variability of water quality at Little Broadwater was due to a combination of climatic variables, tidal exchange management and flushing characteristics of the wetland. Adjustments to the tidal exchange structures were usually in response to rainfall, which made it difficult to determine if changes in water quality were due to climate or management. Spatio-temporal variation of water quality was predominantly due to salinity, which was a function of rainfall, evapotranspiration, tidal exchange management and circulation. Acid sulfate soil oxidation products also contributed to the spatio-temporal variability and were influenced by surface water cover, rainfall and circulation. Spatial zonation based on water quality characteristics of the wetland indicated flushing and circulation patterns. The susceptibility of the wetland to desiccation and subsequent acidification was exemplified during the study, but provided the opportunity to graphically map improvements in water quality through management of tidal exchange.

Although acidic water was present in the wetland throughout the study period, the severity, duration and frequency of acid discharge events was greatly reduced. This was achieved through dilution of acidic water via tidal exchange, prior to discharge to the estuary, and reflooding of the wetland reducing the occurrence of ASS oxidation. However, the wetland was still susceptible to acid events during dry periods when tidal exchange was restricted due to high creek salinity, demonstrating the need for adaptive management.