Soil Indicators of Queensland Wetlands

Phase 1: Literature Review and Case Study

Australian Government

Queensland Wetlands Program

Queensland Government

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Authors

Sue-Ellen Dear and Terri Svensson

This is the final report for the Soil Indicators of Wetlands: Status, Margins and History project (WL NRM 03) of the Queensland Wetlands Programme, a joint initiative of the Australian and Queensland governments. The Queensland Wetlands Programme was established in 2003 to protect and conserve Queensland's wetlands.

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Enquiries regarding this document should be addressed to:

Daniel Brough
Department of Natural Resources and Water
Block C, 80 Meiers Rd
INDOOROOPILLY QLD 4068
Telephone: 07 3896 9333

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Executive summary

Soils are potentially powerful indicators of wetland dynamics because of the specific morphological features that develop in wet environments and therefore provide a reliable way to identify and manage wetlands (such soil morphology has been termed 'hydromorphic soils' throughout this text). In Queensland, and arguably Australia, the understanding of soil indicators of wetlands has not been adequately developed to support a regulatory and management environment. Further research into this area is warranted if wetlands are to be adequately protected, effectively utilised and/or restored.

This review of national and international scientific literature evaluates the ability of indicators often found within soils to reliably predict wetland boundaries in a defensible and scientifically robust manner. The use of soil indicators has been used extensively in the United States of America for the purpose of delineation. Consequently, much of the literature cited in this document is based on research and mapping of hydric soil indicators originating from the United States over the past ten years. This report also contains two case studies of wetlands in Southern and Central Queensland to test if predictable relationships between soil indicators and wetland status exist for Queensland wetland soils.

A number of significant findings from the literature review and case studies are included in this report. Key research findings include:

- soils are useful indicators of wetlands and have easily identifiable features
- a number of morphological and chemical features are correlated to wetland status
- soil indicators used in the United States may have applicability in Queensland, but they are not readily transferable
- additional investigations are required to assess relationships between soil indicators and periodically saturated wetlands.

Observations of vegetation or surface hydrology cannot be used to reliably predict the margins of a wetland. Due to the morphology that develops in reduced or waterlogged conditions, soils may potentially provide additional indicators to assist wetland delineation. A reliable set of Australian indicators of hydromorphic soils have not been developed or tested and this report relied on indicators devised for the United States that showed promise for use in Queensland. Further investigations are warranted to determine if the robust relationships between soil indicators and wetland status used in the United States are applicable to Australian soils.

The case studies in this report indicate that wetlands that are only periodically saturated appear to exhibit only a few indicators, largely based on the accumulation of organic matter. Such wetlands may require more extensive monitoring over a series of wet and dry periods to enable a more accurate prediction for wetland delineation purposes.

Regardless, the relationships predicted for a range of wetland types must be assessed against the changes that are evident across soil groups, landform, climate and vegetation associations. This should help facilitate the establishment of a more scientifically robust and defensible means of wetland delineation in Queensland, particularly one that may be used in a regulatory framework.

1. Introduction

Background

Soils are potentially powerful indicators of wetland dynamics because of the specific morphology that develops in wet environments. This may provide a reliable way to identify, and consequently manage, wetlands. Wetlands — and in particular their margins — can change over time, especially if influenced by human activities. Even in the well-watered lands of North America and Europe, observations of vegetation or surface hydrology will not by themselves answer the question of where a wetland begins or ends. Often this information is lacking or unreliable.

In Queensland, and arguably Australia, the understanding of soil indicators of wetlands has not been adequately developed to support a regulatory and management environment. Further research into this area is warranted if wetlands are to be adequately protected, effectively utilised and/or restored. Therefore, use of soil indicators requires the establishment of tested relationships between wetland dynamics and the changes evident in soils across geomorphic processes, landform, climate and vegetation associations.

For the purpose of this report, the emphasis was on identifying soil morphological characteristics that may help to define soils within wetlands. This project has concentrated on natural or undisturbed processes. Identification of the characteristics of soils that are unique to wetlands should allow differentiation between wetland and non-wetlands soils. With accurate field identification of wetlands soils, it is expected wetland margins can be identified and assist with regulation and protection. There is also the potential to provide validation and support to the wetland mapping projects and the wetland definition projects currently being undertaken under the Queensland Wetlands Programme.

The beneficial roles that soils play in a wetland environment are becoming increasingly recognised on a global scale, particularly in relation to these soils acting as a filter; a source and/or a sink for pollutants, nutrients and other mineral elements (such as acid, aluminium, iron, sulfate and other heavy metals); and issues associated with greenhouse gas emissions.

Observations of vegetation or surface hydrology will not by themselves answer the question of where a wetland begins or ends, but soils are particularly powerful indicators of wetland dynamics because the morphology that develops reflects their formation conditions. The literature suggests that the delineation of hydric soils is particularly useful in areas of unreliable or unavailable hydrology, in areas where the vegetation is transitional, has been cleared or in areas where the list of hydrophytic vegetation does not provide delineation assistance (Hurt and Brown 1995; Hurt and Carlisle 2001).

Soil characteristics that can be indicative of waterlogged conditions in the upper layers include redox potential, soil oxygen content and water table depth. However, morphological characteristics created by the accumulation of organic matter, the depletion and/or concentration of iron and manganese, and the reduction of sulfur and carbon, and geomorphic indicators are also of use to isolate wetland soils.

Research in the United States has developed hydric soil indicators across the country, implemented delineation or mapping programmes for ten years, and is now proving to be a reasonably reliable and extensively tested process. In most areas, the boundaries of wetlands for jurisdictional and restoration purposes can be reliably predicted using this research. While the legislation in the United States may not be as inclusive as previously administered (resulting from the Solid Waste Agency of Northern Cook County v. US Army Corps of Engineers (SWANCC) decision), it has been particularly useful for ensuring that many United States wetlands are protected and restored in an effective manner. There has been a reduced level of disturbance of coastal wetlands containing acid sulfate soils in the United States, with a preference for low impact structures, rather than excavations for canals. This is thought to be a reflection of the strength of wetland legislation (Ahern pers. comm. 2006), and the appreciation of the relationship to delineate the boundaries of wetlands in a defensible manner by an assessment of hydromorphic soil indicators. Such defensible legislation and an appreciation of the importance of the relationship between wetland soils and the protection of wetlands has been absent in wetland protection in Queensland to date.

1. Introduction

Objectives

This report is the product of a six-month project, funded by the Queensland Wetlands Programme, titled Soil Indicators of Wetlands: status, margins and history. The Queensland Wetlands Programme has the aim of advancing wetland conservation and management across Queensland and in Great Barrier Reef Catchments in particular. The stated objectives of the project were to:

1. Review national and international scientific literature on the morphological characteristics of soils within wetlands, and create the initial framework for field identification of the wetlands.
2. Review methods that are used nationally and internationally to identify wetland soils for regulation and protection purposes.
3. Investigate a case study of two contrasting wetlands in Central Queensland to illustrate the process and the research and development needed to establish a defensible regime.
4. Identify research gaps in this area in Queensland.

The definition of a wetland

The definition of a wetland varies worldwide in their content, meaning and application. For example, the definition used to define internationally significant wetlands from the Ramsar Convention[1] is:

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 6 m.

In Queensland, the agreed definition of a wetland, as used by the Environmental Protection Agency (EPA 2006) is:

Wetlands are areas of permanent or periodic/intermittent inundation, 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 6 m. To be classified as a wetland the area must have one or more of the following attributes:

- *at least periodically the land supports plants or animals that are adapted to and dependent on living in wet conditions for at least part of their life cycle*
- *the substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers*
- *the substratum is not soil and is saturated with water, or covered by water at some time.*

The combination of the waterlogged soil, hydrophytic vegetation and hydrology are used as diagnostic indicators of wetlands in Queensland. The association or connection between all three indicators needs to be recognised for any application of the definition, particularly for any regulatory uses.

Wetlands in Queensland

Wetlands are located throughout Queensland and include seagrass beds, tidal flats, mangroves, saltmarsh, wet heathland, shallow coastal inshore waters including coral reefs, freshwater and saline lakes, creeks, rivers, billabongs, lagoons, underground streams, springs, swamps and water storages such as dams and reservoirs. Almost four percent of the state, nearly 71,000 square kilometres, is permanently or periodically covered in water (EPA 2006) and these areas are therefore classified as wetlands. These figures will be reviewed as the current wetland mapping and classification programme is completed. A number of these wetlands are of national and international significance, with five wetlands listed under the Ramsar Convention and 210 recognised in A Directory of Important Wetlands in Australia — June 2005 (EPA 2006). Many of Queensland's wetlands are ephemeral, and thus remain dry for extended periods. Others are permanently inundated, including near coastal and marine environments.

[1]The Convention on Wetlands, signed in Ramsar, Iran in 1971, is a treaty that provides the framework for the conservation and wise use of wetlands and their resources.

The function of wetland soils

Wetlands provide a variety of ecological functions, especially hydrophytic vegetation that is capable of growing in wet, reduced soil conditions. Most plants cannot tolerate waterlogging, unless specific plant adaptations to the anaerobic soil conditions have evolved, such as oxygen transport from the aerial parts and anaerobic respiration. The oxygen brought to the roots through various processes is often released through pores in the rhizomes into the surrounding soil, creating an oxidised rhizosphere. This oxidation allows the plant to prevent the accumulation of reduced dissolved compounds, such as iron and manganese, which are toxic to most plants. Hydrophytic vegetation include *Melaleuca* species, mangroves and other water plants.

While the ecological importance of wetlands has been long recognised, they continue to be degraded, altered, drained, filled and destroyed often to facilitate urban, agricultural, industrial or tourist development. Sediment accumulation, alterations to water flow regime into or out of the wetland and changes to the quality of inflow waters can adversely affect the wetland leading to permanent changes in hydrology or species composition. Weed infestations also result in significant environmental impacts in wetland ecosystems. Many internationally significant wetlands have been destroyed according to Turner *et al*. (2000), including the:

- Aral Sea — due to water supply extraction upstream for cotton farming
- Wadden Sea, Netherlands — due to gas exploration drilling
- Florida Everglades, United States — due to nutrient inflows from reclaimed sugarcane areas
- Irish peatlands — due to excavation for fuel.

Globally, wetlands are under significant pressure. They are still considered by many to be of little or no value and the market has failed to protect them because of the public nature of several wetland goods and services. The lack of appreciation of the value of wetlands and their subsequent low priority in decision-making processes has resulted in their loss and extensive modification, causing a social cost that is still not recognised (Turner *et al*. 2000).

The potential for utilising microbial activity of wetlands to break down detergents and pesticides is an area of interest for pollution control. Biodegradability of most detergents and pesticides in anaerobic media is poorly understood, but it may be used in conjunction with aerobic conditions to enhance decomposition (Ponnamperuma 1972). Heavy metals may be removed from inflowing water by sedimentation and can precipitate on the clay and organic fractions of the soil (Patrick 1994).

The microbial conversion of nitrogen compounds to nitrate in the oxygenated section of the wetland system, and the subsequent microbial reduction of nitrate to nitrogen gas when it moves into anaerobic zones of the wetland, is a beneficial aspect of wetland soils (Patrick 1994). The effects of oxidation, reduction and adsorption cycles on the mobility of nutrients are of significance in relation to the wetland soil as a source or sink for pollutants (Willet 1983). Phosphorus can also be removed by sedimentation within the wetland where water flow is reduced during its passage through vegetation, allowing the phosphorus-rich sediment to be no longer suspended. The oxidised soil–water interface can act as a sink for phosphate and other plant nutrients and as a chemical barrier to the passage of some nutrients from soil to water. The sediment in wetlands can intercept and hold large influxes of dissolved phosphorus and slowly release it as needed by algae and other hydrophytic plants (Harter 1968, Patrick 1994, Ponnamperuma 1972).

Greenhouse gases, such as methane and nitrous oxide, are of particular relevance for wetlands. Methane is responsible for approximately one-fifth of the enhanced greenhouse effect and has a warming potential more than 20 times greater than carbon dioxide on a volume basis (DEH 2006). Methane is produced as a result of organic matter decomposition in reducing environments such as wetlands and is thought to account for approximately 25 percent of the global methane flux (US Climate Change Programme 2006). The production of methane is controlled by the flow of carbon and electrons to the microbial community of methanogens during the process of anaerobic organic matter degradation. Recently, the potential of using electron acceptors to stop methane production and emission from flooded rice soils has been identified as an area worthy of investigation for reducing methane production from wetland rice soils (Sahrawat 2004). Yu and

1. Introduction

Patrick (2004) concluded that both nitrous oxide and methane production can be minimised within a wetland rice soil but soil must reduce enough to favour complete denitrification, but not be so reduced that methanogenesis is initiated. Such a situation can theoretically be simulated with manipulations to irrigation, organic matter and fertilisation regimes. Yu and Patrick (2004) also suggest that the capacity of these soils as carbon sink to reduce greenhouse gas emissions is limited — and unlikely on a long-term basis — as natural or human-induced disturbance of these areas may lead to higher future emissions (Intergovernmental Panel on Climate Change (IPCC) 2001).

Peatlands contain reservoirs of carbon that are significant on a global scale and in their undeveloped state act as sinks for atmospheric carbon dioxide (Lafleur *et al*. 1997). The disturbance of peatlands may lead to exacerbated carbon losses, while undisturbed or restored peatlands may have a major role in moderating climate change (Holden 2005). The potential impacts of climate change on wetland hydrology, and consequently wetlands soils and vegetation, is an area that has not attracted significant research to date. It has been predicted, however, that repeated drought cycles have the potential to affect the structure and function of mangrove forests and upstream ecosystems (Drexler and Ewel 2001).

Acid sulfate soils are soils or sediment containing highly acidic soil horizons or oxidated iron sulfides. These are known as 'actual acid sulfate soils'. Soil with sulfides or other sulfidic material that has not been exposed to air are known as 'potential sulfate soils'. Acid sulfate soils are found predominantly in coastal wetlands — generally below 5 m Australian Height Datum (AHD)[2] in Australia — in aqueous environments rich in organic matter and have available sulfate ions, iron and sulfate reducing bacteria. Under reducing conditions, sulfide and Fe(II) react to form FeS and ultimately FeS_2. However, if these soils are exposed to oxygen (for example, upon drainage) the sulfides are oxidised to sulfate and the pH may drop to 2.5. The monosulfide form can sorb heavy metals that can be released into the wetland if these soils are disturbed. The presence of acid sulfate soils within a wetland requires specific management, particularly if the wetland is drained or disturbed. Inland acid sulfate soils are found in other states of Australia in wetland and saline situations, but the extent in Queensland is largely unknown.

The importance and function of wetlands have been recognised extensively in the scientific literature (EPA 2006, Davis and Negus 2006, Environment Australia 2001; Blackman *et al*. 1999; Finlayson *et al*. 2001; Mitsch and Gosselink 1993; Paijmans *et al*. 1985). In summary, wetlands:

- play a role in the containment and release of floodwaters
- assist in groundwater recharge and water storage
- protect coastal areas from wave action and erosion
- filter nutrients, pollutants and sediments and improve water quality
- provide habitat, a food source and breeding grounds for fish, crustaceans, domestic and migratory waterbirds, frogs, tortoises, reptiles, insects and other aquatic flora
- have a role in biodiversity conservation
- are socially, aesthetically and historically significant and have cultural heritage values for indigenous Australians.

Artificial wetlands constructed for wastewater treatment can also have many of the above functions if designed correctly. Wetland soils also perform a variety of roles that are of significance both locally and globally in terms of ecological functions and act as both source and sinks for pollutants, nutrients and other mineral elements.

[2]Australian Height Datum (AHD) is the datum for which all vertical control for mapping is to be referred.

The importance of wetland rehabilitation

Wetlands are a vital element of national and global ecosystems and economies. The disruption of wetland functions has a high cost economically, socially and ecologically. Disturbance of their natural balance can destroy gene pools required for medical and agricultural purposes, affect their ability to naturally improve water quality and spoil their use for educational and recreational purposes. There are a number of good reasons for taking up restoration and rehabilitation activities in degraded wetlands. These are essentially the same reasons for conserving natural wetlands, the valuable functions and services they provide.

When the Ramsar Convention was developed and signed by the first member countries in 1971, few wetland restoration activities had taken place. Moller (1999) stated that wetland destruction was prolific at this time and all efforts were devoted to the conservation of wetlands that were still existing.

Subsequently, restoration and rehabilitation of wetlands became a theme at the Fourth (1990) and Sixth (1996) Conferences of the Contracting Parties (COP), where recommendations 4.1 (Montreux) and 6.15 (Brisbane) urged member states to consider and promote wetland restoration. By March 1999, it was evident global attitudes toward wetland rehabilitation were changing, as 75 out of 107 national reports submitted by the Contracting Parties for COP7 indicated wetland restoration or rehabilitation was taking place in that country (Moller 1999).

Despite the fact that it is often difficult, perhaps impossible, to restore wetlands to their former condition, there are now many examples of restoration projects in which at least some functions and values are being re-established. Some of these examples include the:

- The Great Barrier Reef Coastal Wetlands Protection Pilot Programme, Australia
- Waza Logone Floodplain, Cameroon
- Katarapko Lagoon and other wetlands in Australia
- Lake Karla in Greece, where salt water intrusion was prevented
- Louisiana River Delta in the United States, where major river deltas have been restored.

Artificial wetlands constructed for wastewater treatment can have many of the functions of a natural wetland if properly designed. Wetland soils also perform a variety of roles that are of significance both locally and globally in terms of ecological functions and act as both source and sinks for pollutants, nutrients and other mineral elements.

2. Review of wetland hydrogeomorphology

Waterlogged soils form in certain landscape positions and landforms, with concave landforms retaining more water than convex landforms. The landforms include depressions, floodplains, toes of slopes, drainage paths, seepage slopes and occur along the margins of coastal and inland waterbodies. In humid climates, broad and flat terrain tends to favour the formation of wetlands, particularly when drainage outlets are missing (Tiner 1999). Fine-textured, clayey and silty soils retain more moisture in the small soil pores than coarse- textured soils with good internal drainage, and consequently wetlands can even form in fine-textured soils on sloping landscapes in humid areas. Wetlands with sandy soils are generally created due to poor external drainage, high water tables or are located adjacent to a water body where they are regularly inundated or have a high groundwater levels (Tiner 1999).

Hydrologic processes

The term 'wetland hydrology' as defined in the US Army Corps of Engineers (1987) Wetlands Delineation Manual encompasses all hydrologic characteristics of areas that are periodically inundated or have soils saturated to the surface at some time during the growing season (see Duration of inundation or saturation, page 13 for more information on growing season).

Water (at the surface, beneath and adjacent to a wetland) is the primary factor controlling a wetland environment. Soil type and hydrophytic plant communities are primarily influenced by water in a wetland, which may be present as permanent or seasonal flooding and ponding, or it may simply saturate soils near the surface with no free water present.

The formation, persistence, size and function of wetlands are controlled by hydrologic processes. In turn, the hydrologic processes can be influenced by the landscape position, underlying geology, precipitation patterns, groundwater relations, surface water run-off and tidal action (Tiner 1999).

To understand wetland function it is important to recognise the source and movement of water. Linkages between wetlands, uplands and deepwater habitats provide a framework for protection and management of wetland resources. Which processes can and will occur within the wetland are determined by the characteristics of the water entering the wetland, and the characteristics of the wetland itself (i.e. size, shape, soils, plants and position in the landscape or 'geomorphic setting').

Hydrologic processes occurring in wetlands are the same processes that occur outside of wetlands and collectively are referred to as the hydrologic cycle.

Climate and weather

As the hydrologic cycle and climate are intertwined, changes in climate are likely to bring changes to the hydrological cycle. The term 'climate' is generally used when seeking to explain global, regional and other long-range changes or patterns. Weather, on the other hand, can be defined as the individual state of the atmosphere for a given place over a shorter time period (Richardson *et al*. 2001).

Long-term drought and pluvial periods can often distort the distinction between climate and weather. The hydrology of a wetland during a decade-long drought could easily be perceived as the 'norm', especially in the case of seasonal wetlands or in wetlands of hydrologically altered areas. Richardson *et al*. (2001) suggest the principal difficulty is one of context and ask 'is the period in question characteristic of normal conditions or not?'.

Hydrologic cycle

The endless circulation of water between solid, liquid and gaseous forms is called the hydrologic cycle. Major components of the hydrologic cycle are precipitation, surface water flow, groundwater flow and evapotranspiration. The atmosphere, rivers, lakes, wetlands, groundwater and adsorption surfaces (interception) serve as temporary storage components of the cycle.

The water balance of an individual wetland is a fundamental, unique and distinctive property in which gains equal losses. That is, the sum of precipitation, run-off and groundwater discharge (the inputs) are equal in magnitude to the sum of evapotranspiration, surface flow, and groundwater recharge (the outputs), plus or minus a change groundwater and surface water storage. If inputs exceed outputs, balance is maintained by an increase in storage (i.e. water levels in the wetland rise).

If outputs exceed inputs, balance is maintained by a decrease in storage or water levels fall.

The water balance of a wetland, in conjunction with information on the local geology, provides a basis for understanding the functions and hydrologic processes of a wetland.

The soil hydrologic cycle

The soil hydrologic cycle is defined as the portion of the global hydrologic cycle that includes progressively more detailed examination of water movement on and in the landscape.

Precipitation that falls on the landscape is the ultimate source of water in the soil hydrologic cycle. Precipitation, which is infiltrated, percolates along positive hydraulic gradients until the gradient decreases to zero, whereupon movement stops. This movement is reversed via unsaturated flow, as water is removed by evapotranspiration, or water movement continues until the wetting front merges with the water table.

Plants are important in increasing infiltration and decreasing run-off and erosion (Bailey and Copeland 1961). Vegetation creates more porous soils by both protecting the soil from pounding rainfall, which can close natural gaps between soil particles, and loosening soil through mechanical action. A lack of vegetative matter on the soil surface can lead to the detachment and erosion of soil particles, surface sealing and impede the infiltration process.

Shallow but extensive transient and saturated groundwater flow systems can form in sloping upland soils in a wetland's catchment because of the influence of a permeable surface combined with the presence of slowly permeable subsoils. Slowly permeable subsoil horizons include:

- argillic horizons that have accumulated extra clay
- indurated pans such as duripans
- coffee rock fragipans which are cemented with low permeability.

Lateral flow through the more permeable surface soil is relatively unrestricted and is driven by a hydraulic gradient produced by the sloping ground surface within a wetland's catchment. The water in this transient groundwater system flows slowly downslope, a portion of which may be discharged to the soil surface upslope of the wetland as reflow, a component of run-off. Another portion is discharged to the wetland through seepage at or about the water level of the wetland.

Concave hillslopes, particularly those that are concave in more than one direction, tend to concentrate overland flow, thus maximising throughflow, interflow and reflow. During precipitation events, the saturated zone that contributes to the reflow increases in area upslope. These saturated areas are potential sites for the genesis of hydromorphic soils.

Groundwater

Groundwater is the portion of free water located beneath the surface of the earth that fills pores and cracks in soil and rock formations. It is an important component of the hydrology of wetlands and originates as precipitation, as seepage from surface waterbodies or is the part of the soil hydrologic where the wetting front merges with the water table.

Groundwater generally moves very slowly, movement or flow takes place from a recharge area to a discharge area. The rate of groundwater flow is affected by topographic position (the difference between recharge and discharge points), the permeability of rock formations and regolith, the distance over which the water is moving and sediment and soil characteristics (Stone and Bacon 1994).

The dynamic nature of groundwater flow strongly influences the intensity and rate of soil chemical and physical processes that leave numerous morphological indicators in the soil. Thus, in addition to the presence or absence of a high water table in a soil, knowledge of the direction, magnitude, and rate of groundwater flow is necessary to place the morphological characteristics of hydromorphic soils in the context of a wetland and its landscape (Richardson *et al*. 2001).

The degree of connection between surface waterbodies by way of groundwater flow is controlled to a large extent by the hydraulic conductivity of the geological materials through which the groundwater flows (Winter *et al*. 2003). In some types of carbonate and volcanic rocks, groundwater flows freely through large rock openings, such as solution channels in limestone, as if through pipes. Conversely, groundwater moves through clay at an almost undetectable rate. The rate of groundwater movement is also controlled by factors such as the porosity of earth materials and the hydraulic gradient of groundwater flow systems.

Recharge and discharge

Wetlands can play an important role in the function of groundwater recharge. In most instances, the downward movement of rainfall, snowmelt, or surface water through the soil and upper layers of the unsaturated soil profile is the principal mechanism for recharging groundwater. Recharge rates in wetlands can be much slower than those in adjacent uplands, if the upland soils are more permeable than the slowly permeable clays or peats often found in wetlands.

Conversely, Carter (1996) suggests that wetlands are most commonly groundwater discharge areas; that is, where upward or lateral movement of groundwater discharges at the ground surface. Groundwater usually discharges into waterbodies such as wetlands and rivers for the sole reason that they tend to occupy topographically low areas in the landscape.

Wetland groundwater recharge or discharge incidence and position is affected by topographic position, hydrogeology, sediment and soil characteristics, season, evapotranspiration, and climate and may not occur uniformly throughout a wetland.

Using hydrogeology to measure wetland status

Given the reliance on the presence of free water for an area to be classed as a wetland, measurement of the water table is thought to be a useful method of detection for the presence of a wetland. A water table has been defined by the USDA (1985) *as the zone of saturation at the highest average depth during the wettest season that is at least 152 mm thick and persists for more than a few weeks, and by the SSA (2001) as the upper surface of the groundwater or that level in the ground where the water is at atmospheric pressure.*

In addition, it has been reported that redoximorphic features (see The depletion and/or concentration of Fe and Mn, page 13) do not illustrate the depth of a seasonally high water table. Instead, these features show where the soil has been saturated for a period long enough for anaerobic conditions to develop (Vepraskas and Caldwell 2006). Consequently, the presence of redoximorphic features cannot be used to provide an accurate estimation of the depth of a seasonal high water table.

Piezometers

An accurate measure of water table depth can be made with the use of correctly installed piezometers. However, the recommended measurement period is at least one year, with a minimum measurement period capturing a dry–wet–dry cycle (USDA 2006). Piezometers are monitoring wells that consist of a section of unslotted pipe that is open at both ends or slotted only at the bottom. The construction of piezometers often requires:

- backfilling with sand
- surface sealing with bentonite to prevent surface or near surface inflow
- a need to take into consideration the presence of any confining layers.

Hydraulic heads from at least two piezometers installed in different locations within the one wetland can provide the information necessary to quantify the direction of groundwater flow. Consequently, the accurate measurement of groundwater is usually a costly procedure. Unless continuously monitored using electronic equipment such as data loggers, the data gathered is generally of limited use.

Tensiometers

Tensiometers measure soil matric potential at a given depth in the soil profile, with saturated conditions being represented by soil water pressure heads ≥0 cm. They require no calibration and they measure the actual availability of water in the soil. In theory, piezometers and tensiometers at a given depth should: a) provide an indication of the same soil saturation duration; and b) measure equal positive soil water pressure heads when the soil is saturated. However, Thompson and Bell (1998) noted moderate differences between the two. It was hypothesised that such differences were due to the existence of near-saturated conditions above the water table that cannot be measured by the piezometer, but are reflected in the tensiometer data.

Water table wells

Water table wells also measure water table depth and flow. Monitoring wells commonly consist of a plastic pipe slotted along a portion of its length and placed in boreholes dug below the water table. The pipes are not sealed, reflecting water contributions from all sources including rainfall, surface flow and groundwater. When water table wells are installed at the same location as one or more piezometers, vertical direction of groundwater flow can be determined by comparing elevations of water levels in the wells (Carter 1996).

Stagnant or no flow conditions result from there being no difference in water elevations. Groundwater recharge is indicated by the water level in the piezometer being lower than that in the water table well, hence water flow is downward. The reverse, indicates groundwater discharge, or upward flow is occurring.

Duration of inundation or saturation

Saturation to the surface for some period is an apparent requirement for wetland hydrology. Direct use of this parameter for delineating wetlands in the field is difficult and costly, and varies greatly between regulating agencies in the United States.

The US Army Corps of Engineers Environmental Laboratory suggest soils need to be inundated and/or saturated at the surface for 5 to 12.5% of the growing season in order to support a wetland environment, and areas that are only intermittently or never inundated or saturated (i.e. less than 5% of the *growing season*, will not support wetland creation).

Equally, the USDA (1985) stated that areas would need to be inundated for 7 to 15 days, or 10% of the growing season, to support a wetland system. Contrary to the above, Faulkner *et al*. (1989) found that annual surface saturation between 14 and 28 days may induce sufficient anaerobic conditions to develop hydric soil morphology.

To accurately assess the hydrology of a site, several years of records are necessary; this however, is rarely available. Determination of the hydrologic status of a site is usually based upon secondary indicators and short-term saturation records, such as the presence of hydromorphic soils and hydrophytic vegetation within an appropriate topographic setting. Consequently, short-term saturation monitoring to determine if wetland hydrology exists for a given site is often questionable (Skagg *et al*. 1991).

Secondary measures

Vepraskas *et al*. (2004) documented the limitations of identifying wetland hydrology indicators and proposed that if two secondary indicators of hydrology can be identified, these may be substituted for a primary measure of hydrology. For example, a primary measure of hydrology is the frequency and duration of saturation of soil horizons while secondary measures include an oxidised rhizosphere, the use of soil survey data on water table depths or measurements of the abundance of certain plants. The use of secondary indicators of hydrology has not been verified (Vepraskas *et al*. 2004).

Calibrated simulation models

The use of calibrated simulation models to determine saturation frequency and duration for individual horizons in conjunction with historic rainfall data has been used in the United States to compute daily water table levels over extended periods (He *et al*. 2003; Vepraskas *et al*. 2004). The use of models is based on the assumption that saturation data could be correlated to the abundance of redoximorphic features of soils within the area to predict the saturation frequency and duration in similar soils simply by measuring the percentage of redoximorphic features at a certain depth. These models require further application and calibration of all hydromorphic soil field indicators to wetland hydrology requirements for a wide variety of sites (Vepraskas *et al*. 2004).

2. Review of wetland hydrogeomorphology

Tiner's methods of observation

Tiner (1999) summarised some observations that can be used to identify flooded or ponded soils:

- water carried debris or drift lines
- sediment deposits
- scoured or bare areas
- wetland drainage patterns reflect the movement of the water
- moss lines on tree trunks where mosses and lichens that live above and below the flood line can be identified
- plant morphological adaptations such as buttressing
- water marks
- water stained leaves
- surface films of iron crust on the ground.

These indicators simply provide evidence that an event has occurred and provide limited information regarding the frequency or duration of inundation. Measurements of rainfall are also of relevance when wetlands are being delineated.

Geomorphic indicators

Geological features and past patterns of weathering make some landforms more favourable to wetland creation and subsequent production of hydromorphic soils.

Topographic depressions, slope breaks, areas of stratigraphic change, geological faulting, floodplains and other low land landscape areas are all examples of major hydrogeological settings that favour wetland formation in Queensland.

Topographic depressions

One of the strongest controls of the surface water balance for a wetland is topography. Run-off, in particular, is strongly controlled by topographic factors, including slope gradient, which influences the kinetic energy of run-off, and slope length, which influences the amount of water present at points on the landscape (Richardson *et al*. 2001).

Landforms consist of slopes that have distinctive morphologic elements with widely differing hydraulic characteristics. Surface water content progressively increases downslope as run-off, deep drainage and lateral subsurface flow from upslope positions is added to that of downslope positions. High slope gradients, convex curvature and relatively low soil water content generally characterises the highest position.

Footslope and toeslope positions are characterised by maximum water content and low slope gradients. Footslopes and toeslopes in concave positions are logical locations for wetlands because they occur in areas of maximum water accumulation and discharge (Carter 1996).

Slope discontinuities

The water table sometimes intersects the land surface in areas where the land is sloping, often corresponding to an upward break or change in slope (Winter and Woo 1990). Where groundwater discharges to the land surface, wetlands form. This constant groundwater seepage maintains soil saturation and permits establishment of wetland plant communities. Wetlands of this nature are found throughout the Cooloola Coast and Noosa region (refer to Thompson and Moore 1984, who studied various aspects of landscape dynamics in subtropical coastal dunes in the Cooloola region).

Subsurface stratigraphy

Groundwater movement is affected where stratigraphic changes occur near the land surface, resulting in the layering of permeable and less permeable rocks or soils. When water flowing through the more permeable rock encounters the less permeable rock, it is diverted along the surface of the less permeable rock, potentially toward the land surface. Any continual seepage that occurs at the surface provides the necessary moisture for a wetland to form.

2

Classification of the hydrogeomorphology of wetlands

In Australia, many regional classifications of wetlands exist (Goodrick 1970, Stanton 1975, Paijmans *et al*. 1985, Blackman *et al*. 1999), with Pressey and Adam (1995) summarising wetland classifications for Australia and founding a wide diversity of ideas on survey and classification. Brinson (1993) noted, however, that many wetland systems fail to address certain abiotic (hydrogeomorphic) features that are directly linked to wetland functions.

As a result, Brinson (1993) developed the hydrogeomorphic (HGM) approach to aid in performing functional assessments of wetlands. The system used the first principles of geomorphology, hydrology and hydrodynamics to separate wetlands into functional classes at a gross level. The HGM method recognises seven broad geomorphic settings, three potential water sources and three hydrodynamic categories. The seven geomorphic categories are listed in Table 1. The three water sources are precipitation, overland flow and groundwater discharge, while the hydrodynamic categories embody the strength and principal directions of flow: vertical fluctuation, unidirectional horizontal flow and bi-directional horizontal flow.

Novitzki (1979) devised a system to classify the hydrologic characteristics of Wisconsin's wetlands with regard to water source and landform, recognising four components:

1. surface water depressions
2. groundwater deressions
3. groundwater slope wetlands
4. surface water slope wetlands.

Surface water depressions receive precipitation and overland flow, whereby losses are through evaporation and downward seepage into a surficial aquifer. Groundwater depressions intercept the water table, so they receive ground water in addition to direct precipitation and overland flow. Groundwater slope wetlands occur where there are breaks in slopes, have an outlet and they also tend to occur on slopes where the groundwater has stronger flow than would normally be encountered in depressions. Surface water slope wetlands receive water from lake or river flooding and the water can readily drain back into the lake or river as the stages fall.

Table 1: Hydrogeomorphic classes of wetlands showing associated dominant water sources and hydrodynamics (source: Brinson *et al*. 1993).

Hydrogeomorphic class	Dominant water source	Dominant hydrodynamics
Riverine (along rivers and streams)	Overbank flow from channel	Unidirectional, horizontal
Depressional (within topographical depressions)	Return flow from groundwater and interflow	Vertical
Slope	Return flow from groundwater	Unidirectional, horizontal
Mineral soil flats (broad, wetlands with inorganic soils)	Precipitation	Vertical
Organic soil flats (extensive peatlands)	Precipitation	Vertical
Estuarine fringe (tidal wetlands)	Overbank flow from estuary	Bidirectional, horizontal
Lacustrine fringe (lakeshore wetlands)	Overbank flow from lake	Bidirectional, horizontal

3. Review of wetland soils

The definition of wetland soils

Hydromorphic soils are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers of the soil profile. As a result of a prolonged saturation period, anaerobic and/or saturated conditions will influence the genesis of the wetland soil as well as the wetland plant species that will grow (Hurt and Carlisle 2001).

Traditionally, descriptions of wetland soils tend to focus on chemical processes and observations that occur, or have occurred, in the upper layers of the soil profile as a wetland soil must also support hydrophytic plants. In the United States, the term 'upper part' of the current definition relates to the root zone (Mausbach and Parker 2001), which is generally reported as the upper 0.3 m of the soil profile. Consequently, a soil will not be considered hydromorphic if the morphological indicators of reduction occur below 0.3 m in the soil profile (Vepraskas 2001).

By definition, a soil that has saturated upper layers but has unsaturated deeper soil layers due to the presence of impermeable layers restricting water infiltration would be classified as a wetland soil. Conversely, in situations where subsoils are reduced and the corresponding surface horizons are unable to support hydrophytic vegetation, due to the presence of an oxidised root zone, the soil would not be classified as hydromorphic.

Chemical processes in submerged soils

The flooding of an air-dry soil causes various electrochemical changes to take place. The dilution of the soil solution will increase pH in acid soils, decrease electrical conductance and alter the diffuse double layer of colloidal particles. These changes are, however, fairly insignificant when compared with the chemical changes caused by reducing conditions in the soil (Ponnamperuma 1984).

According to several authors, including Vepraskas and Faulkner (2001), there are four conditions required for a soil to become anaerobic and support reducing reactions. The four conditions are:

1. the soil must be saturated to the point of excluding atmospheric oxygen
2. the soil must contain a source of organic matter that can be oxidised or decomposed
3. there must be a population of respiring microbes that will oxidise the organic matter
4. water must be stagnant or moving slowly.

These four conditions are explained in detail in the sections that follow.

The exclusion of atmospheric oxygen

Oxygen, and lack thereof, plays a major role in soil chemistry. In aerobic soil conditions, where soil oxidation occurs, atmospheric oxygen diffuses readily through the gas-filled pore spaces of the soil profile and is sufficient for soil and plant root respiration. In comparison, wetland soil profiles are generally saturated, more or less continuously, for prolonged periods in most years. The gaseous diffusion of oxygen into the soil profile is significantly limited and is quickly consumed due to respiration by aerobic organisms (Faulkner *et al*. 1989; Gambrel and Patrick 1978; Mitsch and Gosselink 1993; Ponnamperuma 1972). When oxygen is absent, some soil micro-organisms produce toxic substances that cannot be tolerated by most plants, with the exception of hydrophytic plants (Patrick 1994).

A source of organic matter and respiring microbes

The soil must contain a source of organic matter and a population of respiring microbes to oxidise it. The respiring bacterial population is widespread, abundant and adapted to function in the climates in which they occur (Vepraskas and Faulkner 2001). In saturated conditions, a new population of anaerobic micro-organisms will become established and reducing conditions will be created in the order of several hours to a few days after the soil has been inundated with water (Gambrell and Patrick 1978). Submerging a well-aerated soil can result in a surge of microbial activity that may result in the rapid development of strong reducing conditions, especially when there is a large source of organic energy available (Gambrel and Patrick 1978).

In an aerated soil, the decomposition of plant residues occurs with the assistance of a large group of micro-organisms and soil fauna. Due to the high-energy release that is associated with aerobic respiration, decomposition proceeds rapidly, with end-products such as carbon dioxide (CO_2), nitrate, sulfate and humus. In a submerged soil, the decomposition of plant residues is almost entirely done by facultative and obligate anaerobes. Anaerobic bacteria operate at a lower energy level of fermentation than aerobic bacteria and thus decomposition is much slower, resulting in the greater accumulation of organic matter in submerged areas. The end-products include CO_2, hydrogen, methane, ammonia, hydrogen sulfide and partially humified residues (Ponnamperuma 1972, 1984). Iron sulfides can also accumulate under certain conditions.

Stagnant or slow moving water

The final precursor for a soil to become anaerobic is the requirement that water within the soil be stagnant or slow moving. Water that is moving can delay the onset of reduction, particularly the reduction of iron, because oxygen is difficult to deplete (Vepraskas and Faulkner 2001). Cogger and Kennedy (1992) stated that while the velocity of shallow groundwater on sloping sites may affect reduction it would be expected to contain more oxygen than the slower moving groundwater in flatter landscape positions. Evans and Franzmier (1986) reported that as slope steepness increases, or horizontal distance to a steeper slope decreases, the velocity of lateral water movement within the soil was expected to increase. Increased velocity of the groundwater was reported to have caused, at least partially, an increased oxygen content of the soil-water. Where water movement is sufficient, the development of peat can also be inhibited. The process of decomposition of humus occurs at a faster rate and decomposers such as earthworms, insect larvae and other arthropods may flourish with the soil remaining largely mineral (Gilman 1994).

Oxidation and reduction within hydromorphic soils

The unique characteristics of hydromorphic soils are formed by oxidation and reduction process in the soil. Oxidation reactions are more likely to occur in well-drained soils, while reduction reactions are more likely to occur in poorly drained soils, or in situations where excess water is present. Cyclic fluctuations between oxidising and reducing conditions can be common in wetland soils, creating unique morphological characteristics. The accumulation or loss of iron, manganese, sulfur or carbon compounds resulting from oxidation and reduction will cause chemical changes to the soil. Reduction of elements such as iron and manganese can occur in saturated conditions, leaving soil morphological evidence such as the presence of mottles, segregations, gleyed soils and colour. The reduction of sulfate that can occur in strongly reducing conditions will result in the generation of hydrogen sulfide[3] (H_2S) and the formation of iron sulfides. Wetland soils can also contain high levels of organic carbon, which accumulates because anaerobic conditions impede the decomposition of organic matter. The oxidised constituents that dominate an aerobic soil, such as iron, manganese and sulfate, Fe(III), Mn(IV) and SO_4^{2-} respectively, virtually disappear in submerged soils being replaced by their reduced counterparts, Fe(II), Mn(II) and H_2S. Physical changes to the soil may also occur upon waterlogging, because of the pore spaces of the soil being filled with water. This can cause some finer textured soils to expand, thus further reducing the permeability of the soil (Tiner 1999).

Most of the reducing reactions will be reversed if the submerged soils are drained and this too influences some of the soil properties, particularly as a medium for plant growth. For example, when a soil is submerged and reduced, base cations are displaced from cation exchange sites on clay by reduced Fe(II), resulting in the loss of cations from the clay. Upon drainage of the soil, the reduced iron is re-oxidised and precipitated. As Hydrogen (H+) ions are now the only major cation, the soil becomes acidified and the clay will disintegrate (Ponnamperuma 1972). This process is unlikely to occur in carbonate-rich soils or soils receiving large amounts of basic salts with floodwaters (Mooremann and Van de Watering 1985).

Soil morphology as an indicator of hydromorphic soils

Over the last 10 years in the United States, the saturation and reduction of hydromorphic soils

[3]Hydrogen sulphide is a colourless, toxic and flammable gas responsible for the foul odour of rotten eggs.

3. Review of wetland soils

has been routinely inferred on the basis of soil morphology. Hurt *et al*. (2003) authored *Field Indicators of Hydric Soils* in the United States, to be used as a field guide to the identification and delineation of hydric soils in the United States.

The USDA defines hydric soils as *soils that have formed under conditions of saturation, flooding or ponding long enough during the growing season to develop anaerobic conditions in the upper part of the soil.*

The method of delineating wetlands on the basis of soil morphology is cheap, qualitative and is most conclusive in soils that are poorly drained — that is, in soils that are subject to seasonal ponding and/or perched water tables, and can remain wet for several months (Vepraskas 1999; Vepraskas and Wilding 1983). A similar system in Australia may assist in providing a more consistent means of identifying wetlands, particularly the margins of wetlands where the presence of hydrophytic vegetation and hydrology indicators are not always reliable, predictable or present. This may occur in examples where vegetation has been cleared or where quantitative hydrological data may not be available. Even in the United States, these subjective indicators are only indicative within the local landscape and climate context and cannot be used for a quantitative assessment of soil wetness without additional measurements, such as the groundwater dynamics and degree of reduction (Mooremann and Van de Wetering 1985). Morphological indicators of soil wetness have been discussed extensively in the literature (Hurt *et al*. 2003, Vepraskas 2001, Vepraskas and Guertal 1992) and those of relevance to Queensland are discussed below.

Accumulation of organic matter

The level of organic matter in the soil is a broad indicator of soil condition and its concentration is largely determined by the addition of surface litter (fallen leaves, manure and dead organisms), root material and the rate at which microbes break down organic compounds.

Organic matter can accumulate in the horizons of soils that are permanently or frequently waterlogged to form peat layers of fibric, hemic or sapric material and other surface horizons rich in organic matter such as O horizons and humose horizons (see Appendix 2). This occurs because anaerobic oxidation is slower than aerobic oxidation due to less efficient microbial action and waterlogged conditions hinder the aerobic breakdown of the bulky organic materials such as leaves, stems and roots. The net result is the accumulation of organic matter (Tiner 1999, Willet 1983). The preservation of sapric material requires long-term anaerobic conditions to retard decomposition (Mitsch and Gosselink 1993). The accumulation of organic matter can influence vegetation types, plant growth rates and soil nutrient status.

Measuring soil organic carbon provides an estimate of the amount of organic matter in a soil. Soil organic carbon varies with depth, and levels are usually highest in the topsoil where they decrease rapidly with depth. It has been reported that when soil organic carbon levels are greater than 0.03 g/g, a saturated soil will become anaerobic in approximately three days. In a situation where organic carbon levels are less than 0.03 g/g, then anaerobic conditions tend to develop after 3 to 50 days of saturation. Longer durations are needed as the organic carbon level decreases (Vepraskas and Caldwell 2006). Couto *et al*. (1985) reported a situation where reduction did not occur in the field in the lower part of the soil profile even when a high water table was present. This was reportedly caused by a lack of an energy source for microbial activity. The organic matter present in the lower profile was too stable and/or too low in available nutrients for the reduction process to take place during the period of soil saturation.

Dark colours typically value 3 or less and chroma 1 or less, generally occur in situations where there is organic matter present in the soil. Wet mineral soils, those soils without large amounts of organic matter, may develop dark colours due to the influence of the organic matter. In general, the darker the soil the more organic matter is present (Tiner 1999). Vertosols (Isbell 1996) are soils that have open cracks at some stage of the year. During these periods, surface soil can fall into cracks and small amounts of organic matter can influence the soils dark colour deep in the profile. However, it is predominantly the calcium-dominant montmorillinite clay that gives a Black Vertosol its organic colour. Naturally, this can occur in non-Aquic Vertosols and Aquic Vertosols, so the presence of dark colours as an indicator of wet soils must be used cautiously in the case of Vertosols.

The organic-based features of reduction can form distinct horizons of organic material or black A horizons (Vepraskas 2001). Organic soils, Organosols in the Australian Soil Classification (Isbell 1996), can be recognised by the presence of the thick layers of black to brown to orange organic material. Organosols are defined as *having more than 0.4 m of organic materials within the upper 0.8 m or have organic materials extending from the surface to a minimum depth of 0.1 m that overlie rock or other hard layers, partially weathered or decomposed rock or saprolite or fragmented materials in which the interstices are filled or partially filled with organic material* (Isbell 1996). According to Isbell, organic materials are plant-derived organic accumulations that have either 12% organic carbon in soils with no clay, or 18% organic carbon for soils with a mineral fraction of 60% or more clay. In acid sandy soils, organic matter will be transported by percolating acidic water from the surface to accumulate at the water table as a subsoil horizon rich in dark reddish-brown humate (B*h* horizon) found in Podosols. Isbell (1996) contains a further discussion on Podosol soils and horizons and the presence of significant amounts of organic material in soils other than Organosols.

In the United States, a 0.02 m layer of sapric material is recognised as a reliable indicator of a hydromorphic soil, especially when overlying a gleyed subsoil. A thinner layer of sapric material (0.01 m) on top of a sandy soil indicates that the soil is at least seasonally wet (Tiner 1999). Wet sandy soils can also contain streaks of organic matter in the upper layers of the subsoil, or *organic bodies* growing along the roots that are the result of frequent die back of some roots when subjected to prolonged saturation (Tiner 1999). Further indicators of hydromorphic soils include thin layers of organic matter or predominantly organic-coated soil particles. Stratified layers have also been used in the United States as an indicator of hydric soils whereby at least one layer must consist of layers of organic muck (sapric), modified mineral texture (i.e. an intermediate between organic and mineral soil materials such as mucky sand or mucky sandy loam) or mineral soil of chroma 1 or less and a value of 3 or less and one layer must contain 70% or more soil grains with organic coatings. These soils should not be confused with stratified alluvial soils that are not anaerobic (Tiner 1999).

As previously stated, the concentration of organic carbon is greatest at the surface and decreases with depth. At depths greater than 1 m, the organic carbon is usually located around roots and becomes scarce as depth increases. Features beyond this depth should be interpreted cautiously as it is not always clear when the features formed. Features formed deeply in the soil profile can, in all probability, be preserved for thousands of years (Vepraskas 2001). Organic carbon levels will generally decrease if the soil is drained as there will be a shift in microbial activity with the re-establishment of aerobic oxidation of the organic materials.

Organic matter can be characterised via an assessment of its physical and chemical properties, including bulk density, water holding capacity and organic carbon content. The bulk density tends to increase with increasing decomposition of the organic matter (Collins and Kuerhl 2001). Organic carbon is routinely measured in the laboratory by the Walkley-Black method or by Loss on Ignition (Rayment and Higginson (1992) contains a description of these methods). If drained, carbon-based organic features of saturated soils are likely to decompose too quickly for them to be preserved as relict soil features for more than 30 years (Vepraskas 2001). The system of classifying organic material in Australia is outlined in Appendix 2.

Figure 1: An Organosol from a wetland on Bribie Island (South-East Queensland). Organic material extends down to 0.7 m.

3. Review of wetland soils

The depletion and/or concentration of Iron (Fe) and manganese (Mn)

The depletion and concentration of iron (Fe) and manganese (Mn) in reduced soils can create unique soil morphological characteristics in terms of soil colour, mottles and segregations. These features develop as a result of the processes of reduction, translocation and oxidation of iron and manganese oxides (Vepraskas 1992).

In the United States, the term redoximorphic features has been devised to account for mottles and low chroma colours formed by the reduction and oxidation of Fe and Mn compounds, as the term mottles includes carbonate accumulations and organic stains that do not indicate saturation and reduction (Vepraskas 1992). In Australia, carbonate accumulations are described as segregations (see Segregations, page 26).

When describing redoximorphic features, it is necessary to describe type, colour, inferred composition (Fe, Mn or Fe/Mn), size, abundance, contrast, boundaries and whether the concentrations or depletions occur along macropores or within the soil matrix (Vepraskas 1992).

Hayes and Vepraskas (2000) reported that for soil horizons below 0.75 m, longer durations of saturation are most likely required for iron reduction to occur. Presumably, this is because there is less soluble organic carbon available to promote rapid iron reduction and subsequent formation of redox depletions (see Soil texture, iron and manganese, page 29). This is of most relevance for soils that are near the margins of wetlands where the saturation is seasonal and not permanent.

Soil colour

Soil colour is an important soil property used in the classification of soils. It is of particular importance in the characterisation of wetland soils due to the unique colours that can form under reducing conditions. The Munsell colour system uses hue (related to the spectral wavelength of the colour); value (lightness or darkness) and chroma (the purity or strength of the colour) to describe colour. Hues of red (R), yellow red (YR), and yellow (Y) are typical of both saturated and unsaturated soils, while blue (B), blue green (BG) purple blue (PB) and green (G) are the gleyed colours that are associated with the chemical reduction process — for example, waterlogged soils (Tiner 1999).[4]

Stable iron oxides (ferric form) dominate well-drained soils and cause yellow and red colours. Prolonged periods of saturation convert iron to its mobilereduced, soluble colourless form, resulting in greyish colours in coarse textured soils or bluish-grey, green or gleyed colours in finer textured soils (Mausbach and Parker 2001; Tiner 1999). As the ferrous iron is colourless, the actual colour of the soil will be determined by the colour of the sand, silt and clay particles (Vepraskas 2001). Like ferrous iron, reduced Mn2+ can move with the soil water and be removed from the soil (Mausbach and Parker 2001). The colours of waterlogged soils can be modified by a variety of compounds formed during reduction including black iron sulfides, green hydrated magnetite [$Fe_3O_4{\cdot}nH_2O$ or $Fe_3(OH)_8$], black hydrotroilite ($FeS{\cdot}nH_2O$), white siderite ($FeCO_3$) and blue vivianite ($Fe_3PO_4{\cdot}8H_2O$). If reducing conditions persist, the precipitates may age producing magnetite (Fe_3O_4) and pyrite (FeS_2), among other minerals (Ponnamperuma 1972, 1984). In anaerobic soil profiles saturated with stagnant water (e.g. drainage depressions) that have not been flushed of dissolved and reduced iron oxides, the soils often take on bluish or greenish colours (Diers and Anderson 1984).

Grey or low chroma colours have been reported in the literature as being useful for the identification of wetland soils. Anecdotal evidence observed at Booliga in New South Wales by Paijamans *et al*. (1985) supported an observed relationship where the longer period of inundation corresponded to greyer soils. It is generally expected that soils develop colours with chromas of 2 or less at a faster rate in soils with low pH and higher organic carbon (Vepraskas 1999). Mineral soils that are permanently saturated tend to be uniformly gleyed throughout the profile (Tiner 1999).

The soil at the soil-water interface will sometimes be a brown or reddish-brown colour because of the presence of the ferric iron in the oxidised layer. Within the oxidised layer, chemical transformations are similar to those in aerobic soils (Mitsch and Gosselink 1993). The soil-water interface operates in this manner for as long as the interface is supplied with oxygenated water and the oxygen supply

[4]In the gley colours, the iron is reduced rather than oxidised, giving the soil a black to bluish–grey colour.

exceeds the demand at the interface. Oxygenated water can be supplied by turbulence due to wind or thermal movements, or oxygen produced from algae and plants in the water column.

Mottles

Mottles are spots, blotches or streaks of sub-dominant colours that are different from the soil matrix colour and different from the colour of the ped (McDonald *et al*. 1990). They are described by their type, abundance, size, contrast, colour and distinctness of boundaries, which can be used to infer soil drainage characteristics and are better developed in seasonal rather than permanently wet soils (Kotze *et al*. 1996). They are formed after intermittent periods of wet and dry conditions where relatively insoluble iron remains in the soil long after it has been drained and thus re-exposed to oxidising conditions. Mottles will not form in saturated conditions if the water contains sufficient oxygen to serve the biological needs for organic matter digestion and if the soil or water temperatures are below 5°C (Diers and Anderson 1984).

Figure 2: Soil mottling is usually an indication of poor drainage or water fluctuation throughout a soil profile. The dominant soil colour for this particular soil is grey.

The reduction of the sparingly insoluble Fe(III) to the more soluble Fe(II) species and reoxidation will produce the orange reddish–brown mottles while the oxidation and reduction of manganese will produce dark reddish–brown black mottles. These mottles seen throughout an otherwise grey or gleyed soil matrix is a further indicator of soils that have experienced intermittent reducing conditions (Mitsche and Gosselink 1993; Willet 1983). Grey mottles are generally connected with longer periods of soil saturation than are the brown or red mottles, and appear at a greater depth in the soil profile than the brown or red mottles (Coventry and Williams 1984). Iron mottles can include ferrihydrite, goethite (FeOOH), haematite (Fe_2O_3) and rusty-coloured floccules of ferric hydroxide ($Fe(OH)_3$).

Mottles from the oxidation products of acid sulfate soils include jarosite ($KFe_3(SO_4)_2(OH)_6$) and natrojarosite ($NaFe_3(SO_4)_2(OH)_6$), which form distinct butter- or straw-coloured mottles with a hue of 2.5Y or more yellow and a value and chroma of 6 or more. These mottles are relatively insoluble and are stable at low pH and under dry conditions (Dear *et al*. 2002). Schwetmanite ($Fe_8(OH)_{5.5}(SO_4)$1.25) has recently been identified as a dominant iron mineral of acid sulfate soil landscapes. Schwertmanite mottles can range in colour from reddish–brown to yellowish–brown (Sullivan and Bush 2004).

Mottles tend to accumulate above the depth of the fluctuating water table and the measurement from the soil surface to the upper area of mottling can be an interpretative tool in the determination of a seasonal water table (Diers and Anderson 1984; Guthrie and Hajek 1979). Mobile Fe^{2+} and Mn^{2+} ions tend to diffuse upward from the reduced areas in a concentration gradient, precipitating out as relatively insoluble oxides or oxyhydroxides in the capillary fringe above the water table (Harmsen and van Breeman 1975). Lateral diffusion can also occur in response to a concentration gradient (Wilding and Rehage 1985). A slow rate of re-precipitation of iron is reflected by a coarse pattern, whereas a fine, diffuse pattern reflects a fast re-precipitation process.

High chroma iron concentrations were identified in saturated soils in the United States in the state of Georgia. It was, however, concluded that these features were not reliable indicators of the upper zone of seasonal saturation as measured by piezometers, as they may reflect either capillary action up from the zone of saturation, or high water table levels that occur during extreme rainfall events (Jacobs *et al*. 2002). Isbell *et al*. (1997) reported that hydrosols found in ephemeral fresh water lakes of eastern Victoria and the saline playas of the arid interior of Australia often lack mottles. When mottles are present, there can, however, be no direct assumption made as to whether the mottles are indicative of current hydrological conditions or are relict.

3. Review of wetland soils

Problems with using colour

The use of low chroma colours and mottles in classifying wetland soils is subjective and can at times be misleading. Soil saturation for long periods may fail to develop low chroma soil colours (<2) or mottles due to the supply of oxygenated water (Franzmeier *et al*. 1983, Vepraskas and Wilding, 1983). Short periods of inundation may retard colour change (Vepraskas *et al*. 1999). Low chroma colours may form in some soils where the parent material contains little or no iron or where iron has been stripped from the soil particles by organic compounds, such as chelates, in non-reducing conditions (Vepraskas 1992). Vepraskas and Wilding (1983) reported a situation where soils that were not currently waterlogged contained extensive low chroma colours indicative of relict landscape features.

It was suggested by Megonigal *et al*. (1993) that some soil profiles containing fine-textured materials, especially in horizons deep in the profile, may have reduced rates of oxygen diffusion relative to soil respiration. Consequently, redox features such as mottled and gleyed horizons are maintained in the soil, even though the soil is only periodically saturated.

Soils formed in reddish parent material can be misleading as they sometimes fail to develop redoximorphic colour patterns, even if reduced. The colours of the uncoated mineral grains may themselves be brownish in colour, thus showing a high chroma rather than a low chroma when reduced and the iron coatings have been removed (Rabenhorst and Paikh 2000). Sprecher and Mokma (1989) noted that hematitic soils required more reduction than those rich in goethite to turn grey and Fey (1983) hypothesised that aluminium substitution in the structures of hematite and goethite may lessen their propensity for being reduced and solubilised. A simple colour change index has been devised in the United States that may be useful for these problematic soils (Rabenhorst and Parikh (2000) contain information).

Segregations

Segregations accumulate in the soil because of the concentration of some constituent, usually by chemical or biological action. They are indicative of waterlogging and may be formed *in situ* by current pedogenic processes or can be relict (McDonald *et al*. 1990). Nodules and concretions, two specific forms of segregations, are suspected to form when air penetrates into a soil quickly, perhaps in an area of the wet matrix containing Fe(II) and Mn(II) (Vepraskas 1992).

An investigation into segregations in a Cambodian floodplain identified concentrations of quartz and kaolinite, the major constituents of the surrounding soil matrix, within ferruginous nodules. The nodules were proposed to have formed through the impregnation and cementation of the soil matrix with iron oxide that was imported from the surroundings. The hardening occurred when the concentration and crystallisation of iron oxides had proceeded to the extent that they formed continuous and rigid frameworks within the nodules (Mitsuchi 1976). The ferruginous nodules have been reported to increase in size and abundance at a certain depth in the soil profile, beyond which they decrease and finally disappear.

Nodule formation is generally retarded in the deeper horizons of continuously wet soils. It has been suggested that these conditions retard the dehydration and crystallisation of iron, or results in the complete translocation of the mobile forms from the soil profile (Mitsuchi 1976, Schwertmann and Fanning 1976). The periodic wetting and drying appears to be essential for the soil to develop segregations, with more permanent wetness leading to mottling or even total loss of manganese and iron from the soil profile. For these reasons, it has been proposed that the maximum concentration of segregations is generally positioned above the depth where maximum mottle development occurs (Schwertmann and Fanning 1976).

Water soluble Mn(II) and Fe(II) can diffuse from reduced soils to the oxygenated interface where it is precipitated to form Mn(IV) and Fe(III) oxide hydrates that can oxidise to form nodules of manganese or iron. Manganiferous nodules or mottles represent the earlier stages of development of ferri-manganiferous concretions and possibly ferruginous concretions (Mitsuchi 1976).

Soft segregations of Fe(III) oxides and hydroxides of any shape and ranging in size from 1 mm to greater than 150 mm can occur in the soil matrix away from cracks and root channels. These segregations are easily crushed with the fingers because the iron concentration is not sufficient enough to cement the particles into a solid mass. The colour of the segregation will be determined by the type of iron mineral present (Vepraskas 2001).

Hard nodules or concretions are made up of cemented particles of iron oxides or hydroxides that range in size from 1 mm to greater than 150 mm. If broken open, they generally consist of concentric layers. Hard nodules are not recommended to be used as an indicator of soil saturation as it is never clear whether the features formed *in situ* or were brought into the soil by flooding or from deposition from material eroded from upslope (Vepraskas 2001).

Nodules tend to be absent from coarse textured soils. This has been attributed to a deficiency of iron to cement the soils and rapid desiccation upon drainage, which means that the segregations do not have time to form. This tends to lead to more uniform colouration of the soil (Mitsuchi 1976).

Oxidised rhizosphere and pore linings

The concept of an oxidised rhizosphere is a further morphological characteristic of some waterlogged soils with a source of soluble iron (Mendelsohn *et al*. 1995). Some species of hydrophytic vegetation have the capacity to transport oxygen through above-ground stem and leaves to below ground roots.

Figure 3: An example of a wetland soil with oxidised rhizospheres.

build up of oxidised iron coatings on roots in fine textured soils compared to coarse textured soils. It was hypothesised that more iron was bound in the finer textured soils making it less available, providing a more reduced soil environment. This may counterbalance the oxidative capacity of the roots, resulting in less accumulation of iron coatings.

When oxidised rhizospheres are found in association with a living root, it can be assumed that the anaerobic conditions have occurred within the lifespan of the plant root. This is most likely to be within that growing season and is a reasonably reliable indicator of current, or at least recent, hydrological conditions (Mendelssohn *et al*. 1995).

Oxidised pore linings can be found in reduced soils and can be evidence of a reduced matrix where oxygen enters the pores and oxidises the ferrous iron around the pores. Pore linings occur as oxidisation along open pores in the soil matrix and produce an orange halo that extends into the soil matrix. The pore may have formed after the bacterial decomposition of a dead root. While the channel contains air, the only reduction that occurs is that of O_2 to water. If the channel fills with water and the supply of oxygen from the atmosphere is excluded, bacterial respiration will still occur and the organic tissue will continue to be oxidised until the system becomes anaerobic. With the commencement of the reduction of iron, dissolved Fe(II) moves off the particle surfaces and can diffuse through the soil matrix or be moved to other parts of the soil by water. The colour of the soil around the channel will gradually change to grey with the loss of iron. When oxygen enters the system the newly formed iron depletion will retain its grey colour because it is the colour of the uncoated minerals. The oxygen that penetrates the soil matrix may oxidise the Fe(II), thus causing a pore lining or iron mass to form (Tiner 1999; Vepraskas 2001).

Diffuse versus sharp boundaries

A diffuse boundary around accumulations of iron in the soil is assumed to indicate that the feature is forming or has formed in the recent past and is a reflection of current hydrological conditions. When iron accumulations begin to dissolve (e.g. as a result of oxidation) or are mixed into the soil, they tend to attain a sharp boundary with the soil matrix and are relict features of the hydrological regime. A situation where all the boundaries were sharp would most

3. Review of wetland soils

likely indicate a year-round oxidising environment (Vepraskas 2001). However, research presented by D'Armour *et al*. (2004) reported an example from Oregon in the United States, where redox concretions of a spherical nature were formed in place in a saturated environment, that is not relict, but were no longer actively growing by accretion of iron or manganese because the supply from the matrix had been exhausted. These concretions consequently possessed sharp boundaries.

Soil texture, iron and manganese

Coarse textured soils tend to exhibit no consistent pattern associated with the depletion or concentration of iron and manganese as water moves through both the soil matrix and macropores (Tiner 1999). They do not have large cracks or root channels that remain open for extended periods, allowing large features to form, as the channel will collapse once the root dies and is decomposed. Many coarse textured sediments are low in iron oxides and consequently the contrast between the matrix and the depletions is not as obvious as those in the finer textured sediments (Vepraskas 2001). In addition, manganese oxide remains dissolved longer than iron oxide and consequently is carried farther by soil water and is usually leached from the soil profile if the period of saturation is significant (Collins and Buol 1970, Diers and Anderson 1984).

Mottling in surface water-dominated, medium-textured wetland soils can be different whereby the interior of the soil peds retains a brown colour, often with rusty accumulations of iron. The material lining cracks and wide pores is to be reduced. Iron then diffuses into the peds, with oxygen still present in the peds causing the formation of reddish or brownish rusty mottling (Mooremann and Van de Wetering 1985).

Bluish-grey coarse textured soils and gravels containing dissolved and reduced iron oxides tend to brighten fairly rapidly after drainage. This is because the oxygen will freely penetrate the porous materials, thus oxidising the iron oxide (Diers and Anderson 1984).

Clay depletions appear as grey-coloured coatings that can line channels or may form on the outer surfaces of the soil peds and are formed by a loss of both iron and clay. Removal of manganese and iron oxide may cause the clay along a former root channel to disperse when the soil is wet, and move downward in the soil profile (Vepraskas 1992). Thus, the adjacent soil matrix and the underlying soil layers will have higher clay content. The depletion areas generally form along ped surfaces or large root channels and are not commonly found in the upper 0.3 m of the soil profile (Tiner 1999; Vepraskas 1992, 2003).

Microrelief

Debil-debil and Swamp Hummock are variations of microrelief that may also be indicators of hydromorphic soils. Debil-debil are small hummocks rising above the planar surface, which can be rounded or flat-topped and usually regularly spaced ranging from 0.06 m to 0.6 m in both vertical and horizontal dimensions (McDonald *et al*. 1990). They are usually found on soils with impeded internal drainage and in areas of short seasonal ponding.

Swamp Hummocks are steep-sided hummocks rising above a flat surface. The upper surface is most commonly vegetated whilst the lower surface may be free of vegetation. Swamp Hummocks are usually present in areas subject to prolonged seasonal flooding.

The reduction of sulfur and carbon

The detection of hydrogen sulfide (H_2S) odour is a further indicator of hydromorphic soils, particularly coastal acid sulfate soils where the seawater provides the source of sulfate for reduction. The reduction of sulfate is caused by a group of obligate, anaerobic bacteria of the genus *Desulfovibrio* that use sulfate as an electron acceptor in respiration (Armstrong 1982). The reduction of sulfate usually requires a relatively long period of inundation and anaerobic respiration (Vepraskas 2001). The detection of H_2S indicates sulfate-sulfur has been reduced and the soil is currently reduced where these features are detected (Vepraskas 2001).

While large amounts are produced in water logged soils, the concentration of water soluble H_2S may be small as it is often removed as insoluble sulfides such as FeS, and thus the solutions of submerged soils rarely contain more than 0.1 ppm H_2S (Ponnamperuma 1972). The detection of H_2S odour is a qualitative measure and if detected within the top 0.3 m of the soil surface is usually a reliable indicator of hydromorphic soils (Tiner 1999) and acid sulfate soils. Upon drainage and the re-establishment of aerobic conditions the production of H_2S will cease (Vepraskas 2001). As H_2S is toxic, there are workplace health and safety issues to consider. In addition to the detection of H_2S, the presence of iron sulfides are also reliable indicators of soil saturation as the iron sulfides are stable under reducing conditions. Both field screening tests using hydrogen peroxide and laboratory procedures are available to detect the presence of iron sulfides. Ahern *et al*. (2004) contains further information on this topic.

The reduction of carbon dioxide produces methane gas (CH_4). Methane can be monitored in the field if collected in water-filled plastic bags that are inverted and placed on the surface of a waterlogged soil for a 24-hour period. A bubble of gas that collects in the bag is assumed to be methane if it is allowed to escape through a pinhole placed in the bag and ignites in the presence of a flame (Vepraskas and Faulkner 2001).

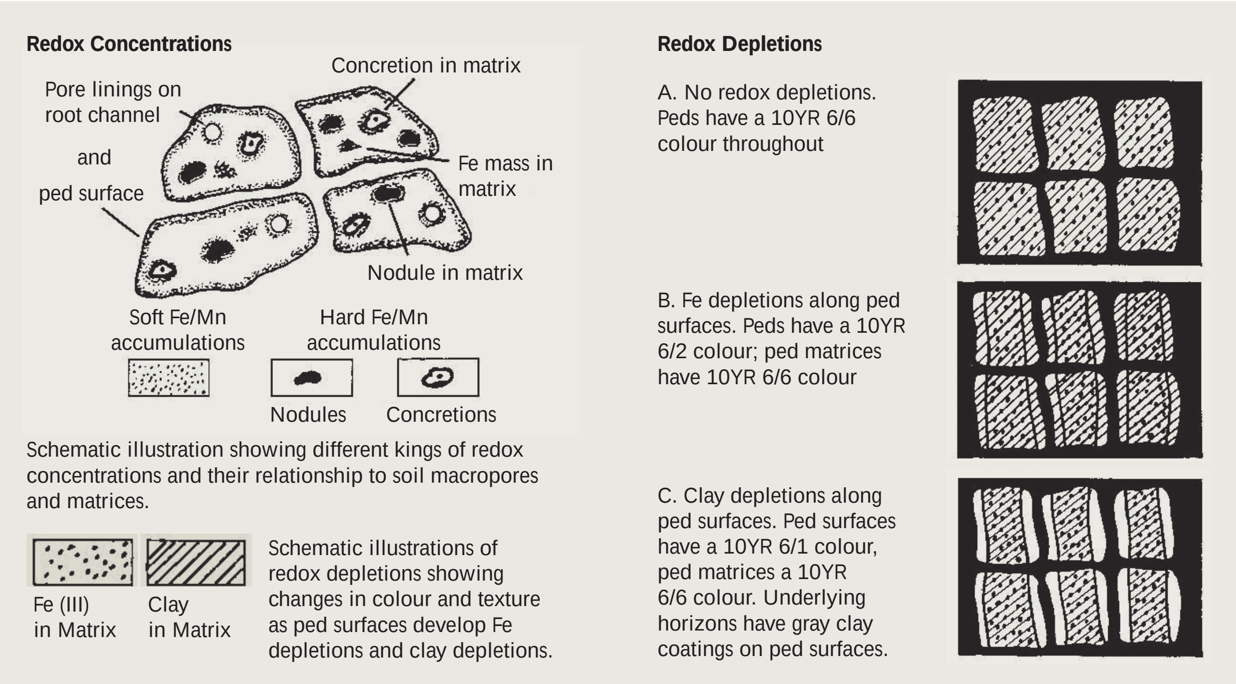
United States terminology and concepts

A variety of terms have been devised in the United States to describe wetland soil morphology. For example, the concept of redox depletions have been devised to reflect bodies of low chroma (two or less) having a value of four or more where iron and/or manganese oxides have been stripped or when both the oxides and the clay have been stripped from the matrix (Hurt *et al*. 2003). The term redox concentrations is commonly used to describe bodies of apparent accumulation of iron and manganese oxides (Figure 4, Vepraskas 1992). These concepts and many others have been devised on the basis of more than 10 years' research on wetland soil morphology and are applicable to the climatic, soil, landform and vegetation conditions of the United States. It is unclear whether these concepts could be modified to reflect Australian conditions. Hurt *et al*. (2003) contain further details of the terminology used in the United States.

Time periods required for a soil to become hydromorphic

It is generally not known how long it would take for a soil to become hydromorphic, display hydromorphic indicators or the period of saturation required in order to support the growth of hydrophytes (Mausbach and Parker 2001). However, in a deep marsh created near Chicago, United States in 1989, wetland soil indicators indicating reduced conditions and the accumulation of organic matter occurred over a five-year study period. Coventry and Williams (1984) reported that in semi-arid

Figure 4: Redox concentrations, from Vepraskas (1992).



Australia, soil hydromorphic properties may develop in soils that are saturated for less than five weeks at a frequency of once every three years. Some morphological indicators will persist in the soil once they have been formed, with some features developing during saturation events that occur rarely (e.g. average of one in every 10 years) (Vepraskas and Caldwell 2006).

Morphological evidence of wetness such as low chroma colours, mottles and iron segregations provide evidence of saturated reducing and oxidising conditions. However, many of these features will change if the soil becomes aerobic, due to biotic homogenisation and further weathering. The changes are slow and fine textured soils in temperate zones retain low chroma colours for several centuries. The morphological characteristics of hydromorphic indicators in coarse textured soils may disappear a few decades after drainage (Mormann and Van de Wetering 1985). Consequently, complications can arise when it is unclear if morphological indicators reflect former climates, current saturated conditions and duration or current landscape conditions (Vepraskas 1992).

Monitoring as an indicator of hydromorphic soils

An option to determine indicators of hydromorphic soils entails *in situ* monitoring and laboratory analysis of various soil components. There are a range of alternatives available to measure the characteristics of a soil that are indicative of waterlogged conditions. Faulkner *et al*. (1989) reported that soil identifiers of interest to characterise wetlands are oxidation-reduction (redox), soil oxygen content and water table depth. Simple, reliable and reproducible methods that are inexpensive are considered to be the most desirable. While many of the methods described below are accurate and quantitative, they are expensive and time-consuming and consequently measurements can only be taken on a few benchmark soils for any large area (Vepraskas and Wilding 1983).

Redox potential

The redox potential (Eh) is a quantitative measure of the electrochemical potential or electron availability in chemical or biological systems. Electrons are essential to all chemical reactions. Reduction occurs as a chemical species gains electrons, while oxidation occurs as a chemical species loses electrons. The nature and proportions of the oxidising and reducing substances contained in the soil will affect the redox potential. Redox potential will indicate the intensity of oxidation or reduction of a chemical or biological system — that is, the tendency of the soil to oxidise or reduce substances (Gambrell and Patrick 1978; Faulkner *et al*. 1989). Further information regarding redox potential is detailed in Appendix 2.

The principal reducing reactions that form the morphological features of a reduced soil are outlined in Table 2. Some soils are more likely to have one group of the above features than others. An example reported by Vepraskas (2001) related to a sandy soil that was low in Fe, due to low Fe in the parent material. The morphological features of reduction found in this soil would be based on organic carbon, manganese and occasionally sulfur-based features.

The dominant redox system in soils is usually that of the iron hydroxides, rather than the manganese. Iron is generally the most abundant chemical element in the soil (Tiner 1999) and the influence of manganese is weak because manganese dioxide is insoluble in water and is used as an electron acceptor in respiration by a limited number of bacteria (Ponnamperuma 1972). Once manganese is reduced it is more soluble and can be readily leached from the system. The presence of manganese dioxide or manganic compounds may delay iron reduction to the ferrous state as they retard an Eh decrease in flooded soils and prevent the build up of high concentrations of Fe(II) (Armstrong 1982, Ponnamperuma and Castro 1964).

The most obvious change associated with iron reduction occurs when the brown iron-rich soils change to shades of grey, green or blue when large amounts of iron come into solution. The reduced iron acts as a sink for oxygen that diffuses into the soil and is a source of Fe(II) ions (Ponnamperuma 1984). The rate and magnitude of the decrease in Eh after submergence will depend on the amount and type of organic matter present, the nature and content of electron acceptors, temperature and the duration of the submergence event (Ponnamperuma 1972).

While Eh is a useful indicator of aerated or waterlogged soils, it does not account for seasonal variation in soil saturation and will confirm anaerobic

Table 2: Four of the major reducing reactions related to morphological features of a hydromorphic soil (after Vepraskas 2001 and Mitsche and Gosselink 1993).

Reducing reaction	Approx. Eh (ph7) mV	Morphological feature formed Group name example	
O_2 + 4e¬- + $4H^+ \rightarrow 2H_2O$	600	Organic C features	Organic and some black A horizons
$MnO_2 + 2e^- + 4H^+ \leftrightarrow Mn^{2+} + 2H_2O$	225	Manganese-based features	Mn masses, black and grey mottles
$2FeOOH + 4e^- + 6H^+ \leftrightarrow 2Fe_2+ + 4H_2O$	120	Iron-based features	Fe masses, red, yellow and grey mottles
$SO_4^{2}- + 8e^- + 10H^+ \rightarrow H_2S + 4\ H_2O$	–75 to –150	Sulfur-based features	Odour of rotten egg gas

conditions only when the soil micro-organisms are active. Redox measurements can be affected by temperature and are dependent on pH, which complicates its use in comparing soils. Corrections for pH can introduce error as the correction value is variable according to the nature of the redox reaction involved. Consequently, both pH and Eh should be specified separately (Armstrong 1982).

The sequential reduction of elements can be influenced by changes in pH and the activity of the reactants. Errors associated with redox measurements include electrode malfunction, pH effects, absence of true equilibrium, liquid junction potential errors, heterogeneity of the medium and discrepancies between the potentials measured in muds and those of their equilibrium solutions. There are also potential errors associated with sampling muds and extracting interstitial solutions without altering their composition (Ponnamperuma 1972). Reproducible results are often difficult to achieve in aerated systems, possibly because these systems are not in equilibrium with themselves (Ponnamperuma 1972).

Ferrous iron detection

The use of the alpha-alpha-Dipyridyl method for measuring ferrous iron (Fe(II)) content of a soil can be used as an indicator of soil reduction. This technique can be a useful diagnostic indicator for wetland soils, as the reaction will not occur in the presence of oxygen as the iron-reducing bacteria are obligate anaerobes. The test yields a single point in time determination that requires repeated measurements to determine frequency and duration of reducing conditions (Tiner 1999). For example, the USDA (2006) state a soil is hydric when a positive reaction to alpha-alpha-Dipyridyl is achieved for at least 14 consecutive days.

The colourless dye solution is applied to the freshly broken surface of wet soil samples. Fe(II) is indicated when the dyed soil changes to a reddish colour (Vepraskas 1992). The best time to verify that iron reduction is taking place is when the soil is saturated (Vepraskas 1992). The presence of Fe(II) in the soil indicates that the soil was reduced at the time of sampling and does not suggest reducing conditions at other times of the year.

While alpha-alpha-Dipyridyl may be useful for iron-rich soils that are anaerobic at all times, it may not be particularly useful for soils with low iron contents or in wetland margins which are not necessarily reduced at all times. Tiner (1999) concluded that the alpha-alpha-Dipyridyl test is not highly recommended.

There can be errors associated with this technique due to the oxidation of Fe(II) when exposed to bright sunlight and to air (Faulkner *et al*. 1989). For validity of the procedure, the soil must contain iron and an amount of organic matter suitable for microbial reduction, and there can be problems associated with this technique in soils with low iron contents that have been reduced (Tiner 1999). Soils that have been in contact with steel augers, probes, knives or hydrochloric acid (to test for carbonates) can also yield false positives (Vepraskas 1996). The dye is also sensitive to sunlight and is toxic if ingested (Vepraskas 1992).

Ferrous iron can also be detected by some further means, including the analysis for total dissolved iron in groundwater by atomic absorption spectrometry. In a Texan toposequence where iron was measured via this method, it was assumed that all soluble iron was Fe(II) (Vepraskas and Wilding 1983). Samples must be collected under vacuum and stored in capped bottles. Again, there is a reliance on correctly installed piezometers.

There has recently been interest in the use of synthetic iron oxides as an indicator of reduction in soils, whereby PVC pipes coated with a paint prepared from an iron oxide suspension are inserted into the wetland soil. When removed, the pipes are analysed to assess the loss of ferrhydrite paint from the surface, which provides information on the amount of reduction occurring in the soil profile (Castenson and Rabenhorst 2006, Jenkinson and Franzmeier 2006). This technique has shown promising results and is being further researched in the United States (Rabenhorst and Burch 2006).

Soil pH

In general, for most acidic and alkaline soils, as the soils become saturated after submergence the pH tends towards neutral values. The flooding of alkaline soils will generally cause the pH of the soils to drop, probably as a result of CO_2 build up and carbonic acid formation. The pH of acidic soils, once flooded, tends to increase due to the reduction of ferric iron (Mitsch and Gosselink 1993), assuming that iron is present. The pH of acid organic soils high in iron does not necessarily increase after submergence and fibric soil material will often remain acidic due to the slow oxidation of sulfur compounds, producing sulfuric acid (Mitsch and Gosselink 1993). The draining of acid sulfate soils will lower the pH following the oxidation of iron sulfides. Organic wetland soils will often have an acidic pH, particularly in fibric materials if there is limited inflow of groundwater (Mitsch and Gosselink 1993).

The pH value of waterlogged soils influences the equilibria of hydroxide, carbonate, sulfide, phosphate and silicate. The equilibria of these elements regulate the precipitation and dissolution of solids, the sorption and desorption of ions and the concentrations of ions such as Al^{3+}, Fe^{2+}, H_2S, H_2CO_3 and some organic acids. Because excess water-soluble aluminium and iron are toxic, the influence of pH on the solubility of these elements is of concern (Ponnamperuma 1972). For example, the species of aluminium that is most toxic to fish is still soluble at pH 5.2 (Sammut 1996).

In general, there are too many variables involved for pH to be a particularly useful indicator of wetland status.

Oxygen content

A technique to assess soil oxygen status of anaerobic wetland soils is based upon polarography, using a platinum microelectrode. This method was originally described by Lemon and Erickson (1952, 1955). Electro-oxidisable or electro-reducible substances provide unique current-voltage curves, permitting their identification and assay (Armstrong 1982). This method can be complicated; for example, plateau shifts accompany the lowering of oxygen status or increase in soil acidity and the technique is expensive.

Oxygen diffusion chambers were set up to measure soil oxygen content in an experiment presented by Parker *et al*. (1985), with soil oxygen content measured with an oxygen meter, whereby soil oxygen generally decreased with increasing wetness and increasing depth.

A further method of measuring oxygen content is the dissolved oxygen content of the groundwater using a dissolved oxygen meter. As discussed previously, there are similar problems associated with the use of correctly installed piezometers and well construction.

Acid Al-clay

With the reduction of soil, the Fe(II) will displace the other cations from the exchange complex, thus increasing the soil solution concentration of Ca^{2+}, Mg^{2+}, K^{+} and Na^{+}. These cations may be slowly leached from the soil, so that on reoxidation an acid aluminium- dominated clay is produced (Willet 1983). A measure of the cation exchange capacity of a soil, including the exchange acidity, will provide information on the clay mineralogy. However, the presence of an acid Al-clay is not entirely suggestive of a wetland soil as other highly weathered soils can also retain these characteristics.

If the hydrology of a wetland is altered — that is, the wetland is drained — the soil morphology tends to reflect the previous water regime for a longer period than the hydrophytic vegetation. Thus, an evaluation of the soil properties is important for the accurate delineation of wetlands (Tiner 1999).

If present, the oxygenated soil-water interface can also have a role regulating nutrient cycles and toxins in lakes, whereby phosphate, Fe(II), Mn(II), silica and other soluble substances escape from the soil into the next layer during warm weather. Upon the return of cold weather, the layers in the lake mix, Fe(II) and Mn(II) are oxidised and sink as precipitates to the bottom, carrying with them phosphate, silica and sulfate (from Hutchinson 1957, documented in Ponnamperuma 1972). The reduced subsurface soils are often bluish-grey in colour, are anaerobic, contain a low oxidation-reduction potential and contain reduced products such as ammonia, nitrous oxide, ferrous and manganous salts, sulfides and the products of anaerobic decomposition of organic matter including CO_2 and CH_4 (Ponnamperuma 1972). Some of the end products will escape, and will be leached, precipitated or transformed into other insoluble compounds.

3. Review of wetland soils

Soil morphological generalisations for hydromorphic soils

Based on the information available in the literature and discussed previously in this section, there are a variety of indicators of wetland soils that are generally applicable and can be morphologically identified and/or directly measured. These indicators may prove useful as a tool for identifying the difference — and thus the boundary — between wetland and non-wetlands soils. The indicators potentially useful for Queensland are summarised in Table 3.

Table 3: Summary of hydromorphic soil indicators for wetlands of relevance to Queensland conditions.

Indicator	Description
Organic material	Wet soil conditions favour the accumulation of thick organic horizons, that are often only partially decomposed soil consists predominantly of decomposed plant material (e.g. fibric, hemic or sapric peat); has a thick layer of decomposing plant material on the surface, or has a dark surface. Organic material can be measured in the laboratory.
Streaked organic matter	Soil is sandy and has dark stains or dark streaks of organic material (decomposed plant material attached to the soil particles) in the upper layers. When the soil from these streaks is rubbed between the fingers a dark stain is left on the fingers.
Gleyed matrix	A gleyed matrix (bluish-grey or grey colour below the surface) that occupies 60% or more of a layer starting within 30 cm of the soil surface.
Redox depletions	Bodies of low chroma (2 or less) having a value of 4 or more where Fe and/or Mn have been stripped or when both the oxides and clay have been stripped from the matrix. The redox depletions may be referred to as iron depletions or manganese depletions.
Decreasing matrix chroma, mottle hue and chroma	Matrix chroma steadily decreases when moving from dry to wet soil conditions. Mottle hue and chroma initially increase, but decrease as the wet extreme of soil saturation is approached.
Mottles	Abundance, size and colour of the mottles in a depleted matrix usually reflects the duration of saturation and can be used to infer soil saturation status. Overall mottle abundance initially increases then steadily decreases as the soil becomes increasingly wet. Soils that are predominantly grey with brown or red mottles are often waterlogged for a longer period than those that are yellow or brown with grey mottles. Soils with only a few grey mottles near the surface are not usually wet enough for the soil to be identified as hydromorphic. The higher the percentage of brighter mottles the more dry oxidising conditions predominate. In some soils, the visibility of mottles can be masked by the presence of organic matter.

Indicator	Description
Segregations	Predominantly black nodules are replaced by red nodules as the soil becomes increasingly wet.
Mottle and segregation boundaries	Contemporary and recent hydric soil morphologies have diffuse boundaries, with relict hydric features having sharp boundaries.
Soil-water interface	A thin layer of oxidised soil is present at the surface of the soil-water interface. Evidence is via the red-orange colours of the oxidised soil.
Oxidised rhizosphere	Thin iron deposits present in an otherwise grey matrix along small roots.
Pore linings	Orange halo that extends into the soil matrix.
Groundwater table depth	Present ≥0.30m from the surface.
Redox potential	Redox potential ≤175mv at pH 7.
Ferrous iron detection	Positive detection using alpha-alpha-Dipyridyl method.
pH	pH is not an indicator that can be used to verify the presence/absence of hydromorphic soils. However, it can provide useful interpretive information.
Cation exchange capacity	Cation exchange capacity is not an indicator that can be used to verify the presence/absence of hydromorphic soils. However, it can provide useful interpretive information.
Particle size analysis	Particle size analysis can provide useful interpretive information.
Soil oxygen	≤5ppm
Hydrogen sulfide	Rotten egg gas odour detected within the upper 30 cm.
Methane gas	Methane detected in the field.
Pyrite present	Hydrogen peroxide field test can be of use to detect the presence of hydrogen sulfides present, which are stable under reducing conditions.
Salt profile	The salt profile is not an indicator that can be used to verify the presence/absence of hydromorphic soils. However, it can provide useful interpretive information.

3. Review of wetland soils

Soils with inconclusive or misleading indicators

For a variety of reasons, some soils may show no morphological evidence of submergence, or may appear to be inconsistent with the landscape, vegetation and hydrology. The following list is summarised from the previous discussion in Soil morphology as an indicator of hydromorphic soils (see page 19):

- soils that are low in iron will fail to develop a depleted matrix or mottles
- soils with no organic matter may not undergo reduction, even though the soils are anoxic, as reduction reactions are facilitated by microbial activity that require an adequate supply of organic carbon and a temperature above 5°C
- soils with an impermeable subsoil layer that few large root channels penetrate may not be reducing at depth, even though the surface is ponded, as water is unable to penetrate the impermeable layer
- reducing conditions will be prevented in a soil that is submerged if there are lateral inflows of oxygenated water
- in a grey sand, no iron-based morphological features will develop, but given sufficient time the accumulation of organic carbon and the reduction of sulfur may develop (Vepraskas 2001)
- the reduction of iron may be inhibited in soils with a high pH, as under such conditions iron reduction will only occur at very low Eh values
- iron reduction may be inhibited in soils with high levels of manganese oxides or compounds, that delay or prevent iron reduction to the ferrous state
- soils developed in semi-arid conditions with low precipitation sometimes fail to display clear morphological indicators due to the irregular frequency of inundation
- some coarse textured soils will not develop mottles due to the low levels of iron present
- the retardation of reduction can also occur in areas of high salinity where salinity interferes with the growth of the reducing micro-organisms
- many wetland soils are not reduced at all times, and consequently procedures that indicate a soil is not in a reduced state may not be an accurate reflection of the situation at other times of the year
- it may not be clear whether some of the morphological indicators such as mottles reflect current conditions or relict soil saturation conditions
- it is not always clear whether some morphological indicators have formed *in situ* or were brought into the soil by flooding or by deposition of material eroded from upslope
- soils formed in reddish parent material can be misleading as the soils sometimes fail to develop redoximorphic colour patterns, even if reduced
- horizons in the subsoil that develop a grey colour through soil-forming processes that may or may not include iron reduction
- some parent materials can contain insignificant amounts of iron or manganese.

4. Review of statutory and non-statutory measures for the protection of wetlands

International conventions and agreements

Wetlands are the only sole group of ecosystems to have their own international convention, the Convention on Wetlands of International Importance (Ramsar, Iran 1971) which came into force in 1975. This is the inter-governmental conservation treaty for international co-operation for the conservation and wise use of wetlands and their resources (Ramsar 2006). There are currently 1608 wetland sites listed in the Ramsar Convention[5], five of which are in Queensland. The main focus of the Ramsar Convention is the conservation of the habitats of migratory birds.

The Bonn Convention, to which Australia is a signatory, allows for the protection of migratory wild animal species (Turner *et al*. 2000). Australia has also entered into the Camba[6] and Jamba[7] agreements.

The protection of wetlands in Queensland

There is a variety of legislative, policy and planning mechanisms available to protect wetlands in Queensland. Legislative mechanisms for protecting wetlands in Queensland occur predominantly within the *Environmental Protection Act 1994, Coastal Management and Protection Act 1995, Integrated Planning Act 1997, Water Act 2000, Nature Conservation Act 1992, the Vegetation Management Act 1999, Fisheries Act 1994 and the Transport Infrastructure Act 1994*. At the Commonwealth level, the *Environmental Protection and Biodiversity Conservation Act 1999* also applies to the protection of wetlands when activities are undertaken that may significantly impact on matters of national environmental significance, including World Heritage listed areas.

The *Coastal Management and Protection Act 1995* includes specific policies on avoiding the degradation of wetlands. The Act was recently tested in the Queensland Planning and Environment court, where a developer argued their proposed development area was not a 'wetland' and that the development would not compromise the policies and principles of the Act. It was evident from the transcript of the case that the reliance on soil morphology in the definition of a wetland needs to be further defined.

There are additional policies and plans that add to the protection of wetlands in Queensland by providing further guidance to the day-to-day administration of legislation and/or government agendas or directions. Policies and plans can be statutory or non-statutory — e.g. The *State Planning Policy 2/02 Planning and Managing Development Involving Acid Sulfate Soils; the Reef Plan Water Quality Protection Plan 2003; the Strategy for the Conservation and Management of Queensland Wetlands 1999; and the Wetlands Policy of the Commonwealth Government of Australia (1997).*

Many of Queensland's wetlands are protected by tenure as they are within national parks and gazetted environmental reserves. However, wetlands that fall outside of such tenure regimes are under the most pressure where local economic development demands can override wetland conservation.

To a degree, there is some overlap with respect to various legislation, policy and plans that affect wetlands in Queensland. All legislation, policies and plans are, however, attempting to achieve ecological sustainable development (ESD) of wetlands while avoiding the loss or further degradation of wetlands. It should be noted that to be effectively regulated a wetland must first be identified and delineated; thus, the clear identification of the hydrophytic vegetation, hydrology and hydromorphic soil is a precursor to any effective regulation regime to protect Queensland's wetlands. For more information on the various legislation, policies and plans that impact on Queensland's wetlands, refer to the Environmental Protection Agency review, which is to be completed during 2006.

[5]See www.ramsar.org/sitelist_order.pdf for an updated list.

[6]Agreement between the Government of Australia and the Government of the People's Republic of China for the Protection of Migratory Birds and their Environment, Canberra, October 1986.

[7]Agreement between the Government of Australia and the Government of Japan for the Protection of Migratory Birds in Danger of Extinction and their Environment, Tokyo, February 1974.

4. Review of statutory and non-statutory measures for the protection of wetlands

The protection of wetlands in the United States

In the United States, there are two pieces of legislation of relevance to the regulation and protection of wetlands: Section 404 of the *Clean Water Act 1972* and the *Food Security Act 1985*. The current joint definition of hydric soils that is applicable to both pieces of legislation is *soils that are formed under conditions of saturation, flooding or ponding long enough during the growing season to develop anaerobic conditions in the upper part* (National Technical Committee for Hydric Soils 1994).

The Clean Water Act

The *Clean Water Act 1972* was enacted in 1972 to address rapidly declining water quality. The objective was to maintain and restore the chemical, physical and biological integrity of the waters of the United States. Section 404 of the Act authorises the issuing of permits for fill material or discharge into waters including wetlands. The Chief of Engineers defined the term 'wetland', devised a Wetland Delineation Manual for identifying and delineating wetlands (Mausbach and Parker 2001) and makes a final determination of whether an area is a wetland and if the activity will require a permit.

The legal definition of a wetland as used by the US Army Corps and referred to in the Federal *Clean Water Act 1972* is:

The term 'wetland' means those areas that are inundated or saturated by surface or groundwater at a frequency and duration sufficient to support, and that under normal circumstances to support, a prevalence of vegetation typically adapted for life in saturated soil conditions. Wetlands generally include swamps, marshes, bogs and similar areas.

Thus, a jurisdictional wetland must contain three characteristics:

1. wetland hydrology
2. hydrophytic vegetation
3. hydric soils.

The term 'under normal circumstances' was included in the definition to account for instances in which the vegetation or soils have been removed or altered as a result of recent natural events or human activities (Mausbach and Parker 2001). Unless an area has been altered or is a rare natural situation, wetland indicators of all three characteristics must be present during some portion of the growing season for an area to be deemed a wetland.

When considering applications for permits, the administering authority is required to consider all factors in the public interest, including economic development and environmental protection (US Army Corps of Engineers, 2006). All applications go through an assessment of public interest whereby both the project-specific and cumulative impacts of the proposed action are assessed against wetland functions. Applications are also subject to a functional assessment whereby the level of wetland performance of hydrological, biological and habitat maintenance processes is assessed. This helps to make an assessment of whether the project will result in mitigation (gains) or impacts (losses) of functions (Montgomery *et al*. 2001). The SWANCC decision of the United States Supreme Court (January 2001) reduced the scope of the *Clean Waters Act 1972* by limiting the definition of 'waters' of the United States. The burden for protecting waters no longer subject to the Act has been shifted to state and local government authorities. Half of the states of the United States contain no state programmes to address the reduction in federal government jurisdiction (Christie and Hausmann 2003).

4

The Food Security Act

The *Food Security Act 1985*, commonly called the Swamp Buster Act, contains provisions designed to prevent or discourage the conversion of wetlands into non-wetlands on agricultural lands. The Act also contains provisions that promote the protection of wildlife habitat and water quality. The swamp buster provisions deny federal farm program benefits to primary producers who converted wetlands after 23 December 1985. The benefits include commodity price support or production adjustment payments, farm storage facility loans, disaster payments, payments for storage of grain owned or controlled by the Commodity Credit Corporation, federal crop insurance and loans. The Act also allows for inadvertent wetland converters to regain lost federal benefits if they restore converted wetlands. It also allows for the protection and restoration through permanent and temporary easements.

A converted wetland is one that has been drained, dredged, filled, levelled or otherwise manipulated to make production of an agricultural commodity possible, if production would not have been possible but for this action, and before this action. A wetland under this Act is defined as: land that (1) has a predominance of hydric soils; (2) is inundated or saturated by surface or groundwater at a frequency and duration sufficient to support a prevalence of hydrophytic vegetation typically adapted for life in saturated soil conditions; (3) under normal circumstances does support a prevalence of this vegetation.

The Act requires that wetlands be mapped and delineated and if requested, an effort must be made to conduct an on-site wetland determination prior to delineation. To be consistent with the *Clean Water Act 1972*, the definition of a wetland includes an area comprising the criteria of hydrophytic vegetation, hydric soils and hydrology.

5. Review of classification and delineation of wetland soils

Australia

Wetland soils can be classified as belonging to the orders of Hydrosols, Organosols, Podosols, Anthroposols and Vertosols as defined by the Australian Soil Classification (ASC, Isbell 1996). The soil orders of the ASC are grouped according to their most defining feature. For example, Hydrosols are seasonally or permanently wet soils with saturation of the greater part of the profile for prolonged periods (2–3 months) in most years. The ASC has no definition of the 'greater part' of the profile and consequently a soil that is saturated in lower horizons may constitute a Hydrosol. A Hydrosol would, however, be unable to support hydrophytic vegetation if the upper part of the profile is drained or in an oxidised state and consequently would not theoretically be considered a wetland soil. Both the Podosol and Vertosol are orders of the ASC and contain suborders that relate to aquic soil conditions. Podosols are soils with horizons with a dominance of iron or organic-aluminium compounds, that is Bs, Bh or Bhs horizons. Vertosols are clay soils with shrink-swell properties that exhibit strong cracking when dry and at depth have slickensides and/or lenticular structure. Organosols do not contain an aquic suborder; instead, the description states that they are dominated by organic materials of specified thickness and organic carbon content (see Appendix 2). Their nature and occurrence will be dictated by wetness and the resulting plant communities that grow in a given climatic environment. Many Organosols would also qualify as Hydrosols. Anthroposols differ from other soils of known origin, and there is a question as to whether soils with a human induced rising water table should be regarded as Anthroposols and therefore a wetland soil (Isbell *et al*, 1997).

To date no specific classification or delineation system that has been devised specifically for hydromorphic soils in Australia.

United States

The classification of hydric soils in the United States

In the United States soil classification system, Soil Taxonomy (Soil Survey Staff 1975), an aquic subclass is defined for most soil orders. An aquic moisture regime occurs in soils that are periodically saturated with groundwater for sufficient periods to cause the soils to be anaerobic, although the period of inundation is not specified. Soils that are periodically saturated throughout the profile belong to aquic suborders and those saturated only in the lower horizons are included in aquic subgroups (Wilding and Rehage 1985). Originally, the aquic moisture regime was defined on the basis of saturation and reduction, or removal of dissolved oxygen, but measurements of these soil properties were not required for the classification. Instead, morphological indicators of low chroma colours were used to establish if the soil was saturated and reduced at some stage during the year. However, using colours requires that iron, not oxygen be reduced (Vepraskas 1992, Vepraskas and Wilding 1983). In addition, there were issues with the characterisation of saturated soils for surface water versus groundwater dominated wetland soils, where the upper horizons are saturated but lower horizons are not (Mooreman and van de Wetering 1985). Consequently, soil taxonomy was not considered to be a satisfactory system for classifying or delineating wetland soils for regulatory purposes. Instead, specific tools for delineating hydric soils were developed.

USDA 2006 states hydric soils are typically defined as:

1. all Histels except Folistels and all Histosols except Folists;
2. soils in aquic suborders, great groups or subgroups, Albolls suborder, Aquisalids, Historthels and Histoturbels great group and Cumulic or Pachic subgroups that are:
    a. somewhat poorly drained with a water table equal to 0.0 m from the surface during the growing season, or
    b. poorly drained or very poorly drained and have either:
        i. water table equal to 0.0 m during the growing season if textures are coarse sand, sand or fine sand in all layers within 0.5 m (20 inches) of the soil surface, or for other soils; or
        ii. water table at less than or equal to 0.15 m (6 inches) from the surface during the growing season if permeability is equal to or greater than

0.15 m (6 inches) per hour in all layers within 0.5 m (20 inches) of the soil surface; or

iii. water table at less than or equal to 0.3 m (12 inches) from the surface during the growing season, if permeability is less than 0.15 m (6 inches) per hour in any layer within 0.5 m (20 inches) of the soil surface; or

3. Soils that are frequently ponded for long or very long duration during the growing season or soils that are frequently flooded for long or very long duration during the growing season.

The delineation of hydric soils in the United States

Wetlands are delineated in the United States for regulatory purposes and as a component of restoration programmes. There are two main regulatory documents used to delineate wetlands, *Field Indicators of Hydric Soils in the United States* (Hurt *et al*. 2003), which is used to administer the *Food Security Act 1985*, and The US Army Corp *Wetland Delineation Manual*, which is used to administer the *Clean Waters Act 1972*.

Field manual of hydric soils

The delineation of hydric soils in the United States is aided by the list of hydric soil indicators as published by Hurt *et al*. (2003). The list of indicators is considered to be dynamic, with changes made when necessary. They are specific to land resource regions or major land resource areas of the United States and are used to identify the hydric soil component of wetlands. The indicators should be used in conjunction with lists of hydric soils, soil maps and landscape position when the wetland is being characterised (Mausbach and Parker 2001). They must be confirmed by field investigations (Hurt and Puckett 1992). However, all hydric soils must satisfy the requirements of the definition, they must be saturated or inundated during the growing season and the soil must be anaerobic (Hurt and Carlisle 2001). Such conditions favour hydrophytes but would damage most plants unless drained (Tiner 1999). The majority of hydric soils possess morphological indicators that result from repeated periods of saturation and/or inundation for more than a few days (Hurt *et al*. 2003).

The indicators focus on visible features that form by one of three processes:

1. the accumulation of organic carbon
2. the depletion and/or concentration of iron and manganese
3. the reduction of SO4.

The soil is classified as hydric when any of the three indicators are present, regardless if the soil is still periodically saturated and anaerobic or not. If these three indicators are not present, the soil may still be hydric. Specific requirements have been devised for colour, thickness, depth and organic carbon content (Vepraskas 1999). Four categories of soil are accounted for:

1. all soils
2. sandy soils
3. loamy
4. clayey soils and test indicators.

Wetland Delineation Manual

The purpose of this manual is to provide users with guidelines and methods to determine if an area is a wetland for purposes of Section 404 of the *Clean Waters Act 1972*. The US Army Corps of Engineers Wetland Delineation Manual lists a few simple readily identifiable properties of hydric soils, from Tiner (1999) these are:

1. a fibric material or sapric material surface layer 0.2 m (8 inches) or thicker
2. dominant colours in the mineral soil matrix of chroma 2 or less, if there are mottles (usually orangish, yellowish or reddish-brown) present
3. dominant colours in the mineral soil matrix of chroma 1 or less, if there are no mottles present
4. organic streaking or blotchiness of organic soils
5. the smell of hydrogen sulfide in the upper 0.3 m (1 foot) of the soil
6. observed reduction (ferrous iron test).

The *Wetland Delineation Manual* is not as comprehensive as Hurt *et al*. (2003).

5. Review of classification and delineation of wetland soils

Europe

In Europe, wetland soils are characterised by four hydromorphic features depending on the nature and size of the catchment area:

1. halomorphic
2. gypsimorphic
3. calcimorphic
4. redoximorphic.

Four types of water saturation have been distinguished:

1. permanent and total saturation (submerged soils)
2. temporary and total saturation (floodwater soils)
3. permanent saturation in the subsoil (groundwater soils)
4. permanent-temporary saturation in the top soil (surface water soils) (Blume and Schlichting 1985).

Submerged soils contain humic horizons; however, the substratum is not soil and is saturated with water, or covered by water at some time. In groundwater soils, the subsoil is permanently wet and is blue-green colour in the finer textured soils due to Fe(II) and Fe(III) hydroxy salts, black if iron sulfides have accumulated, white in carbonate-rich material due to calcite or siderite and in sands the subsoil is generally light grey to white and often low in iron and manganese. Reddish-brown colours can be present in the capillary fringe and along root channels. The surface aggregates of subsoil clays of the surface water soils are often bleached and depleted of iron and manganese oxides with ped interiors enriched with iron and manganese oxides and the profile sometimes appears marbled. Coarse textured surface water soils are often brownish-grey with iron and manganese concretions (Blume and Schlichting 1985).

Table 4: A provisional three-class system for determining the degree of wetness of wetland soils based on soil morphology (Kotze et al. 1996).

			Degree of wetness	
Soil depth (mm)	**Soil indicator**	**Temporary**	**Seasonal**	**Permanent/semi-permanent**
0–100	Matrix chroma Mottles Organic carbon	0–3 few/nil low/ intermediate	0–2 common intermediate	0–1 nil/few high
300–400	Matrix chroma Mottles	0–2 few	0–1 many	0–1 nil/few

High organic carbon indicated if greater than 5%, often exceeding 10%.
Low organic carbon indicated if less than 2%.

5

South Africa

Kotze *et al*. (1996) recommended the use of soil taxonomy rather than the South African Soil Classification system for describing South African hydric soils, largely because it accounts for depth of waterlogging. It was, however, stated that soil taxonomy did not adequately account for the degree of wetness.

A three-class water-regime classification, with distinguishing characteristics, was developed and was to undergo further field testing and modification (Kotze *et al*. 1996). However, no further scientific literature was found on this system.

International classification

The World Reference Base for Soil Resources (FAO/ISRIC/ISSS 1998), formerly the FAO World Soil Classification System, classifies waterlogged soils as gleysols and uses diagnostic properties such as gleyic properties and organic soil material to further classify waterlogged soils. Soils with gleyic properties develop if they are completely saturated with groundwater — unless drained — for a period that allows reducing conditions to occur, which may range from a few days in the tropics to a few weeks in other areas and shows a gleyic colour pattern.

6. Case studies

In September 2006, two wetlands in Queensland were sampled to illustrate the process and the research and development needed to establish a defensible regime of wetland delineation. Given that only two wetlands would be studied, contrasting sites (i.e. differing in landform, soil and hydrological influence) were selected to test concepts and issues relevant to the development of a defensible approach in Queensland. The sites were to be used as a pilot to evaluate the hydromorphic indicators identified during the literature review.

Sampling for each of sites was based across the wetland boundary from areas without seasonal saturation through to, where present, permanently wet areas. Detailed soil morphological description to depth and sampling was undertaken for a range of soil physical and chemical characteristics. The investigations identified wetland soil indicators at both case study sites. The number and frequency of indicators present varied significantly between the wetlands.

The results demonstrate that additional investigations across a comprehensive range of wetland types are necessary to highlight all factors influencing the development of soil indicators and to further quantify the relationship between hydromorphic soil indicators and the presence of a wetland. Given the unreliable rainfall in Queensland, a single site inspection of many wetlands may not be a realistic indication of the wetness regime regardless of the soil morphology.

Green Swamp

Green Swamp is located approximately 60 km north of Rockhampton in Central Queensland. The soils here are Vertosols; heavy black cracking alluvial clays, derived from Serpentinite. Table 5 provides an overview of the characteristics of the study site, while Table 6 provides a summary of the hydromorphic soil indicators that were observed.

Table 5: General site information — Green Swamp

Wetland name	Green Swamp
Location	MGA: E221358, N7463779 Zone 55
Climatic zone	Subtropical, with distinct wet and dry season Average annual rainfall 794 mm at Rockhampton
Vegetation association	Melaleuca quinquenervia (Paperbark Tea Tree)
Landform	Closed depression on alluvium derived from Serpentinite
Soil types	Vertosols
Period and frequency of inundation	Length of inundation unknown Frequency — 1 to 2 times per year but varies significantly
Nature of inundation	Overland flow
Water quality of surface and subsurface waters	Quality of surface water is good Subsurface water quality unknown
Fresh or saline ecosystems	Fresh

Table 6: Hydromorphic soil indicators for wetlands as observed as Green Swamp.

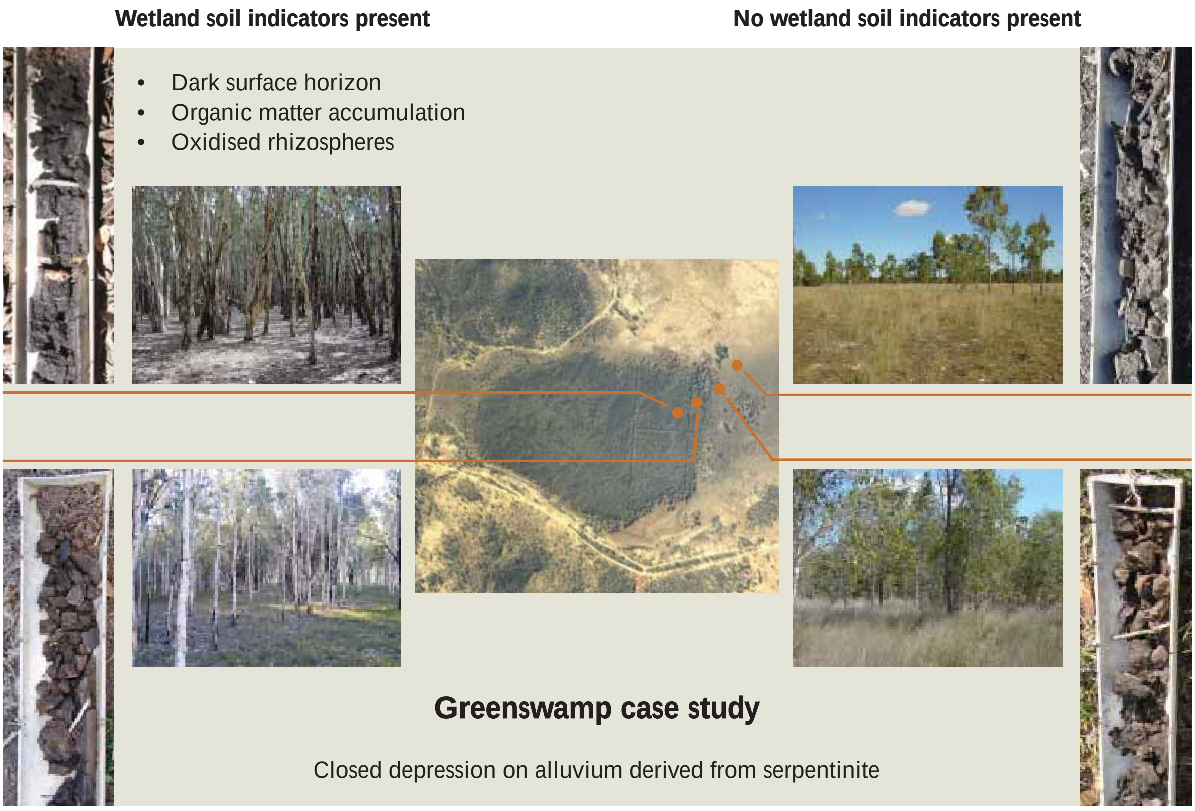
Indicator name	Description
Organic material	Organic carbon levels between 4 and 6% in surface horizons
Streaked organic matter	Not observed
Matrix colour	Black soil colour in upper 30 cm
Gleyed matrix	Not observed
Decreasing matrix chroma, mottle hue and chroma	Matrix chroma decreasing at depth with pale subsurface colours
Mottle intensity	No mottles observed in upper 0.30 m of profile Few mottles 2–10% observed at depth
Mottle and segregation boundaries	Faint mottles and segregations at depth with clear boundaries
Soil-water interface	Not observed
Segregations	2–10% <2 mm ferruginous segregations observed from 0.1–0.3 m
Oxidised rhizosphere	2–10%, 2–6 mm red-coloured rhizospheres observed in the top 0.2m
Pore linings	Not observed
Groundwater table depth	Not encountered to 3m
Redox potential	n/a
Ferrous iron detection	n/a
pH	Range 5.0 to 8.5
Cation exchange capacity	Approximately 9 meq/100 g soil for the surface horizons
Particle size analysis	Fine sandy light clay to medium heavy clay in upper 0.3 m, loamy surface
Soil oxygen	n/a
Hydrogen sulfide	Not observed
Methane gas	Not observed
Pyrite present	Not observed
Salt profile	Low salt levels in upper 0.3 m and salt bulge observed at depth in some profiles
Other properties	Calcareous segregations observed at depth Manganiferous segregations observed at depth in some profiles

6. Case studies

Summary of field observations

- soil colour is positive indicator of reducing conditions in upper 0.3 m
- impermeable Vertosol at Green Swamp may not become reducing even though water is ponded on the surface
- organic matter may fall down cracks, ensuring high OC levels down to 0.5 m
- unlikely to be particularly wet since 1991 and therefore irregularly inundated
- faint mottles at depth, which may indicate relict climate given recent dry period
- carbonate in some profiles suggests it is irregularly very wet at depth
- few fine segregations in some profiles in the upper 0.3 m indicate occasional reduction
- Melaleuca sp. present at the site shows adaptation to waterlogging with aerial roots evident at 0.3 m above ground level.

Figure 5 Green Swamp case study – site photo showing the location of the transect described through the wetland



Deepwater National Park case study site

Deepwater National Park, on the Central Queensland Coast north of Bundaberg, extends from Rules Beach to Agnes Water, with the study site located just south of Agnes Water. The soils investigated were located on the backslopes of coastal foredunes and comprised coarse textured Podosols and Tenosols. Table 7 provides an overview of the characteristics of the study site while Table 8 provides a summary of the hydromorphic soil indicators observed.

Table 7: General site information — Deepwater National Park.

Wetland Name	Deepwater National Park
Location	MGA: E393366, N7312155 Zone 56
Climatic zone	Mild subtropical climate Average annual rainfall 1123 mm at the Town of 1770
Vegetation association	Coastal wet heath, sedgeland communities
Landform	Backslopes of coastal foredunes
Soil types	Podosols and Tenosols
Period and frequency of inundation	Period of inundation — several weeks Frequency of inundation — several times per year
Nature of inundation	Predominantly groundwater inundation
Water quality of surface and subsurface waters	Good
Fresh or saline ecosystems	Fresh

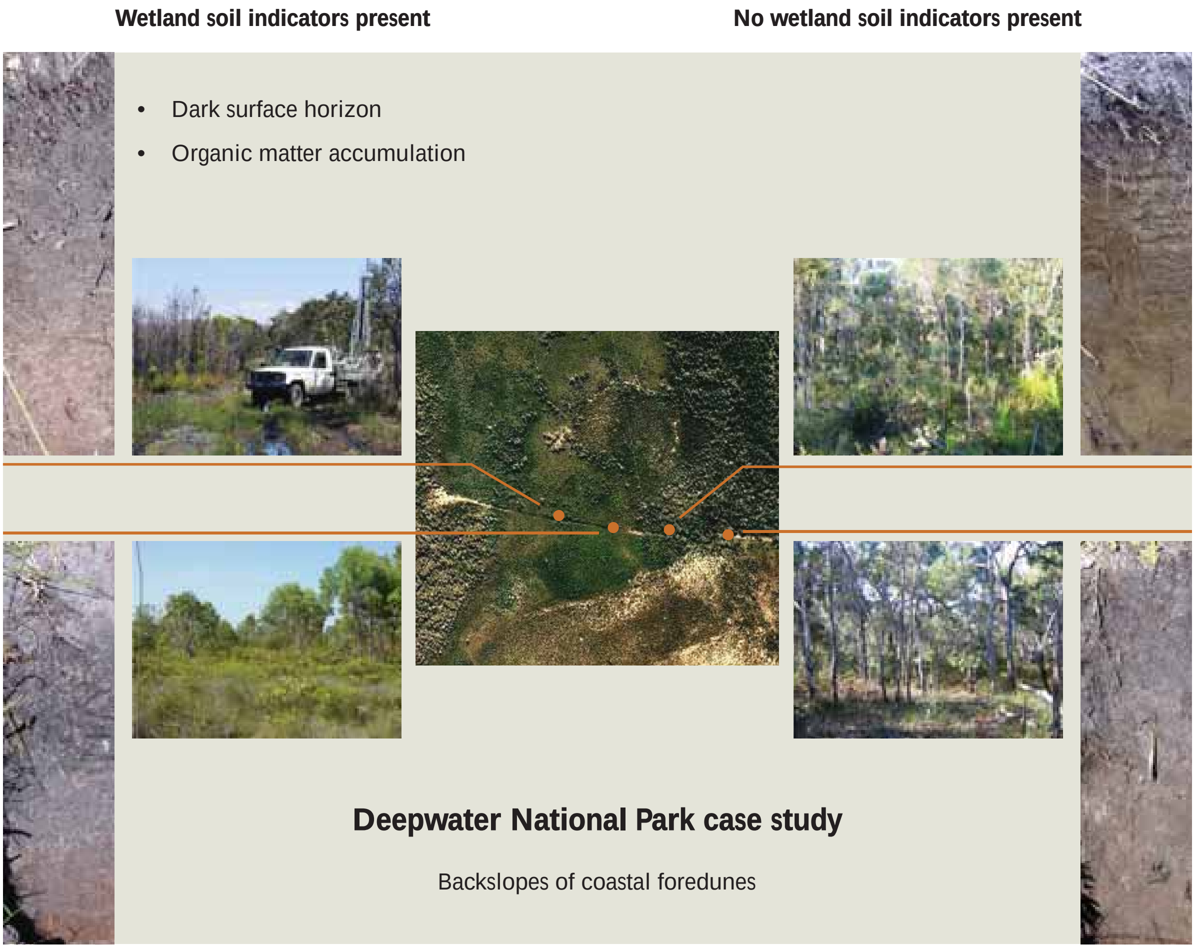
Table 8: Hydromorphic soil indicators for wetlands as observed at Deepwater National Par

Indicator name	Description
Organic material	Sapric material observed in upper 0.3 m of wet profiles. Organic carbon levels ranged from 6 to 16% in surface horizon
Streaked organic matter	Not observed
Matrix colour	Black soil colour in upper 0.3 m
Gleyed matrix	Not observed
Decreasing matrix chroma, mottle hue and chroma	Paler subsurface colours observed
Mottle intensity	No mottles observed in upper 0.3 m of profile Few 2–10% observed at depth in two profiles
Mottle and segregation boundaries	Faint mottles at depth with very diffuse boundaries
Soil-water interface	Evidence at surface
Segregations	10–20% <2 mm ferruginous segregations observed 0.4–0.5 m in one soil profile and at depth in another profile
Oxidised rhizosphere	Not observed
Pore linings	Not observed
Groundwater table depth	Ranged between 0.2–1.1 m over 400 m transect
Redox potential	n/a
Ferrous iron detection	n/a
pH	Range 4.5–6, predominantly strongly acidic in surface horizon
Cation exchange capacity	Approximately 4 meq/100 g soil, in surface horizon
Particle size analysis	Loams: loamy sands to sandy clay loams in upper 0.3 m
Soil oxygen	n/a
Hydrogen sulfide	Not observed
Methane gas	Not observed
Pyrite present	Not observed
Salt profile	Low salt levels in upper 0.3 m
Other properties	Debil debil and swamp hummock present

Summary of field observations

- high organic carbon levels observed in sandy soil indicate reducing
- highly variable water table depth
- soil colour is positive indicator of reducing conditions in upper 0.3 m
- poor external drainage with high water tables or located adjacent to a water body with regular inundation or have high groundwater levels
- limited mottling, although this is expected as there is likely to be limited iron in the coarse textured soil
- low CEC provides little useful information.

Figure 6: Deepwater National Park case study — site photo showing the location of the transect described through the wetland



7. Research programme for wetland soils

Knowledge gaps evident in Queensland

From the literature and the results of the pilot study, it is obvious that there are some significant knowledge gaps in Queensland in relation to the delineation of wetland boundaries in a manner that is scientifically robust and legislatively defensible. Beyond this literature review there has been no previous review to establish a set of indicators of hydromorphic soils to predict the boundary of a wetland in Queensland. A defensible set of criteria of hydromorphic indicators would require an extensive field-based study across the major wetland types found in Queensland and a variety of soil groups, land forms, climates and vegetation associations need to be documented. The concept for hydromorphic soils for Queensland should be defined and field tested, after which the characteristics that are indicative of most Queensland wetland soils could be documented.

The definition of hydromorphic soils

Currently, the definition of a wetland includes a subsection relating to soils. There is a reliance on the establishment of anaerobic conditions in the upper layers, but the concept of 'upper layers' has not been defined. In the United States, the concept of the upper layers is generally restricted to the upper 0.3 m of the soil profile, typically the root zone. Confirmation is required that the upper 0.3 m of the soil profile is appropriate for Queensland wetlands.

The definition of wetland soils refers to the establishment of anaerobic conditions. For a soil to be anaerobic, only the reduction of oxygen is required. However, many of the indicators of wetland soils will not be found until the soil is further reduced, with the reduction of sulphur-based features only occurring after the reduction of iron and manganese. While many features indicating reduction can be measured these techniques are costly and often require extended monitoring periods for validation. To accommodate this issue, specific publications in the United States have been developed that detail soil indicators which are focused on visible features and are easy to recognise in the field. Under wet conditions, the soil could be anaerobic with other elements still in an oxidised state or, alternatively, the soil could be reduced but show no visible features. For a variety of reasons a soil may be wet enough for hydrophytic plants to become established; however, there will be limited or no visible indicators.

The definition of a wetland also relies on the term 'long enough', which is not defined and it is suspected that some wetland soils may be wet long enough for the establishment of hydrophytic plants, but not for the anaerobic conditions to be established. Many Queensland wetlands are located in climatic zones with a highly variable rainfall. Consequently, the status of these wetlands may need to be assessed after a rainfall event has occurred.

The definition of hydric soils as used in the United States refers to minimum periods of inundation during the growing season, necessary due to the cold conditions where temperatures are often below 5°C. Warmer climatic conditions in Queensland mean the reference to a growing season is not warranted, and therefore the duration of any wetness period is undefined and may be open to interpretation. The results of the pilot study revealed that hydrology can change significantly over short distances. Consequently, further research is needed to assess hydrological interactions for wetland delineation purposes.

The current definition would be difficult to interpret in areas where vegetation has been cleared, hydrology information is absent, the soil profile is dry and there are misleading or no visible indicators present in the soil. Such a situation may exist in a coarse textured soil that has been drained and cleared for the production of sugarcane and the organic matter has broken down.

Should additional wetlands legislation be developed in Queensland, it will be important that a set of reliable indicators be devised to support it (significant legislation is already in place to manage wetlands).

The influence of climatic zones

Wetlands in semi-arid and arid Australia need to be assessed, and while many soils in these areas are only periodically wet (e.g. one in every ten years), some may remain wet for months or even years. Some of these soils may not be moist enough or reduced for a sufficient period to establish morphological indicators, but will support a variety of hydrophytic vegetation and wetland wildlife during periods of inundation. Semi-arid and arid wetlands will require field testing, but verification may only be an option

7. Research programme for wetland soils

during periodic inundation events. The resources required to monitor semi-arid and arid areas over an extended period need to be considered.

Organic accumulations

The assessment of limits for organic accumulations need to be considered with the view that the limits, as discussed in the Australian Soil Classification (Isbell 1996), are appropriate for wetland delineation purposes. This will require further field assessment across a variety of fine, medium and coarse textured soils.

Alterations to hydrology

How to deal with artificially drained wetlands, lowered water tables or situations where surface hydrology alterations mean that the wetland is no longer wet are issues warranting further consideration. The impacts of climate change on hydrology also merit deliberation, particularly if the soil retains visible indicators of reduction for many years. This is of particular relevance for the delineation of wetland margins. Furthermore, rising water tables due to human-induced changes to the hydrology need to be considered along with whether these new ecosystems also constitute a wetland.

In the United States, the soils of drained wetlands are identified as drained hydric soils, particularly if they retain some indicators of saturation. However, as they can no longer support hydrophytic vegetation they are no longer jurisdictional wetlands. This issue will need to be considered further in relation to any wetland restoration programmes.

The pilot study

The pilot study has provided a limited insight into two of Queensland's many wetlands. On the basis of the results, additional investigations are required to further quantify the relationship between hydromorphic soil indicators and the presence of a wetland for a more comprehensive range of wetland types. The research programme must focus on testing the relationships predicted by the literature for a range of wetland types and assess indicator changes across soil groups, climate, landform and vegetation associations. A further 18-month period of investigation has been proposed[10] to build upon the concepts of this pilot study. However, given the complexities associated with soil, landform and vegetation associations and the significant climatic range in Queensland, it is unlikely that the 18-month period will be long enough to produce a reliable, scientifically robust set of Queensland indicators. Many of the wetlands will require long-term monitoring so that accurate trends can be identified.

Field technical manual

Any future investigations into the identification of hydromorphic soil indicators and wetland presence in Queensland should work toward developing a technical manual to feature both simple and comprehensive assessment for more detailed studies. The simple field assessment of wetlands should focus on visual morphological characteristics.

Changes and additions to any such technical manual are expected to be regular due to previously unidentified hydromorphic soil indicators being recognised and the need to modify existing definitions as a result of long-term soil and hydrological observations. Consequently, the dynamic nature of the manual may have implications in the development and robustness of any future review to wetland definition[11]. Consultation will be necessary with those involved in any such review process to ensure the applicability of the wetland definition is not compromised.

[10]The Queensland Wetlands Programme has funded a second project to study that began in January 2007 and runs to June 2008. Further details can be obtained by contacting the author or the Queensland Wetlands Programme. (www.environment.gov.au/water/environmental/wetlands/programs/qwp.html).

[11]At the time of publishing, the definition of a wetland for Queensland was under review.

8. Summary of findings

This section contains a summary of the findings as developed by the literature review and case studies. The findings of the twelve-month project, Soil Indicators of Wetlands: Status, Margins and History, funded by the Queensland Wetlands Programme, will be used to advance wetland conservation and management in Queensland. The summarised findings from this report are as follows.

Findings from the literature review

- Some characteristics of hydromorphic soils such as redox potential, soil oxygen content and water table depth can be reliably and accurately measured. However, such procedures are expensive and rely on extensive monitoring periods for meaningful results.
- An assessment of morphological characteristics resulting from the accumulation or loss of compounds of iron, manganese, sulfur and carbon can be of use to characterise hydromorphic soils. Evidence may be presented as an accumulation of organic matter and can be associated with soil colour, mottles and segregations, or as odour associated with the reduction of sulfur and carbon.
- Soil morphology can be used to identify wetland status in situations where there is unreliable hydrology data or where the vegetation is transitional, has been cleared or is poorly characterised.
- A list of general indicators-based soil morphology and other hydromorphic indicators have been developed, along with some general information on soils with inconclusive or misleading indicators of soil saturation.
- The United States system of delineating wetland soils on the basis of soil morphology provides some useful information, but is subjective and is largely of use in localised areas or differing climatic zones within the United States. Further research is needed to predict and assess in the field the relationships between wetland types and the changes evident across soil groups, landform, climate and vegetation associations typical of Queensland conditions.
- The system of classifying organic accumulations with the Australian Soil Classification (Isbell 1996) requires more extensive site investigations and monitoring over a series of wet and dry periods to test its appropriateness for wetland delineation purposes.
- Should additional legislation be developed in Queensland to protect wetlands, further research in wetland soil indicators will be warranted.
- Any relationships predicted between soil indicators and wetland margins must be assessed against the changes evident across soil groups, landform, climate and vegetation associations for a series of wetland types to ensure that there can be a scientifically robust and defensible means of delineating wetlands in Queensland.
- The duration of wetness is not defined in the current wetland definition. Issues associated with defining the duration of wetness in Queensland, where the rainfall can be highly variable, complicates this problem. Additional wetland definition components, such as upper layers, are also undefined. Further consideration of these issues is required particularly if any legislation relying on the definition of a wetland is proposed.

Findings from the case studies

- Results from two study sites established that some soil indicators used to delineate wetland soils in the United States appear to be applicable to soils from the case study sites.
- A range of laboratory analyses were conducted on soil samples from each site. Organic carbon levels of greater than 3% provided the most conclusive measured indicator of reduction. There appeared to be some correlation between levels of organic carbon observed in the surface samples and the proximity of the site to the wetland (i.e. measured organic carbon levels increased in those sites located further into the wetland).
- Preliminary results from the case studies show promise but suggest additional investigations are required to identify if there is a predictable relationship between hydromorphic soil indicators and the presence of a wetland in Queensland, particularly those that are periodically saturated.

9. Glossary of terms

Anaerobic conditions Conditions whereby air (oxygen) is excluded, usually by waterlogging, and reducing conditions prevail.

Growing season The portion of the year when soil temperatures are above biologic zero at 50 cm.

Hydric or hydromorphic soils Soils that formed under conditions of saturation, flooding or ponding long enough during the growing season to develop anaerobic conditions in the upper part of the soil (Federal Register, July 1994).

Hydrophytic vegetation Plants adapted to and dependent on living in wet conditions for at least part of their lifecycle.

Jarosite An acidic pale yellow (straw or butter coloured) iron sulfate mineral: $KFe_3(SO_4)_2(OH)_6$. Jarosite is a by-product of the acid sulfate soil oxidation process, formed at pH less than 3.7 and commonly found precipitated along root channels and other soil surfaces exposed to air.

Oxidation Oxidation is the addition of oxygen, removal of hydrogen, or the removal of electrons from an element or compound. In the environment, organic matter is oxidised to more stable substances. The opposite of reduction.

Redox Redox reactions include all chemical processes in which atoms have their oxidation number (oxidation state) changed. The term comes from the two concepts of reduction and oxidation.

Redoximorphic features Devised to account for mottles and low chroma colours formed by the reduction and oxidation of Mn and Fe compounds.

Reduction The addition of hydrogen, removal of oxygen, or the addition of electrons to an element or compound. Under anaerobic conditions (no dissolved oxygen is present), sulfur compounds are reduced to odour-producing hydrogen sulfide (H_2S and other compounds. The opposite of oxidation.

Upper layers The upper 30 cm of the soil profile, which generally is the root zone of plants, as defined in the United States.

Wetland Wetlands are areas of permanent or periodic/intermittent inundation, 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 6 m.

To be classified as a wetland the area must have one or more of the following attributes:

- at least periodically the land supports plants or animals that are adapted to and dependent on living in wet conditions for at least part of their lifecycle
- the substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers
- the substratum is not soil and is saturated with water, or covered by water at some time.

10. Bibliography

Ahern, C.R., McElnea, A.E. and Sullivan, L.A. (2004). *Acid Sulfate Soils Laboratory Methods Guidelines*. Department of Natural Resources, Mines and Energy, Indooroopilly, Queensland.

Armstrong, W. (1982). Waterlogged soils. E*nvironment and Plant Ecology*. John Wiley and Sons, Chichester, UK.

Bailey, R. and Copeland, O. (1961). Low flow discharges and plant cover relations on two mountains watersheds in Utah. *International Association Scientific Hydrology*, 51: 267–278.

Blackman, J.G., Perry, T.W., Ford, G.I., Craven, S.A., Gardiner, S.J. and De Lai, R.J. (1999). C*haracteristics of Important Wetlands in Queensland. Environmental Protection Agency, Queensland.*

Blume, H.P. and Schlichting, E. (1985). Morphology of wetland soils. *Wetland Soils: Characterisation, Classification and Utilisation*. Los Banos: International Rice Research Institute (IRRI).

Borman F.H. and Likens G.E. (1969). The watershed-ecosystem concept and studies of nutrient cycling. *The ecosystem concept in natural resources management*. G.M. VanDyne, (ed.). Academic Press, New York. pp 49–76.

Brinson, M.M. (1993). A hydrogeomorphic classification for wetlands. *US Army Corps of Engineers, Technical Report* WRP-DE-4, 79p.

Carter, V. (1996). Wetland hydrology, water quality, and associated functions. *In: National Water Summary on Wetland Resources. Water Supply Paper 2425*. United States Geological Survey, Washington, DC.

Castenson, L. and Rabenhorst, M.C. (2006). Indicator of reduction in soil (IRIS): evaluation of a new approach for assessing reduced conditions in soil. *Soil Science Society America Journal*, 70: 1222–1226.

Chen, C.C., Dixon J.B. and Turner F.T. (1980). Iron coatings on rice roots: mineralogy and quantity influencing factors. *Soil Science Society of America Journal*, 44: 635–639.

Christie, J. and Hausmann, S. (2003). Various state reactions to the SWANCC decision. *Wetlands*, 23(3) 653–662.

Clausnitzer, D., Huddleston H.J., Horn, E., Keller, M. and Leet, C. (2003). Hydric soils in a South Eastern Oregon vernal pool. *Soil Science Society of America Journal*, 67: 951–960.

Cogger, C.G. and Kennedy, P.E. (1992). Seasonally saturated soils in the Puget lowland. I. Saturation, reduction and color patterns. *Soil Science*, Vol. 153, No. 6.

Collins, J.F. and Buol, S.W. (1970). Effects of fluctuations in the Eh-pH environment on iron and/or manganese equilibria. *Soil Science*, 110: 111–118.

Collins, M.E. and Kuehl, R.J. (2001). Organic matter accumulation and organic soils. In: J.L. Richardson and M.J. Vepraskas (eds), *Wetland Soils: genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

Couto, W, Sanzonowicz, C. and de O. Barcellos, A. (1985). Factors affecting oxidation-reduction processes in an oxisol with a seasonal water table. *Soil Science Society America Journal*, 49: 1245–1248.

Coventry R.J. and Williams J. (1984). Quantitative relationships between morphology and current soil hydrology in some alfisols in semiarid tropical Australia. *Geoderma*, 33, pp 191–218.

Crown, P.H. and Hoffman, D.W. (1970). Relationship between water table levels and type of mottles in four Ontario gleysols. *Canadian Journal of Soil Science*, 50: 453–455.

D'Armour, D.V., Stewart, S.R. and Huddleston, J.H. (2004). Saturation, reduction and the formation of iron-manganese concretions in the Jackson-Frazier Wetland, Oregon. *Soil Science Society America Journal*, 68: 1012–1022.

Davis, L. and Negus, P. (2006). *Draft Understanding Ecological and Biophysical Processes in Queensland's Wetlands: a Literature Review and Gap Analysis*. Department of Natural Resources and Water, Indooroopilly, Queensland, Australia.

10. Bibliography

Dear, S.E., Moore, N.G., Dobos, S.K., Watling, K.M. and Ahern, C.R. (2002). Soil Management Guidelines, In: *Queensland Acid Sulfate Soil Technical Manual*. Department of Natural Resources and Mines, Indooroopilly, Queensland, Australia.

Department of Environment and Heritage, Queensland (2006). www.greenhouse.gov.au/science

Diers, R. and Anderson, J.L. (1984). Part I, Development of soil mottling. *Soil Survey Horizons*, winter 1984, pp 9–12.

Drexler, J.Z. and Ewel, K.C. (2001). Effect of the 1997–1998 ENSO-related drought on hydrology and salinity in a Micronesian wetland complex. *Estuaries*, Vol. 24, Issue 2, pp 347–356.

Ellis, J. and Clairain, E.J. (2002). *Hydrogeomorphic Approach to Assessing Wetland Functions: Guidelines for Developing Regional Guidebooks*. US Army Engineer Research and Development Centre, Vicksburg, MS.

Environment Australia (2001). *A Directory of Important Wetlands in Australia* 3rd Edition. Environment Australia, Canberra.

Environmental Laboratory. 1987. *Corps of Engineers Wetland Delineation Manual, Technical Report Y-87-1*. U.S. Army Engineer Waterways Experiment Station, Vicksburg, MS.

Evans, C.V. and Franzmeier, D.P. (1986). Saturation, aeration and colour patterns in a toposequence of soils in North-central Indiana. *Soil Science Society America Journal*, Vol. 50: 975–980.

FAO/ISRIC/ISSS (1998). World Reference Base for Soil Resource, FAO, Rome.

Faulkner, S.P., Patrick Jr, W.H. and Gambrell, R.P. (1989). Field techniques for measuring wetland soil parameters. *Soil Science Society of America Journal*, 53 (3), pp 883–890.

Fey, M.V. (1983). Hypothesis for the pedogenic yellowing of red soil materials. *Tech. Commun.* South Africa Department Agiculture and Fisheries. 180: 130–136.

Finlayson, C.M, Davidson, N.C. and Stevenson, N.J. (eds) (2001). *Wetland inventory, assessment and monitoring: Practical techniques and identification of major issues.* Proceedings of Workshop 4, 2nd International Conference on Wetlands and Development, Dakar, Senegal, 8–14 November 1998. Supervising Scientist Report 161, Supervising Scientist, Darwin.

Fisher, H.M. and Stone, E.L. (1991). Iron oxidation at the surfaces of slash pine roots from saturated soils. *Soil Science Society of America Journal*, 55: 1123–1129.

Franzmeier, D.P., Yahner, J.E., Steinhardt, G.C. and Sinclair Jr, H.R. (1983). Color patterns and water table levels in some Indiana soils. *Soil Science Society of America Journal* 47: 1196–1202.

Gambrell, R.P. and Patrick Jr., W.H. (1978). Chemical and microbiological properties of anaerobic soils and sediments. In: *Plant life in anaerobic environments*. Ann Arbor Science Publ. Inc. Ann Arbor, MI.

Gilman, K. (1994). Hydrology and wetland conservation. Water Science Series, Wiley, New York. p 101.

Goodrick, G. (1970). A survey of wetlands of coastal New South Wales. *CSIRO Wildlife Research Technical Memorandum No. 5*. Canberra.

Guthrie, R.L. and Hajek, B.F. (1979). Morphology and water regime of a Dothan soil. *Soil Science Society of America Journal*, 43:142–144.

Harmen, K. and van Breeman, N. (1975). Translocation of iron in acid sulfate soils, II. Production and diffusion of dissolved ferrous iron. *Soil Science Society America Proceedings*, 39: 1148–1153

Harter, R.D. (1968). Phosphorus sorption sites in soils. *Soil Science Society of America Journal*, 31: 630–631.

Hayes Jr, W.A. and Vepraskas, M.J. (2000). Morphological changes in soils produced when hydrology is altered by ditching. *Soil Science Society of America Journal*, 64: 1893–1904.

He X, Vepraskas. M.J., Lindbo, D.L. and Skaggs R.W. (2003). A method to predict soil saturation frequency and duration from soil colour. *Soil Science Society of America Journal*, 67: 961–969.

Holden, J. (2005). Peatland hydrology and carbon release: why small-scale process matters. *Philosophical Transactions: Mathematical, Physical and Engineering Sciences (Series A), 363*, pp 2891–2913.

Hutchinson, G.E. (1957). *A treatise on Limnology Volume I.* Wiley, New York.

Hurt, G.W. and Brown, R.B. (1995). Development and the application of hydric soil indicators in Florida. *Wetlands*, 15(1) pp 74–81.

Hurt, G.W. and Carlisle, V.W. (2001). Delineating Hydric Soils. In: J.L. Richardson and M.J. Vepraskas (eds), *Wetland Soils: Genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

Hurt, G.W. and Puckett, W.E. (1992). Proposed hydric soil criteria and their field identification. In: *Proceedings of the Eighth International Soil Correlation Meeting (VIII ISCOM): Characterisation, classification and Utilisation of Wet Soils*. Louisiana and Texas, October 1990, USDA SCS, USA.

Hurt, G.W. and Richardson, J.L. (1998). Hydric soils of the United States: A guide to their recognition. USDA National Resources Conservation Service, Fort Worth, Texas. CD Rom.

Hurt G.W., P.M. Whited and R.F. Pringle (eds) (2003). *Field Indicators of Hydric Soil in the United States, version 5.01*. USDA, NRCS, Fort Worth, Texas.

Intergovernmental Panel on Climate Change (2001). *The third assessment report, Climate Change 2001*. Cambridge University Press, Cambridge.

Isbell, R.F. (1996). T*he Australian Soil Classification*. CSIRO Publishing, Collingwood, Victoria.

Isbell, R.F., McDonald, W.S. and Ashton, L.J. (1997). *Concepts and Rationale of the Australian Soil Classification.* ACLEP, CSIRO Land and Water, Canberra.

Jacobs, P.M., West, L.T. and Shaw, J.N. (2002). Redoximorphic features as indicators of seasonal saturation, Lowndes County, Georgia. *Soil Science Society of America Journal*, 66: 315–323.

Jenkinson, B.J. and Franzmeier, D.P. (2006). Development and evaluation of Fe-coated tubes that indicate reduction in soils. S*oil Science Society America Journal*. 70: 183–191.

Karathanasis, A.D., Thompson, Y.L. and Barton, C.D. (2003). Long term evaluations of seasonally saturated wetlands in Western Kentucky. *Soil Science Society of America Journal* 67: 662–673.

Keeley, J.E. and Moreton, B.A. (1982). Distribution of diurnal acid metabolism in submerged aquatic plants outside the genus Isoetes. *Photosynthetica* 16(4): 546–553.

Kotze, D.C., Klug, J.R., Hughes, J.C. and Breen, C.M. (1996). Improved criteria for classifying hydric soils in South Africa. *South African Journal of Plant Soil* 13(3).

Lafleur, P.M., McCaughey, J.H., Joiner, D.W., Bartlett, P.A. and Jelinski, D.E. (1997). Seasonal trends in energy, water and carbon dioxide fluxes at a northern boreal wetland. J*ournal of Geophysical Research* 102, 29008–29020.

Lemon, E.R. and Erickson, A.E. (1952). The measurement of oxygen diffusion in the soil with a platinum microelectrode. *Soil Science Society of America Journal*. Vol. 16: 160–3.

Lemon, E.R. and Erickson, A.E. (1955). Principle of the platinum microelectrode as a method of characterising soil aeration. *Soil Science Journal*. Vol. 79: 383–92.

Mausbach, M.J. and Parker, W.B. (2001). Background and history of the concept of hydric soils. In: J.L. Richardson and M.J Vepraskas (eds), *Wetland Soils: Genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

McDonald, R.C., Isbell, R.F., Speight, J.G., Walker, J. and Hopkins, M.S. (1990). *Australian Soil and Land Survey Field handbook*, Second edition. Inkata Press, Melbourne.

10. Bibliography

Meek, B.D., MacKenzie, A.J. and Grass, L.B. (1968). Effects of organic matter, flooding time and temperature on the dissolution of iron and manganese from soil *in situ*. *Soil Science Society of America Journal*, 57: 140–149.

Megonical, J.P., Partrick Jr., W.H. and Faulkner, S.P. (1993). Wetland identification in seasonally flooded forest soils: soil morphology and redox dynamics. *Soil Science Society of America Proceedings*, Vol. 32.

Megonical, J.P., Faulkner, S.P. and Partrick Jr., W.H. (1996). The microbial activity season in southeastern hydric soils. *Soil Science Society of America Journal*, 60: 1263–1266.

Mendelssohn, I.A., Kleiss, B.A. and Wakely, J.S. (1995). Factors controlling the formation of oxidised root channels: a review. *Wetlands*, 15: 37–46.

Mitsch, W.J. and Gosselink, J.G. (1993). *Wetlands*, Van Nostrand Reinhold, New York.

Mitsuchi, M. (1976). Characteristics and genesis of nodules and concretions occurring in soils of the R Chinit area, Kompong Thom Province Cambodia. *Soil Science and Plant Nutrition*, 22: 409–421.

Moller, H.S. (1999). Restoration as an element of national planning for wetland conservation and wise use. Technical Session II, Paper 4: *7th Meeting of the Conference of Contracting Parties to the Convention of Wetlands (Ramsar, Iran, 1971)*. San Jose, Costa Rica. Doc. 17.4.

Montgomery, J.A., Tandarich, J.P. and Whited, P.M. (2001). Use of soil information for hydrogeomorphic assessment. In: J.L. Richardson and M.J. Vepraskas (eds), *Wetland Soils: Genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

Moormann, F.R. and Van de Wetering, H.T.J. (1985). Problems in characterising and classifying wetland soils. In: *Wetland Soils: Characterisation, Classification and Utilisation*. Los Banos: International Rice Research Institute (IRRI).

Mortimer, C.H. (1941). *Journal of Ecology*, 29: 280–329.

National Technical Committee for Hydric Soils (1994). *Hydric soils of the United States*. US Government Printing Office, Washington DC, USA USDA-SCS, Miscellaneous Publication Number 1491.

Novitzki, R.P. (1979). Monthly and annual water budgets on Lake Wingra, Madison, Wisconsin, 1972–77. *US Geological Survey, Madison, WI. Water Resources Division*. Issue WRD/WRI-80/008, WRI-79-100, 1979, 38p.

Paijmans, K., Galloway, R.W., Faith, D.P., Fleming, P.M., Haantjens, H.A., Heyligers, P.C., Kalma, J.D. and Löffler, E. (1985). *Aspects of Australian Wetlands*. Division of Water and Land Resources Technical Paper No. 44, CSIRO, Melbourne.

Parker W.B., Faulker S.P. and Patrick Jr., W.H. (1985). Soil wetness and aeration in selected soils with aquic moisture regimes in the Mississippi and Pearl River deltas. In: *Wetland Soils: Characterisation, Classification and Utilisation*. Los Banos: International Rice Research Institute (IRRI).

Patrick Jr., W.H. (1994). From Wastelands to Wetlands. *Journal of Environmental Quality*, Vol. 23, Issue 5.

Pickering, E.W. and Veneman, L.M (1984). Moisture regimes and morphological characteristics in a hydrosequence in central Massachusetts. *Soil Science Society of America Journal*, 48: 113–118.

Ponnamperuma, F.N. (1972). The chemistry of submerged soils. *Advances in Agronomy*, 24 pp 29–96.

Ponnamperuma, F.N. (1984). Effects of flooding on soils. *Flooding and Plant Growth*, pp 1–44.

Ponnamperuma, F.N. and Castro R.U. (1964). *8th Trans. International Congress Soil Science*, 1964 Vol. 3 pp 379–386.

Pressey, R.L. and Adam P. (1995). A review of wetland inventory and classification in Australia. *Journal of Plant Ecology* 118: Numbers 1–2, pp 81–101.

Rabenhorst, M.C. and Parikh, S. (2000). Propensity of soils to develop redoximorphic colour changes. *Soil Science Society America Journal*, 64: 1904–1910.

10

Rabenhorst, M.C. and Burch, S.N. (2006). Synthetic iron oxides as an indicator of reduction in soils (IRIS). *Soil Science Society America Journal*, 70: 1227–1236.

Ramsar 2006. www.ramsar.org

Rayment, G.E. and Higginson, F.R. (1992). *Australian Laboratory Handbook of Soil and Water Chemical Methods*. Inkata Press, Melbourne.

Richardson, J.L., Arndt, J.L. and Montgomery, J.A. (2001). Hydrology of Wetland and Related Soils. *Wetland Soils: Genesis, Hydrology, Landscapes and Classification*. Los Banos: International Rice Research Institute (IRRI).

Sahrawat, K.L. (2004). Terminal electron acceptors for controlling methane emissions from submerged rice soils. *Communications in Soil Science and Plant Analysis*, Vol 35, Nos 9 and 10, pp 1401–1413.

Sammut, J. (1996). An overview of the ecological impacts of acid sulfate soils in Australia. In: *Proceedings 2nd National Conference of Acid Sulfate Soils*. Robert J Smith and Associates and ASSMAC, Australia.

Schlichting, E. (1973). Pseudogleye and gley-genese und nutzung hydromorpher boden. In: *Pseudogley and gley*. E. Schlichting and V. Schwertmann (eds). Proceedings of ISSS symposium.

Schwertmann, U. and Fanning, D.S. (1976). Iron–manganese concretions in hydrosequences of soils in loess Bavaria. *Soil Science Society America Journal*, 40: 731–738.

Skaggs, R.W., J.W. Gilliam and R.O. Evans (1991). A computer simulation study of pocosin hydrology. *Wetlands*, Vol. 11: 39–416.

Soil Science Society of America (2001). *Glossary of soil science terms*, SSSA, Madison WI, USA.

Soil Survey Staff (1975). *Soil Taxonomy: a basic system of soil classification for making and interpreting soil surveys*, Agric. Handbook. No. 46. USDA, US Govt Printing Office, Washington DC.

Soil Survey Staff (1996). *Keys to soil taxonomy, 7th edition*. US Department of Agriculture, Natural Resources Conservation Service, US Government Printing Office, Washington DC, USA.

Sprecher, S.W. and Mokma D.L. (1989). Refining the color waiver for Aqualfs and Aquepts. *Soil Survey Horizons*, 30(4): 89–91.

Stanton, J.P. (1975). A preliminary assessment of wetlands in Queensland. Technical Memorandum 75/10, Division of Land Use Research, CSIRO, Australia.

Stone, C. and Bacon, P.E. (1994). Relationships among moisture stress, insect herbivory, foliar cineole content and the growth of river red gum Eucalyptus camaldulensis. *Journal Applied Ecology*, 31: 604(12).

Stoops, G. and Eswaran, H. (1985). Morphological characteristics of wet soils. In: *Wetland Soils: Characterisation, Classification and Utilisation*. Los Banos: International Rice Research Institute (IRRI).

Sullivan, L.A. and Bush, R.T. (2004). Iron precipitate accumulations associated with waterways in drained coastal acid sulfate landscapes of eastern Australia. Marine and Freshwater Research, 55: 727–736.

Thompson, J.A. and Bell, J.C. (1998). Hydric conditions and hydromorphic properties within a mollisol catena in southeastern Minnesota. *Soil Science Society America Journal*, 62: 1116–1125.

Thompson, C.H. and Moore A.W. (1984). Studies in Landscape Dynamics in the Cooloola — Noosa River Area, Queensland. In: *CSIRO Division of Soils. Divisional Report No. 73*.

Tiner, R.W. (1999). *Wetland Indicators, a guide to wetland identification, delineation, classification and mapping.* Lewis Publishers, Boca Raton, Florida.

Tiner, R.W. (2003). Geographically isolated wetlands of the United States. *Wetlands*, Vol. 23 No. 3, pp 494–516.

Turner, F.T. and Patrick Jr., W.H. (1968). Chemical changes in waterlogged soils as a result of oxygen depletion. *Trans. 9th International Congress Soil Science*, (4) pp 53–56.

10. Bibliography

Turner R.K., Jeroen J.C.J.M., van den Bergh, Söderqvist T., Barendregt A., van der Straaten J., Maltby E. and van Ierland E.C. (2000). The values of wetlands: landscape and institutional perspectives. *Ecological Economics*, 35: 7–23.

US Army Corps of Engineers (2006). www.usace.army.mil/

US Climate Change Program (2006). www.climatescience.gov

USDA (1985). *Hydric Soils of the United States 1985*. USDA-SCS, US Govt Print Office, Wahington DC. USA.

USDA (2006). *Hydric Soils—Overview*, USDA-NRCS www.soils.usda.gov/use/hydric/overview.html

van Breeman, N. and Harmsen, K. (1975). Translocation of iron in acid sulfate soils: I. Soil morphology, and the chemistry and mineralogy of iron in a chronosequence of acid sulfate soils. *Soil Science Society of America Proceedings*, Vol. 39, pp 1140–1148.

Veneman, P.L.M., Spokas, L.A. and Lindbo, D.L. (1998). Soil moisture and redoximorphic features: a historical perspective. In: M.C. Rabenhorst *et al*. (ed). *Quantifying soil hydromorphology*, SSSA Special Publication 54, SSSA, Madison, WI.

Vepraskas, M.J. (1992). Redoximorphic features for identifying aquic conditions. North Carolina Agricultural Research Service, Raleigh, NC. *USA Technical Bulletin 301*.

Vepraskas, M.J. (2001). Morphological features of seasonally reduced soils. In: J.L. Richardson and M.J. Vepraskas (eds). *Wetland Soils: Genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

Vepraskas, M.J. and Wilding, L.P. (1983). Aquic moisture regimes in soils with and without low chroma colours. In: *Soil Science Society America Journal*, Vol. 47, pp 280–285.

Vepraskas, M.J. and Guertal, W.R. (1992). Morphological indicators of soil wetness. In: *Proceedings of the VIIIth International Correlation Meeting on Wetland Soils*, pp 307–312.

Vepraskas, M.J., Richardson, J.L., Tandarich, J.P. and Teets, S.J. (1999). Dynamics of hydric soil formation across the edge of a created deep marsh. Wetlands, Vol. 19, No. 1, pp 78–89.

Vepraskas, M.J. and Faulkner, S.P. (2001). Redox chemistry of hydric soils. In: J.L. Richardson and M.J. Vepraskas (eds). *Wetland Soils: Genesis, hydrology, landscapes and classification*. Lewis Publishers, Boca Raton, Florida.

Vepraskas, M.J., He X, Lindbo D.L. and Skaggs R.W. (2004). Calibrating hydric soil field indicators to long-term wetland hydrology. *Soil Science Society America Journal*, 68: 1461–1469.

Vepraskas, M.J. and Caldwell, P.V. (2006). Interpreting morphological features of wetland soils with hydrologic models. *Proceedings 18th World Congress of Soil Science*, July 2006. Philadelphia USA.

Wilding, L.P. and Rehage, J.A. (1985). Pedogenesis of soils with aquic moisture regimes. In: *Wetland Soils: Characterisation, Classification and Utilisation*. Los Banos: International Rice Research Institute (IRRI).

Willet, I.R. 1983. Oxidation reduction reactions. In: *Soils: an Australian Viewpoint. Division of Soils*, CSIRO Melbourne.

Winter, T.C. and Woo, Ming-Ko (1990). Hydrology of lakes and wetlands: Surface Water Hydrology: *The Geological Society of America*, Vols 0-1 pp 159–187

Winter, T.C. and LaBaugh, J.W. (1993). Hydrologic considerations in defining isolated wetlands. In: *Wetlands* 23, Issue 3, pp 532–540.

Yu, K. and Patrick Jr., W.H. (2004). Redox window with minimum global warming potential contribution from rice soils. *Soil Science Society America Journal*. 68: 2086–2091.

Zedler, P.H. (1987). The ecology of southern Californian vernal pools: A community profile. *Biological Rep*. 85 (7.11). United States Fish and Wildlife Service, Washington, DC.

Appendix 1. Organic material

From Isbell (1996):

These are plant-derived organic accumulations that are either:

1. saturated with water for long periods or are artificially drained and excluding live plant tissue: (i) have 18% or more organic carbon[12] if the mineral fraction is 60% or more clay; (ii) have 12% or more organic carbon if the mineral fraction has no clay; or (iii) have a proportional content of organic carbon between 12 and 18% if the clay content of the mineral fraction is between zero and 60%; or
2. saturated with water for no more than a few days and have 20% or more organic carbon.

Peat is further described in Isbell (1996) as follows.

Fibric peat

Undecomposed or weakly decomposed organic material.

Plant remains are distinct and identifiable.

Yields clear to weakly turbid water.

No peat escapes between fingers.

Hemic peat

Moderately to well-decomposed organic material.

Plant remains recognisable but may be rather indistinct and difficult to identify.

Yields strongly turbid to muddy water.

Amount of peat escaping between fingers ranges from none up to one-third.

Residue is pasty.

Sapric peat

Strongly to completely decomposed organic material.

Plant remains indistinct to unrecognisable.

Amounts ranging from about half to all escape between fingers.

Any residue is almost entirely resistant remains, such as root fibres and wood[13].

[12]Walkley-Black x 1.3 or a total combustable method (Rayment and Higginson 1992, Methods 6A1 or 6B2).

[13]In the United States, fibric material is commonly referred to as peat, hemic material is commonly referred to as mucky peat, and sapric material is commonly referred to as muck (Collins and Kuehl 2001).

Appendix 2. Redox potential

Redox is generally based on a hydrogen scale and is related in theory to the concentrations of oxidants and reductants in a redox reaction by the Nernst Equation:

$E_H = E^0 + 2.3\ [RT/nF] \log [\{ox\}/\{red\}]$

Where:

E_H = redox potential (mv)

E^0 = potential of reference, mv

R = gas constant = 81.987 cal $deg^{-1} mole^{-1}$

T = temp,°K

n = number of moles of electrons transferred

F = Faraday constant = 23 061 cal/mol-volt.

From Mitsche and Gosselink 1993:

Once the soil is flooded, there is a rapid decline in redox potential of the soil as the facultative and obligate anaerobic micro-organisms use soil components as final electron acceptors instead of oxygen (Willet 1983). There is a sequential reduction of elements (as predicted by thermodynamics), with oxygen being the first element to be reduced, followed by the reduction of nitrate to nitrite and ultimately N_2O or N_2 after the soil becomes anaerobic at approximately 250 mv. Nitrate reduction begins only after the oxygen concentration has dropped to a very low level (Mortimer 1941). Nitrogen transformation (mineralisation, nitrification-denitrification, immobilisation, volatisation and biological fixation) in submerged soils are of significance predominantly to agricultural production. With the decreasing redox potential of 225 mv, manganese is transformed to manganous compounds; iron is transformed from ferric to ferrous at about 120 mv (dissolved oxygen must first be reduced before the reduction of Iron III can proceed); sulfates reduced to sulfides at –75 to –150 mv in conjunction with a group of obligate anaerobic bacteria of the genus *Desulfovibrio*, using sulfate as an electron acceptor in respiration; and organic matter is transformed to low weight molecular weight organic compounds and methane gas at about –250 mv (Armstrong 1982; Mitsche and Gosselink 1993; Ponnamperuma 1972).

Measurements of redox potential provide a rapid and convenient indicator of the intensity of reduction in soils. Redox is not a measure of capacity; instead Eh is an intensity factor, and there is no neutral point as there is with pH. For a soil to have anaerobic conditions, the redox potential will be ≤175 mv at pH 7, and a measurement of –400 mV indicates a strongly reduced soil (Gambrell and Patrick 1978; USDA 2006). The resistance to change of Eh is expressed as poise, and is comparable to pH buffering. In well aerated, strongly oxidising systems, Eh is positive and high, and in reduced systems, Eh is negative and low. Chemical reactions that involve the exchange of electrons will be influenced by the redox potential (Ponnamperuma 1972).

An alternative to the use of Eh is the use of pe that is a measure of electron activity and is calculated as the negative logarithm of the electron activity whereby:

$pe = -\log(e) = Eh/2.303RTF^{-1}$, or $pe = Eh/0.0591$.

In strongly oxidised systems, the electron activity is low and pe is large and positive. The pe is small or negative in reducing systems (Ponnamperuma 1972). The pe is, however, a theoretical concept that cannot be directly measured in the field and therefore is of limited relevance for field studies (Vepraskas and Faulkner 2001).

The redox potential of a soil can be measured as the potential (E) between an inert platinum electrode and a standard reference electrode which are inserted at the depth of interest. When immersed in a system, the inert electrode takes on the electrical potential of that system. Electric potential in millivolts is measured relative to a hydrogen electrode or to a calomel reference electrode. The potential of the inert platinum electrode is usually measured by a pH meter and should be governed by its role as an electron acceptor or donor. The Eh can be obtained by adding the value of the potential of the standard calomel electrode relative to the potential of the standard hydrogen electrode to E (Armstrong 1982; Willet 1983). Redox measurements are of value for wetland soil classification largely because it has been possible to define the potentials at which a number of chemical changes in equilibria occur (Armstrong 1982).

The electrodes should be installed into the soil and remain there for up to one year to ensure a complete wet and dry cycle. Statistical analysis requires that an appropriate number of measurements are made across a soil horizon to ensure that soil variability is adequately accounted for, with at least five platinum electrodes installed for each depth for which the measurement is required (Vepraskas and Faulkner 2001).

Reduction is considered to occur with the reduction of Fe, not just because of the absence of oxygen (Vepraskas 1992).

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