Department of Environment
and Resource Management

Hydroecology of the lower Burdekin River alluvial aquifer and associated groundwater dependent ecosystems

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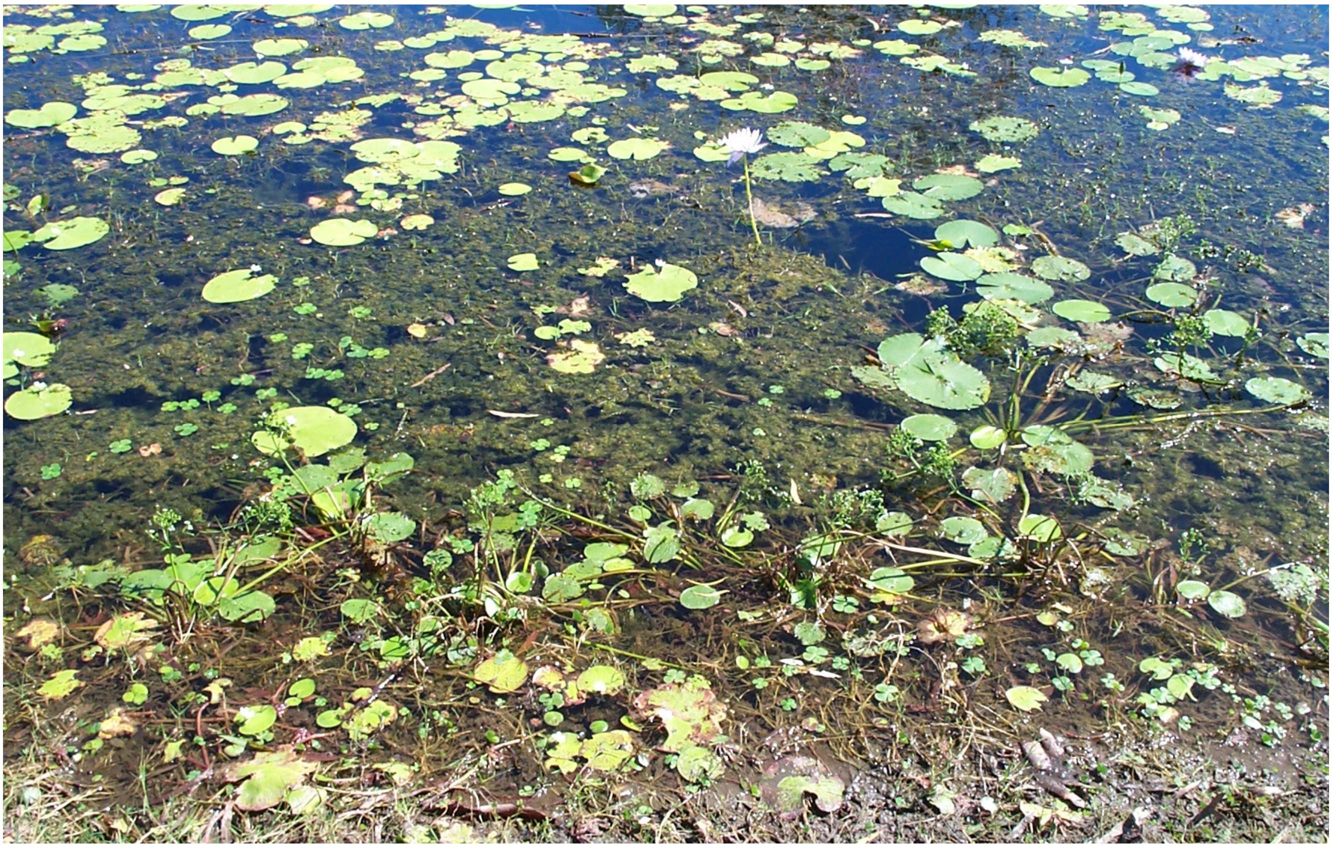
March 2012



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Preface

Through the Raising National Water Standards Program, the National Water Commission (NWC) provided funding to the former Queensland Department of Environment and Resource Management (DERM) to develop a groundwater modelling toolkit for the aquifers of the Lower Burdekin floodplain. The project is titled "Development of a Lower Burdekin Numerical Groundwater Flow and Solute Transport Model". The project was managed by the Queensland Hydrology Unit of the Environment and Resource Sciences section of the Department.

Prior to completion of the project, the Queensland Hydrology Unit became part of the newly formed Department of Science, Information Technology, Innovation and the Arts (DSITIA). Where relevant, all previous references to DERM have been changed to DSITIA.

This report is part of a series of eleven technical reports produced for the project. The overarching title of all departmentally-produced reports is "Development of a hydrological modelling toolkit to support sustainable development of the Lower Burdekin groundwater system." The full list of reports produced for this project are:

1. Review of modelling methods
2. Conceptualisation of the Lower Burdekin aquifer
3. Groundwater flow modelling of the Lower Burdekin aquifer
4. Instructional solute transport model of the Lower Burdekin aquifer
5. A re-evaluation of groundwater discharge from the Burdekin floodplain aquifers using geochemical tracers
6. Quantification of evapotranspiration in a groundwater dependent ecosystem
7. Geochemical assessment and reactive transport modelling of nitrogen dynamics in the Lower Burdekin coastal plain aquifer
8. Predictive uncertainty of the Lower Burdekin groundwater flow model
9. MODFLOW local grid refinement for the Lower Burdekin aquifer
10. Hydroecology of the Lower Burdekin River alluvial aquifer and associated groundwater dependent ecosystems
11. Pesticides in groundwater in the Lower Burdekin floodplain

All reports were produced by DSITIA, with the exception of:

- Report #5 which was authored by the National Centre for Groundwater Research and Training, Flinders University, Adelaide; and
- Reports #10 and #11 which were completed in March 2012 as DERM reports.

Contents

List of tables

List of figures

Executive summary

The Burdekin River in north-east Queensland is significant for its high discharge to the Great Barrier Reef lagoon and the high value aquatic ecosystems and agricultural activity it supports. Agriculture on the Lower Burdekin Floodplain is primarily founded on irrigation, which has given rise to a complex system of water management practices and infrastructure. This includes active recharge of the large alluvial aquifer that underlies much of the floodplain to prevent saltwater intrusion and supply water for crop irrigation. The current Water Resource Plan (WRP) for the Burdekin provides the blueprint for water management in the catchment but doesn't directly address groundwater resources and their management. This report is one element of work funded through the National Water Commission aimed at providing information for an amendment to the WRP to address this gap.

A range of ecosystems that are likely to be groundwater dependent to varying degrees are present in the Lower Burdekin floodplain. These include phreatophytic ecosystems, subterranean ecosystems and systems dependent on the surface expression of groundwater (including river base flows, groundwater dependent wetlands, and estuarine/ near shore marine systems). In a national context, most attention on groundwater and groundwater dependent ecosystems (GDEs) has been on water table draw down and the need to define the ecological water requirements of GDEs. In the Burdekin, however, delivery of irrigation water and aquifer management practices have led to rising water tables.

We investigated how depth to water table had changed since the construction of the Burdekin Falls Dam (and concomitant large-scale flow alteration). A range of ecologically relevant metrics indicated water tables had risen overall and become less variable in their level. Of particular ecological significance was the increased proportion of time water tables were now at levels that would result in seasonally dry wetlands remaining permanently inundated, and potentially also causing waterlogging of phreatophytic vegetation. Increased rate of change in water table level was also evident in a high proportion of the bores assessed. The implications of this for GDEs are less clear but rapidly changing water table levels may be significant for subterranean ecosystems.

We also investigated alterations to the hydrological regime of Barratta Creek. We focused on Barratta Creek because while the current WRP specifically states that there should be no further impacts on its natural flows, increased flow magnitudes have been reported elsewhere since the completion of the Burdekin Falls Dam. Our analysis of flow records from 1975 to present indicate median discharge has increased (particularly in the dry season) and maximum monthly discharges have generally increased, sometimes substantially. Barratta Creek also ceases to flow less frequently since dam completion. These results indicate that Barratta Creek has changed from a creek that dried out regularly to one that flows most of the time, with high flows occurring at less predictable times of year. Our synthesis of existing knowledge into conceptual models illustrates interplay between permanent inundation, deterioration in water quality, aquatic weed infestation and reduced habitat quality for aquatic flora and fauna in river baseflow GDEs like Barratta Creek.

Data collected from wetlands included current macroinvertebrate fauna and the fossil record of diatoms. Diatom and sediment analysis supported the description of changes to wetlands in the conceptual models. A groundwater dependent wetland on the floodplain with minimal hydrological change (Swans Lagoon) has the same dominant diatom species as it did 1300 years ago, indicating background conditions over that time period have been relatively stable. However, another wetland exposed to hydrological alteration (Labatt Lagoon) was found to have experienced significant ecological change with the current diatom community being unique in the fossil record. The current community is dominated by epiphytic diatom species that are indicative of wetlands dominated by macrophytes. These results indicate that a shift to permanent inundation and the associated dense macrophyte cover (including alien species) have resulted in an ecosystem different to anything that has existed in the last 1300 years.

There weren't clear differences in macroinvertebrate communities found in wetlands with different levels of flow regime modification, although unmodified sites tended to have higher overall richness and a higher abundance of taxa that are sensitive to impact. Our ambiguous results may reflect limitations in study design and/or that macroinvertebrate communities are reasonably robust to some change in flow regime if water quality is not severely degraded and habitats are sufficient for feeding and reproduction.

This study was significant in that we collected some of the first information about the subterranean ecosystems of the lower Burdekin, namely the bacteria and stygofauna living within the aquifer. Bacterial communities were indicative of contaminated or fouled organic-rich groundwater. There was evidence of widespread, dynamic surface and groundwater interaction as cyanobacteria that require light for energy and CO_2 as a carbon source were present in all groundwater samples. However, the co-existence and ubiquity of both anaerobic and aerobic bacteria suggests the potential existence of micro-niches or physicochemical gradients such as redox zonation within the aquifers. Stygofauna were found in 23% of bores sampled which is lower than studies of other Queensland aquifers using similar survey techniques. Higher order taxonomic diversity was low with only two groups of crustacea (copepods and syncarids) found. However, genetic analyses of one family (syncaridae) indicated high diversity within that family both at the species and population scales.

No new data were collected regarding phreatophytic vegetation and an extensive review of current knowledge was not undertaken. Phreatophytic vegetation in the lower Burdekin floodplain was recognised as being under a range of pressures including clearing for agriculture, fire, weed invasion and herbicide spray drift. However, rising water tables mean the root zone of deep rooted vegetation is saturated more frequently and this may have direct impacts on these ecosystems.

Five assets were recommended for investigation and inclusion in the amended Burdekin WRP. Assets were identified that have documented responses to hydrological alteration in groundwater that are likely to negatively impact upon their ecological value. All assets were in ecosystems dependent on the surface expression of groundwater. They included two guilds of wetland plants, native fish communities, phreatophytic vegetation and specific individual wetlands. Subterranean ecosystems were not recommended as assets because of our lack of knowledge about their response to changes in groundwater hydrology and their ecological values.

A number of knowledge gaps exist in relation to the hydrological requirements of the ecological assets. Another major gap is the current lack of quantification of the relative contributions of different water sources to wetlands on the floodplain. This makes it difficult to determine whether direct groundwater management is required or surface water management. This gap can start to be addressed with the groundwater model being developed as part of the broader project. Better understanding of the subterranean ecosystems is also essential to determine whether they should be included in future water planning.

Given the range of pressures to which groundwater dependent ecosystems on the LBF are subject to, attempting to return them to their natural state through groundwater management is not recommended. As this project has shown, GDEs have been modified from natural to varying degrees and restoration to natural may no longer be feasible. It is important to realise the inter-relationships between groundwater management, other stressors and ecosystem values are complex and that changing groundwater hydrology could have detrimental effects to some values. Care needs to be taken that cause-effect relationships are well understood before current management is modified.

1 Introduction

1.1 Project scope and objectives

The Queensland Department of Environment and Resource Management (DERM) received funding from the National Water Commission (NWC) to develop an integrated package of modelling and analysis tools to support groundwater management in the lower Burdekin; i.e., a ‘Hydrological Toolkit’. The intent of the project was to contribute foundational biophysical knowledge of the lower Burdekin groundwater system to augment current water resource management approaches in alignment with the National Water Initiative (NWI) and the Queensland *Water Act 2000*. The Toolkit can be used to inform an amendment to the Water Resource Plan that assesses the sustainable groundwater extraction regime for the natural system, and the potential impacts on groundwater-dependent ecosystems.

Two major sub-projects were funded to contribute to the Hydrological toolkit. The hydrology sub-project aimed to construct, calibrate, test and report unsaturated zone groundwater flow and solute transport models for the lower Burdekin (McMahon et al. 2012). The second sub-project aimed to assess and report responses by ecosystems to modified groundwater and associated surface water hydrology, and use these assessments to identify hydrology-ecology relationships for potential application in future modelling of water resource management scenarios.

This report presents the context, analyses and findings of the ecology sub-project; i.e. hydroecology of the lower Burdekin groundwater system. The objectives of this project were to:

12. Characterise hydrological alteration to the lower Burdekin groundwater system;
13. Assess and report responses by ecosystems to altered groundwater hydrology;
14. Distil several key hydrology-ecology relationships that have potential for application in future modelling of water resource development scenarios;
15. Present recommendations for management of groundwater resources in the lower Burdekin; and
16. Present recommendations for future groundwater hydroecology research in the lower Burdekin and in Queensland generally.

1.2 National and Queensland groundwater policy

A number of national and Queensland policy instruments exist with provision for the sustainable management and protection of groundwater resources. The Council of Australian Governments (COAG) made the first Intergovernmental Agreement on the Environment (IGAE) in 1992. This agreement provided a policy instrument for a nationally consistent and cooperative approach towards environmental management in Australia. With respect to water resource management, the IGAE referred to the National Water Quality Management Strategy (NWQMS) which was released in 1994 and provided a framework for protecting the nation’s water resources by improving water quality and reducing pollution. In 1995 the NWQMS’s Guidelines for Groundwater Protection in Australia was released, and in keeping with the over-arching policy directive of the NWQMS, focused on protection of groundwater resources from contamination by pollutants. These guidelines are currently being reviewed to include the water quality needs of groundwater dependent ecosystems and their fauna.

The need for better management of water quantity as well as quality was recognised at the 2004 COAG meeting where the Intergovernmental Agreement on a National Water Initiative (NWI) was signed. The overall objective of the National Water Initiative is "to achieve a nationally compatible market, regulatory and planning based system of managing surface and groundwater resources for rural and urban use that optimises economic, social and environmental outcomes". One of the specific objectives is "recognition of the connectivity between surface and groundwater resources and connected systems managed as a single resource". Also clauses 25 (ii) and 79 (i) (f) refer more specifically to protection of groundwater dependent ecosystems.

The Water Act 2000 is the legislative instrument for implementing the NWI in Queensland, and sets out the need and processes for water resource planning. While inclusion of groundwater in water resource plans is not compulsory water resource plans are progressively being amended to include groundwater and there is also a plan wholly focused on ground water with the Great Artesian Basin WRP.

The Environmental Protection (Water) Policy 2009 is also relevant in relation to Queensland waters. It seeks to achieve the object of the Environmental Protection Act 1994 - to protect Queensland's waters while allowing for development that is ecologically sustainable. Queensland waters include water in rivers, streams, wetlands, lakes, aquifers, estuaries and coastal areas. This purpose is achieved within a framework that includes identifying environmental values for aquatic ecosystems and for human uses and determining water quality guidelines (WQGs) and water quality objectives (WQOs) to enhance or protect the environmental values. Identification of environmental values and water quality objectives are due for the Burdekin in late 2012.

1.3 Overview of groundwater hydroecology

1.3.1 Definition of groundwater

For the purposes of this report, groundwater is defined as water within the saturated zone of the regolith and the associated capillary fringe, but does not include soil water held under tension in soil pores within the vadose (unsaturated) zone (Eamus et al. 2006; Fig. 1). Groundwater may be physico-chemically different from overlaying surface waters, for example groundwater in the Burdekin has naturally high levels of salt, sodium and hardness (EPA 2007). Exchanges between groundwater and surface water (i.e. recharge of the aquifer from surface water systems, and discharge of groundwater from the aquifer to surface water systems) are key hydrological processes of groundwater systems. Groundwater also discharges through transpiration and by diffuse discharge from shallow groundwater (i.e. not associated with surface water).

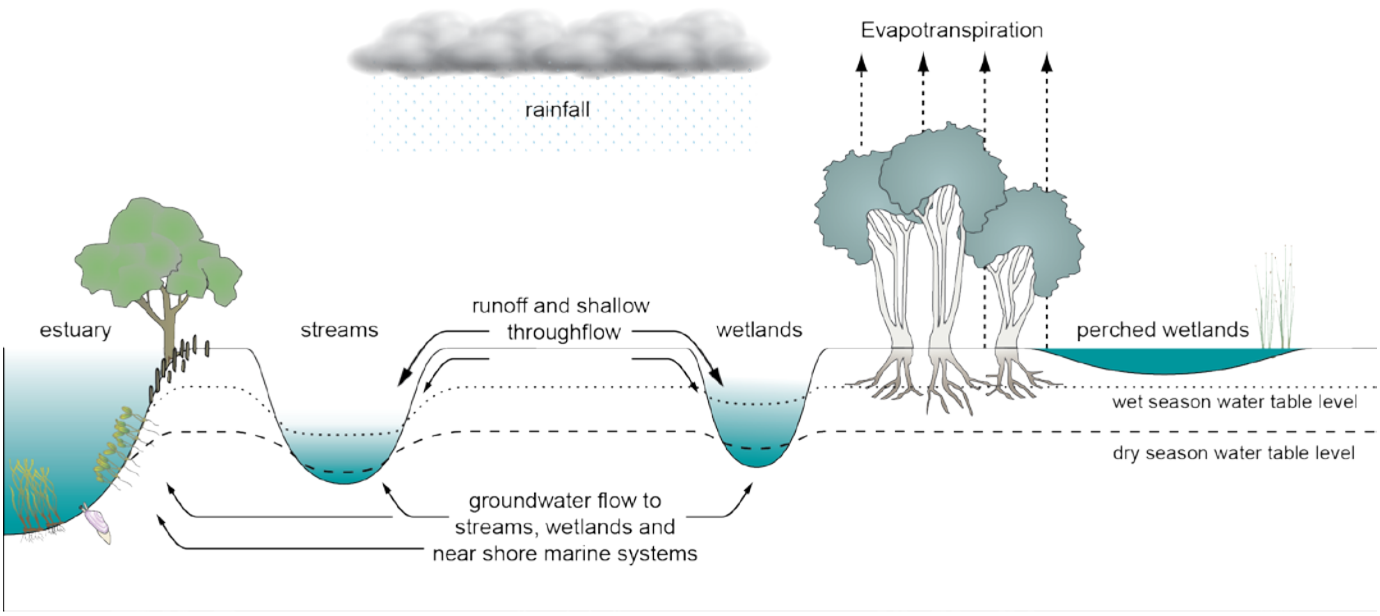


Figure 1 Schematic representation of the major hydrological features of an alluvial groundwater system. Source: G. McGregor.

1.3.2 Definition of hydroecology

Hydroecology is an interdisciplinary science that seeks to establish a bridge between ecology and hydrology (Manfreda et al. 2010). In establishing this bridge, the discipline seeks to generate understanding of how variability in hydrological processes shapes biological patterns and processes in freshwater-dependent ecosystems. The biotic composition, structure, and function of aquatic, wetland and riparian ecosystems depend largely on the hydrologic regime (Gorman and Karr 1978; Junk et al. 1989). In surface waters natural flow regimes sustain high biodiversity and enhance the resistance and resilience of aquatic ecosystems to disturbance (Poff et al. 1997; Bunn and Arthington 2002).

1.3.3 Groundwater dependent ecosystems

Groundwater dependent ecosystems (GDEs) are "terrestrial and water-based ecosystems (both saline and fresh) that rely on groundwater for some or all of their water requirements" (p.1. SKM 2011). Eamus et al. (2006) suggested three broad categories of GDE: a) subterranean aquatic ecosystems, b) ecosystems dependent on surface expression of groundwater, and c) phreatophytic ecosystems (phreatophytes being plants that obtain at least some water from groundwater - in Tomlinson 2011). This classification identifies GDEs with a common groundwater source, the advantage being it helps identify parallels and contrasts in the behaviour of functionally-similar GDEs (Tomlinson and Boulton 2008).

The particular level of dependency a GDE has on groundwater varies (Hatton and Evans 1998). Some ecosystems have complete year-round dependency (such as mound springs), others seasonal dependency (such as permanent river base-flow systems during the dry season) or episodic dependency (such as some terrestrial vegetation during extended droughts). Ecosystems that facultatively utilise groundwater (e.g. shallow-rooted terrestrial vegetation), still use groundwater but dependence may be questioned. Understanding the level and mechanism of groundwater dependency is important for the effective management of GDEs (Boulton and Hancock 2006).

a) Subterranean GDEs

A subterranean GDE (SGDE) is ‘an ecosystem occurring below the surface of the ground that would be significantly altered by a change in the chemistry, volume and/or temporal distribution of its groundwater supply’ (Tomlinson and Boulton 2008, p. 4). All subterranean waters are GDEs (Eamus et al. 2006) although not all subterranean species are groundwater dependent (Tomlinson and Boulton 2008). SGDEs include the great variety of aquifer systems as well as several ‘ecotones’ (transition zones between adjacent patches) that lie at the interface of groundwater and surface water environments.

The **hyporheic zone** is a fluctuating zone between surface streams and underlying groundwater where ecological processes are strongly influenced by the mixing and movement of surface and groundwater (Boulton et al. 1998; Fig. 2). **Parafluvial zones** lie alongside streams in those parts of the active channel, without surface water and can interact with subsurface water of the riparian zone (Boulton et al. 1998).

Subterranean aquatic fauna can be collectively referred to as **stygofauna**. Terminology varies, but fauna in the hyphoreic zone can also be referred to as the hyphoreos and distinctions can be made between types of fauna depending on their affinity to groundwater (Tomlinson and Boulton 2008). Stygofauna predominantly consist of different types of invertebrate such as crustaceans, worms and insects although there are also several records of specialised fish species (Tomlinson and Boulton 2008). Groundwater may also support a diversity of microbial life which together with stygofauna carry out key ecological functions such as maintenance of interstitial voids, the alteration of redox gradients, and the promotion of biofilm activity (NRW 2007).

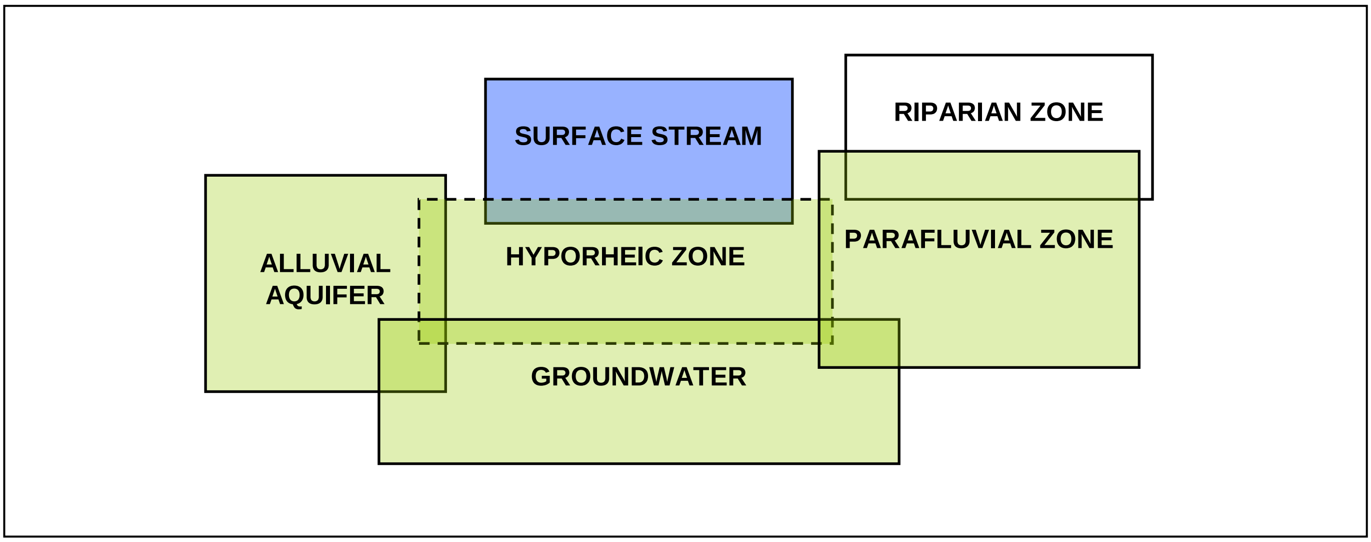


Figure 2. Simplified schematic diagram of subterranean groundwater-dependent ecosystems and hydrological compartments that can interact with them (modified from Boulton et al. 2008)

b) Ecosystems dependent on surface expression of groundwater

Ecosystems dependent on the surface expression of groundwater include river-base flow systems, springs, wetlands and estuarine and near-shore marine ecosystems (Hatton and Evans 1998; SKM 2001; Eamus et al. 2006). This range of ecosystems is similar to those defined as "wetlands" under the Queensland wetlands mapping and classification program - "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 6m" including three necessary attributes (EPA 2005).

River base-flow systems comprise lotic (flowing) aquatic and riparian ecosystems that rely wholly or partially on groundwater for their maintenance, function and integrity (Boulton and Hancock 2006). This means that most streams and rivers are fully or partially groundwater dependent. Most base flow derives from groundwater that seeps in from the channel margins and floodplains via the parafluvial zone or enters from below via the hyporheic zone.

Hatton and Evans (1998) considered that **wetlands** (the palustrine and lacustrine ecological systems identified in the Queensland schema - EPA 2005) provided the most extensive and diverse set of potentially groundwater-dependent ecosystems in Australia. It is well recognised that water regime is a critical determinant of the habitat diversity, communities of organisms, water and sediment chemistry of wetlands (Brock et al. 1999). While relating species distributions to water table behaviour is complex (Wheeler 1999) some broad processes are known. For example, increased inundation periods can result in soil waterlogging and de-oxygenation which can provide conditions suitable for invasive plants (Roberts et al. 2000).

Estuarine and near shore marine systems include coastal mangroves, salt marshes, coastal lakes and sea grass beds (SKM 2001). "Wonky holes" or submarine freshwater springs on the seabed are also a feature of the Great Barrier Reef. Groundwater flux will strongly influence dependency of coastal and estuarine ecosystems as it determines the level of seawater dilution (SKM 2001). These are likely biologically important areas due to the higher nutrient content characteristic of groundwater relative to sea water (Steglitz 2005).

c) Phreatophytic ecosystems

Finally, **phreatophytic ecosystems** are terrestrial GDEs that rely on the presence of groundwater. While these ecosystems are defined by their vegetation communities they also include associated faunal, microbial and fungal populations (Eamus et al. 2006; Clifton et al. 2007). The depth of the vadose zone determines the degree to which terrestrial vegetation is groundwater dependent. The vadose zone is the unsaturated zone above the water table in which soil water is held under tension in soil pores but only partly fills the pores (Tomlinson and Boulton 2008). Shallow vadose zones (i.e. < 10 m) are regarded as allowing deep-rooted vegetation to be connected with groundwater (Eamus et al. 2006).

2 Overview of the lower Burdekin groundwater system

2.1 Geographical setting

The Burdekin River is located on the north-eastern coast of Queensland, approximately 80 kilometres south of the city of Townsville. It drains an area of 129 500 km^2 and has the largest mean annual runoff of any river on the east coast of Queensland and experiences extreme variability that is both highly seasonal and erratic (Fielding & Alexander 1996). The study area in the Lower Burdekin floodplain (LBF) covers approximately 2500 km^2 (Fig. 3) and includes major water courses such as the Burdekin River, the Haughton River, Barratta Creek and their tributaries including Oaky Creek, Lagoon Creek, Woodhouse Creek, Clay Creek, Major Creek, Plantation Creek, Sheepstation Creek and Ironbark Creek (KBR 2002). In addition many minor estuarine channels exist within the tidal zones.

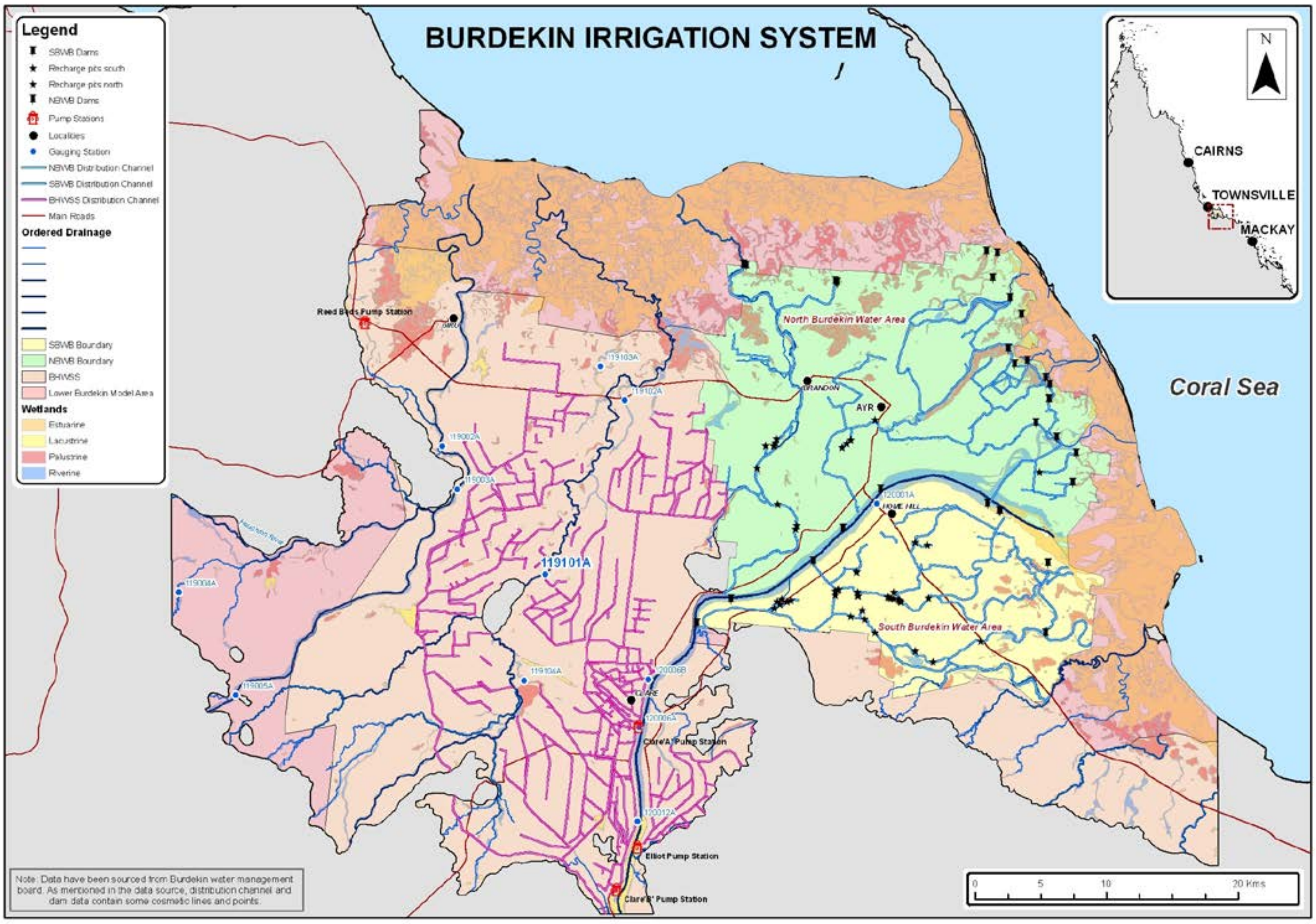


Figure 3 Map of the lower Burdekin floodplain showing major irrigation infrastructure.

Recent mapping of wetlands in the lower Burdekin, (EPA 2005) shows that while the majority of the wetlands in the Lower Burdekin floodplain are Estuarine wetlands near the coastline, all of the five major wetland systems are present. These systems are Marine, Estuarine, Riverine, Lacustrine and Palustrine (based on Cowardin et al. 1979). Marine and Estuarine systems are typically affected by tidal salinity and the other wetland systems are not. Riverine, Lacustrine, and Palustrine wetlands are the wetland systems that are associated with rivers, lakes (and topographic depressions or damned river channels), and vegetated non-tidal swamps, respectively. In the Burdekin wetlands include shallow non-permanent emergent sedge swamps, Melaleuca swamps, deep water lacustrine lagoons and a complex of coastal and estuarine mangrove saltpan habitats (Tait and Perna 2001).

From the point where the Burdekin River enters the floodplain below Blue Valley weir there is one contiguous aquifer (see McMahon et al. 2012). The alluvial floodplain is a complex layering of unconsolidated sediments of Quaternary origin with limited lateral continuity over an igneous basement (Petheram et al. 2008; Hopely 1970). The aquifers within the deltaic sequence extend from the surface to depths of > 60 m. Groundwater levels vary from < 3 m to 10 m below ground level, depending on seasonal conditions. There is a high degree of hydraulic connectivity between the aquifers and surface water features so that rainfall and surface water flows readily recharge the aquifers (NRMW 2006).

There are 5 main towns located within the study area: Ayr, Home Hill, Clare, Brandon and Giru and water use supports a range of industry. The lower Burdekin is largely managed for irrigated sugarcane and mixed horticulture (Perna 2003). Productivity of sugar cane in the Burdekin region is higher than any other sugar-growing region in Queensland—all sugar cane in the region has supplementary irrigation to ensure consistent and high yields.

2.2 Ecological values associated with groundwater

The Water Act provides a statutory framework for advancing sustainable management of water resources in Queensland. The "principles of ecologically sustainable development" are supported by the Act and include public involvement to determine cultural, economic and social values as well as consideration of environmental values (s10, s11 and s47, Water Act 2000). A comprehensive community consultation to identify ecological values associated with groundwater was beyond the scope of this study. Instead groundwater related ecological values of the LBF were identified using a desktop analysis of the primary and grey literature.

2.2.1 Significant wetlands

A number of wetlands within the lower Burdekin have been nationally recognised as of significant value in the Directory of Important Wetlands (Environment Australia 2001; Australian Wetlands database; WetlandInfo). These include the Burdekin - Townsville wetland aggregation, Burdekin delta aggregation, Bowling Green Bay, Barrattas Channels aggregation and Jerona aggregation. The wetlands form one of the largest concentrations of wetlands on the eastern seaboard (Tait and Perna 2001) and the Bowling Green Bay aggregation is internationally recognised under the RAMSAR convention. The Great Barrier Reef off the coast is also internationally recognised through World Heritage listing.

Burdekin floodplain wetlands are highly productive and provide nursery habitats that support commercial and recreational fisheries. The Burdekin supports both extensive commercial (inshore net fin fish and crab fishery) as well as a large recreational fishery with many professional guides fishing the region (Halliday et al. 2001). The wetlands also support significant breeding aggregations of regionally significant waterfowl and provide a major stop over point for migratory wading birds (Tait and Perna 2001).

The wetlands are also important to the traditional owners of the region. The Bindal people made an early request that the wetlands be spared the development seen in other floodplains of northern Queensland to preserve their productivity and environmental values (Tait and Perna 2001). Traditional owners are actively involved on natural resource management through the Gudjuda Reference group who participate in the Burdekin-Bowen Integrated Floodplain Management Advisory Committee and North Queensland Dry Tropics Board.

2.2.2 Ecosystems used to assess impacts arising from changes to groundwater hydrology

Ecosystems in the Lower Burdekin floodplain with potential dependence on groundwater were identified using a desktop analysis of the primary and grey literature. The draft Water Resource Plan (Brizga et al. 2006) stated that the main types of groundwater dependent ecosystems present in the catchment were phreatophytic ecosystems (specifically terrestrial vegetation) and ecosystems dependent on surface expression of groundwater (specifically river baseflow ecosystems and wetlands). It was acknowledged that subterranean aquatic ecosystems (in particular those associated with the aquifer) might be present but there is almost no pertinent information available (Brizga et al. 2006).

Phreatophytic ecosystems

Most of the study area has been cleared and planted with sugar cane, greatly reducing naturally occurring groundwater dependent vegetation cover. There are pockets of remnant terrestrial vegetation across the floodplain which would have some level of groundwater dependence. A large proportion of this vegetation is contained in the Barratta Creek catchment and has been recognized as important habitat (Tait and Veitch 2007), or within the riparian zone of other streams and rivers on the LBF. Remnant terrestrial vegetation has also been identified at Lilliesmere, Healy's and Reedy beds lagoons (Brizga et al. 2006).

In the current situation these GDEs would be in constant contact with the aquifer which may result in waterlogging, rather than being denied access to the aquifer which is the more common scenario in Australia. It is documented that increases in groundwater levels can cause waterlogging and kill phreatophytic vegetation (Naumburg et al. 2005). Remnant phreatophytic vegetation in the LBF is also subject to a complex array of other impacts. Within the Barratta Creek Catchment for example, phreatophytic ecosystems are under stress from changes in fire regimes, grazing pressure, woody weed invasion and exotic grasses that are pyrophytic and burn at very hot temperatures impacting growth and recruitment (Tait and Veitch 2007). Another impact may be spray drift of herbicides from agricultural activities and the presence of agrichemicals in the groundwater itself (Adams and Thurman 1991; Keating et al. 1996).

We have not surveyed phreatophytic ecosystems for this report due to the complex array of impacts they are exposed to and the time constraints under which we were working. This is not to say that they should not be considered as potential assets in the amendment of the Water Resource Plan. Some of the impacts and changes to these ecosystems, however, have been captured in the conceptual models (Section 2.4).

Ecosystems dependent on surface expression of groundwater

There are many ecosystems in the Burdekin that have been identified as having some degree of dependence on the surface expression of groundwater. These include but may not be limited to: near shore upwelling (wonky holes), river base flows, wetlands (groundwater lenses), and shallow estuarine wetlands. Wonky holes have been documented in Bowling Green Bay (Steglitz 2005). These areas are highly valued by recreational fisherman for holding high densities of prized fish such as Red Emperor and Paddle-Tail snapper, however the effects of increased nutrients and pesticides in groundwater on these habitats aren't fully understood.

Ecosystems reliant on river base flows were identified in Lilliesmere and Healy's lagoons (Brizga et al. 2006). Wetlands reliant on groundwater were identified by Brizga et al (2006) as the Burdekin River, Sheep Station and Plantation Creeks. Sections on these riverine wetlands that have permanent surface water are dependent on direct connection to groundwater.

Groundwater dependent wetlands have been the main focus of our assessments because many of them are highly valued (see section 2.3), and previous studies suggest they have been directly impacted by changes in groundwater hydrology (Perna and Burrows 2005; Kellet et al. 2005; Dillon et al. 2009). We focussed on Barratta Creek in particular because it has a specific ecological outcome included in the Water Resource Plan (2007): " *to ensure there are no further impacts on natural creek flows in the Barratta Creek system*". We also examined shallow estuarine wetlands (Fig. 4) which have completely changed to ponded freshwater areas as a consequence of hydrological modifications on the floodplain. Shallow non-permanent palustrine wetlands (Fig. 4) have also changed to either permanent wetlands or have been cut off from flow pathways by constructed channels and receive reduced flows. Lastly permanent riverine wetlands (Fig. 4) have now become static in water level and infested with alien plants and fish and have decreased water quality. Other GDEs are found in proximity to these wetlands and we have attempted to highlight changes to these as well.

Subterranean aquatic ecosystems

Brizga et al. (2006) recognized that the alluvial aquifer of the LBF would likely hold an array of biota however no survey was undertaken. Hydrological modelling undertaken as part of this project also suggests that the granite basement that underlies the sedimentary deposits is often weathered or fractured and can play a significant role in groundwater flow, storage, and quality (McMahon et al. 2012) which may have significance for subterranean ecosystems. Preliminary sampling of the aquifer was undertaken to try to characterise the subterranean ecosystems present in the Lower Burdekin Floodplain (section 4.2.1). Ultimately we have not been able to assess the impacts of groundwater hydrology change on these systems through a lack of data and knowledge. Their role in changing ecosystem function for wetlands has been included in the wetland conceptual models (section 2.5) primarily based on expert opinion elicited at an expert panel workshop (Appendix A).

2.3 Pressures on the lower Burdekin groundwater system

Groundwater resources in many parts of Australia are facing increasing pressure from consumptive uses for agricultural, mining, urban and commercial developments (SKM 2001). A range of threatening processes has been identified in relation to groundwater dependent ecosystems (including wetlands) in Australia. These include water resource development, agricultural land use and climate change (SKM 2001; Davis et al. 2007). In the lower Burdekin, water resource development is the predominant threatening process and also the one that can be addressed through Water Resource Planning. It is important to recognise the other threats, however, as they may constrain the achievement of the plan's ecological outcomes and may confound the detection of an ecological response to the plan's provisions (Tomlinson 2011).

Much of the effort to understand impacts to GDEs in Australia has focused on the implications of water table draw down given water is generally extracted from aquifers at rates greater than replenishment (Hatton et al. 1998). Within the LBF however excessive water infiltration is a major concern because the aquifer is used as a storage for irrigators across the delta (Perna and Burrows 2005; McNeil and Raymond 2011). The implication of super saturation of the shallow aquifer system is that it represents a significant threat to the GDEs of the region (McMahon et al. 2012). A number of related concerns include; agrichemical mobilization, sodium contamination, saltwater intrusion, poor water quality and habitat disturbance (Kellet et al. 2005).The rising water tables in some areas of the LBF (e.g. Mona Park) have caused salination though salt leaching and accumulation in response to irrigation practices (SKM 2009). Increased water levels can also cause anoxia in the root zone, decreased water level variation and increase water levels in surface water features (Dillon et al. 2009).

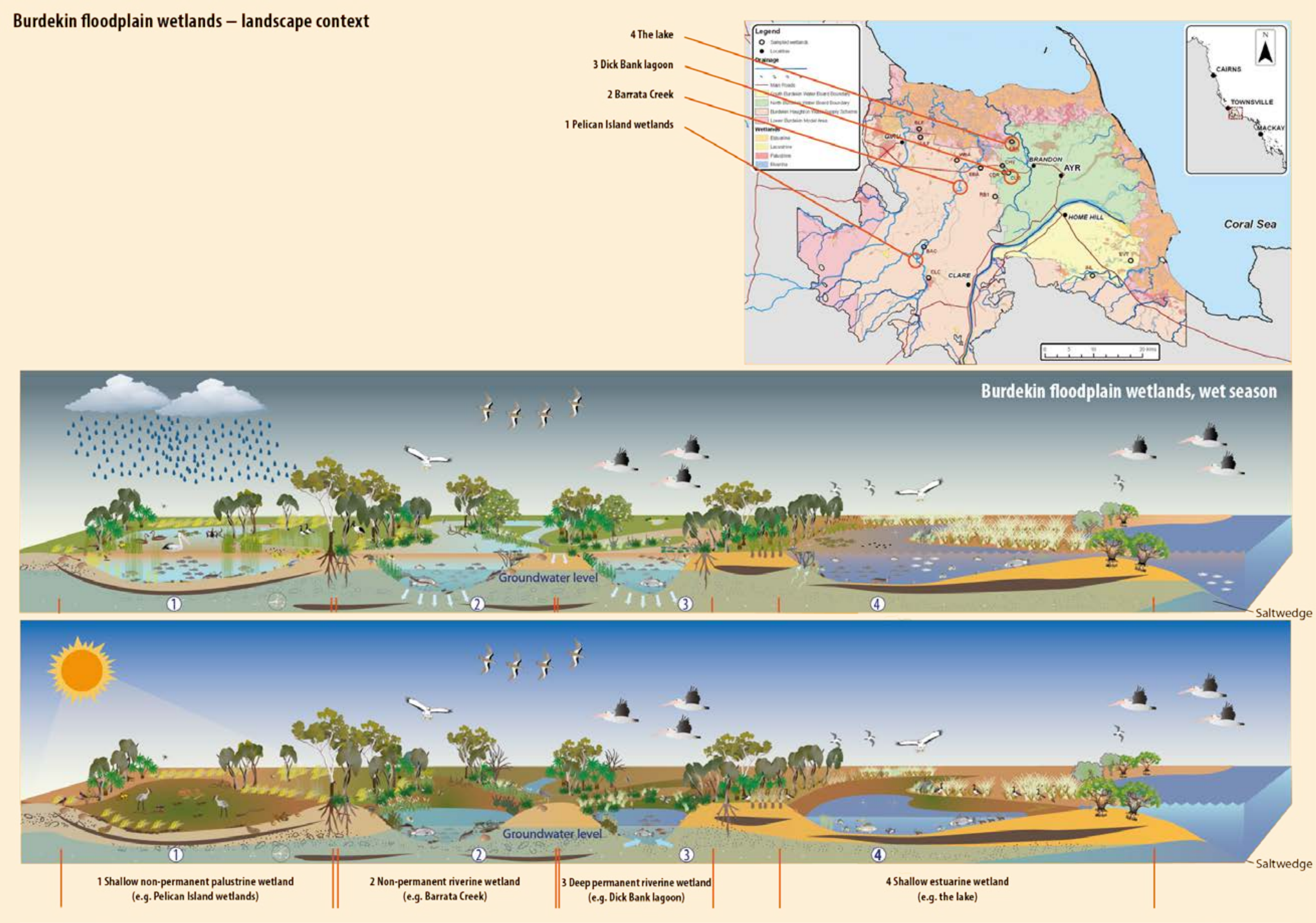


Figure 4 Panorama diagram of the lower Burdekin floodplain showing four key wetland types.

2.3.1 Water resource development

The extensive cropping in the lower Burdekin relies on irrigation because the region lacks sufficient consistent or predictable rainfall. Prior to the development of larger water retention basins such as Clare weir and Burdekin Falls Dam, water was opportunistically harvested to supply irrigation needs to a growing sugar industry. The North and South Burdekin water boards were established in the 1960's to harvest water and actively recharge the shallow aquifer (Tait and Perna 2001). This managed aquifer recharge scheme (Managed Aquifer Recharge MAR) exists over approximately 38,000 ha of the delta (Charlesworth and Bristow 2004 - Fig. 3). Water is extracted from various pump stations along the river downstream from the Gorge weir and pumped into remnant distribution streams such as Sheep Station and Saltwater Creeks or constructed channels in the BHWSS. In some cases, channels have been constructed to better connect the drainages for water distribution purposes (Fig. 3). In the larger lagoons, drop board systems have been installed to increase water levels and thus aquifer recharge (Fig. 5). Water is also pumped into sandy recharge pits to directly replenish the aquifer (Fig. 5). This system distributes water according to groundwater level not volume so the aquifer is recharged as needed to maintain extraction without too much draw down.

Groundwater replenishment allows for irrigation extraction and prevents seawater intrusion by maintaining a head of freshwater at the interface of fresh salt waters (Narayan et al. 2003). It also dilutes the aquifer preventing salt mobilization (Narayan et al. 2003). At the downstream end of these systems the water boards take advantage of the existing network of bund walls to pond water for the purpose of creating freshwater head.

Construction of the Clare weir and Burdekin Falls Dam (BFD) allowed perennial water harvesting in the delta area as well as expansion into the more marginal areas to the north and east of the river (Burdekin Haughton Water Supply Scheme BHWSS - Fig. 3). The BHWSS supplies irrigation water across approximately 50,000 ha (Fig. 3). It includes extensive surface water infrastructure such as water storages, pumping stations and irrigation water supply channels (Fig. 5). The pump stations divert water into distribution channels on the north bank of the river and then to customers by a system of constructed channels and balancing storages. This system does not include groundwater in its distribution or supply. This means that water lost to the aquifer via seepage from earthen channels and irrigation return flows, is not managed (Fig. 5). Over-extraction of groundwater was of concern prior to the completion of the BHWSS. Since then water tables have steadily risen (Petheram et al. 2008). There is only a small area within the BHWSS (Jardine, Mona Park, Northcote and Mulgrave) where groundwater extraction is permitted. Without extraction or dilution, salts can be mobilized in the saturated zone (Petheram et al. 2008). If groundwater levels are high enough they can also directly impact on crop production through waterlogging (Petheram et al. 2008, Charlesworth and Bristow 2004). Therefore, more extraction in this case may be useful in ameliorating or preventing problems.

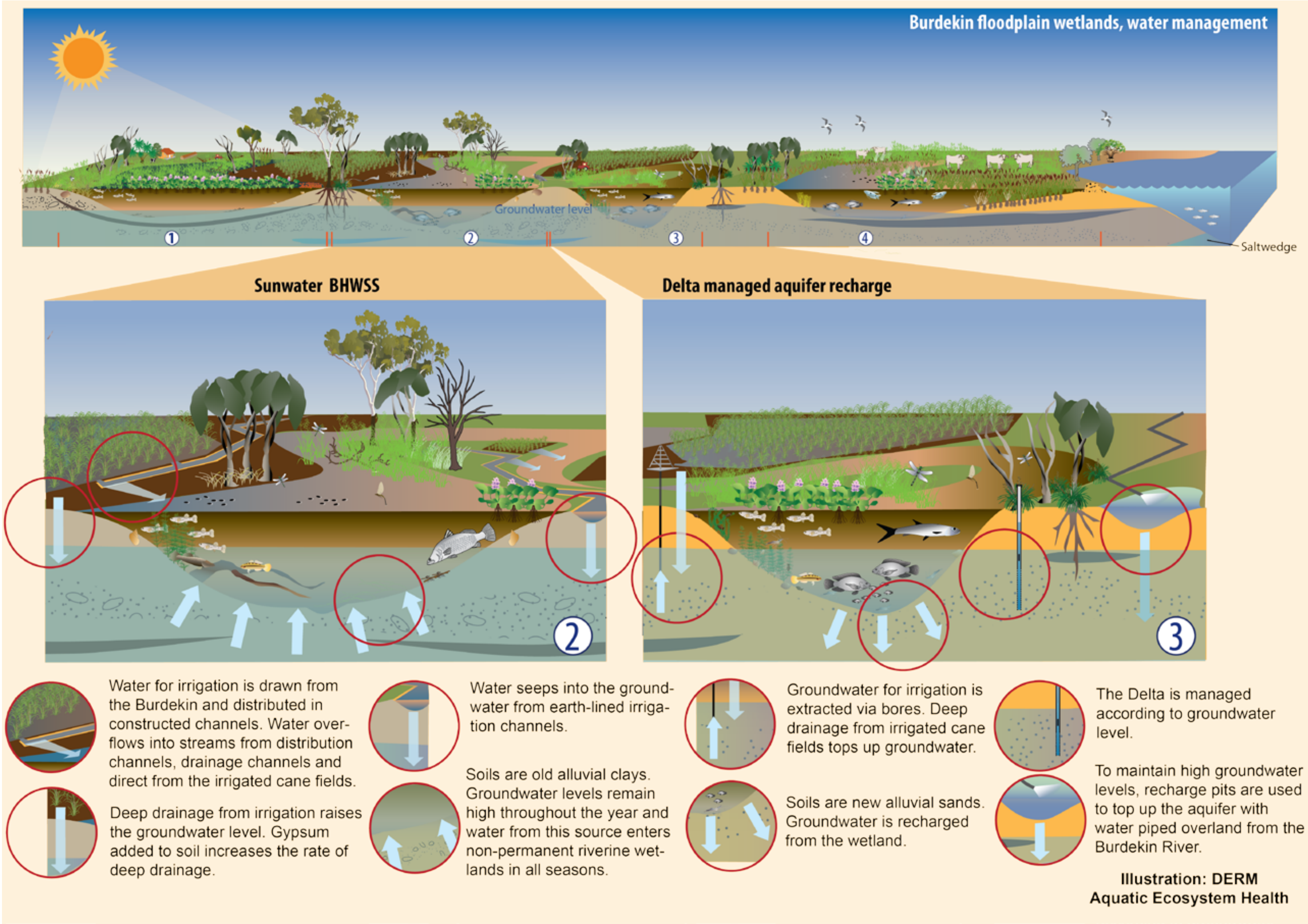


Figure 5 Conceptual model of water resource development in the lower Burdekin floodplain. See Appendix B for legend.

2.4 Conceptual models of hydroecological processes

Conceptual models are a visual depiction of complex ecological process, for natural systems, compiled by current expert knowledge and understanding derived from research. Complex scientific understanding can be presented in pictures, symbols and colours to more simply demonstrate these complex systems. Conceptual models can facilitate communication, integrate knowledge across disciplines, increase understanding of complex issues or natural systems, identify knowledge gaps, inform decision-making and planning process and facilitate participation of stakeholders.

We have produced conceptual models for a subset of wetland systems on the lower Burdekin floodplain to highlight how they have responded to hydrological changes related to irrigation. Change is illustrated by depicting how wetlands may have looked and functioned before European development of the floodplain and how they look and function under current water management regimes.

2.4.1 Conceptual models of groundwater hydroecology pre-development

These pre-development conceptual models are based on knowledge of remnant semi-natural wetlands on the LBF, a literature review, and knowledge of similar systems in other catchments (in particular the Fitzroy floodplain). All models also include feedback provided by an expert panel (Appendix A).

Shallow non-permanent palustrine wetlands (e.g. Pelican Island wetlands)

During the wet season streams flowed, either from local runoff, or in some cases from overflow from the Burdekin main channel. These flows filled shallow wetlands across the floodplain including shallow non-permanent palustrine wetlands. Water quality was influenced by flows, with turbid waters clearing after flows. Waters typically had low electrical conductivity and neutral pH. Dissolved oxygen was variable as plants proliferated in the early wet season producing oxygen during daylight hours, then as water levels dropped and organic material started to decompose, oxygen was consumed.

These wetlands were generally perched over impermeable clays above the water table and so received little input from groundwater (Fig. 6). They could be highly productive for many types of fauna and flora including aquatic plants that flourished during the wet season. The plant communities were characterized by species that required wet/dry cycling for their life histories, such as *Aponogeton, Nymphoides*, *Marsilea* and *Ottellia* (Fig. 6). Semi-aquatic or emergent plants established within the littoral areas and provided habitat for a variety of animals.

The wetlands likely provided habitat for aquatic invertebrates and may have been dominated by highly mobile winged insects including hemiptera, coleoptera and diptera that could move elsewhere in the dry. Fish utilised the wetlands for feeding and breeding while they were connected to more permanent waters (Fig. 6). Fish species likely included grazers such as freshwater herring, broadcast spawners such as spangled perch and generalist consumers such as fork-tailed catfish (Fig. 6). Decreasing water levels and concentrated fish populations would have provided major food source for piscivorous (fish-eating) birds.

As the dry season progressed, these wetlands become disconnected from other surface waters and in most years eventually dried out (Fig. 6). Habitats changed rapidly and with them plant and animal communities. Most of the fish returned to permanent water bodies while crustaceans, molluscs, and tubers and rhizomes of certain plants persisted in the mud (Fig. 6). It is assumed that stygofauna were present in the local aquifers of these wetlands. How the wet/dry cycle impacted stygofauna on the LBF is not understood. Groundwater dependent terrestrial vegetation that had roots within the soil saturation zone during the wet relied on capillary connection to groundwater supplies during the dry season. Aquatic plants and semi-aquatic emergent vegetation along the banks dried out and this triggered the flowering and seeding stages of many species. Grasses and sedges became dominant and supplied food for terrestrial animals such as black throated finches (Fig. 6).

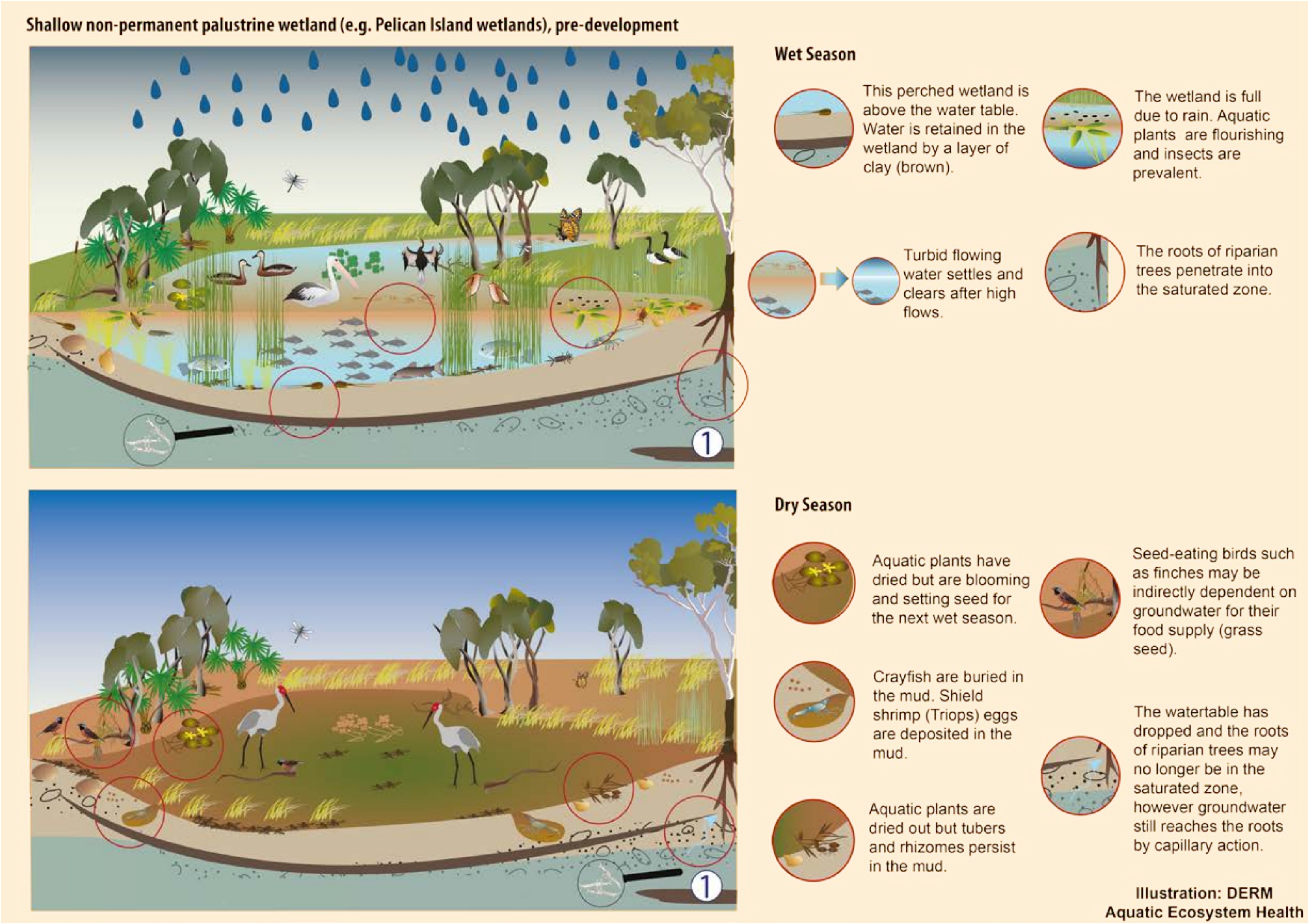


Figure 6 Conceptual model of pre-development wet and dry season conditions in shallow non-permanent palustrine wetlands. See Appendix B for legend.

Non-permanent riverine wetlands (e.g. Barratta Creek)

In the lower Burdekin floodplain non-permanent riverine wetlands were generally found on natural distributary streams. Distributary streams carry floodwaters away from the Burdekin River as well as their own catchment runoff. Here we are focusing on seasonally flowing distributary streams found on the older clays of the floodplain such as Barratta Creek. Streams such as Sheep Station Creek on the younger sandy alluvial soils of the floodplain are captured in the "deep permanent riverine wetlands" models.

Depending on the magnitude of the wet season, these streams flooded and may have recharged the aquifer in some reaches (Fig. 7). During the dry season, waters receded; creating a series of isolated deep water refuges (Fig.7). Groundwater levels also dropped during the dry season resulting in limited connectivity to the refugial waterholes, although the degree of groundwater connection was linked to location in the catchment and soil type. Differences in hydrology drove differences in water quality between these wetlands and the "deep permanent riverine wetlands" found on sandy alluvial soils. The non-permanent wetlands were driven by surface water inputs that created highly turbid flood waters in the wet season that delivered organic material that stained the waters dark by the end of the dry season. In contrast, the deep permanent wetlands that were more strongly influenced by groundwater inputs were clear in the dry season (Fig. 8).

Under wet season flows, full connectivity from the ocean to the upper reaches allowed fish (especially amphidromous species such as; mangrove jack, giant herring and barramundi) to migrate up and down the system in search of feeding and breeding habitats (Fig. 7). In the dry, fish were restricted to permanent refuge pools until the next flow events. Instream vegetation was dominated by submerged feathery species such as *Ceratophyllum* and *Potamogeton* (Fig. 7). These plants were prolific in the wet season where riparian shading was least as clear waters allowed deeper light penetration for instream plant growth. The same applied for emergent vegetation such as *Persicaria* and *Sporobolus* grass (Fig. 7).

Diverse macroinvertebrates communities were present and dominated by groups adapted to seasonal flow regimes. Stygofauna were also likely found in the underlying aquifer but again we have limited understanding of how these communities reacted to wet/dry cycles.

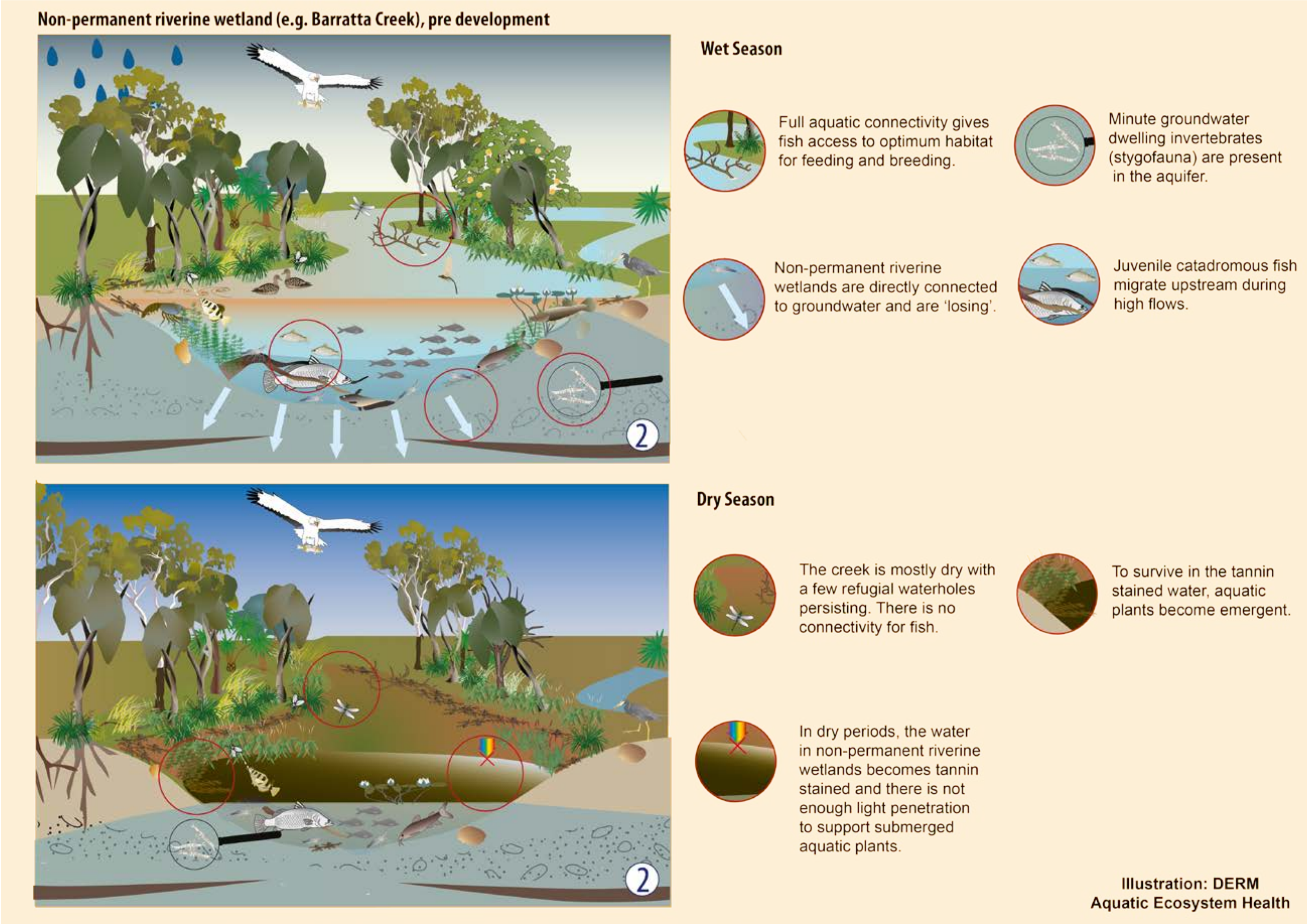


Figure 7 Conceptual model for pre-development wet and dry season conditions in non-permanent riverine wetlands. See Appendix B for legend.

Deep permanent riverine wetlands (e.g. Dick Bank Lagoon)

The distributary streams on the younger delta alluvial sands usually consisted of a series of permanent to semi-permanent deep water lagoons connected by streams. The duration and magnitude of the wet season determined the amount of water persisting through the dry season in these wetlands. In the wet season, the bed of the wetlands may have been in direct contact with the aquifer and losing water to it because of the head height in the wetlands (Fig. 8). The height of the water table influenced wetland depth over the dry season so they only completely dried out in the very driest years. During high flows water flowed into the aquifer through various pathways, including the stream bed and shallow sandy prior stream beds scattered across the floodplain (Fig. 8). In the dry season water coming into the wetlands from the shallow sandy aquifer was clear (in contrast to non-permanent riverine wetlands which received tannins and nutrients characteristic of the clay soils - Fig. 7).

Flood events scoured the lagoons and large amounts of organic material were removed from both the lagoon and stream itself, thereby reducing organic loads for the following dry season. In stream plants such as *Ceratophyllum*, *Potamogeton*, *Utricularia* and lily species were scoured out in high flows, growing back from remaining patches through the dry season. These plants were important in maintaining the oxygen balance in the water column and habitat for aquatic fauna. Emergent vegetation such as *Lomandra* and *Sporobolus* re-established after high flows and provided habitat and organic inputs to aquatic food webs.

Fish communities that were isolated in these waterbodies during the dry season may have migrated out of the lagoons in search of food and breeding habitats as streams started to flow in the wet season (Fig. 8). Diverse communities of macroinvertebrates responded to the various flow and environmental conditions utilising the diverse habitat provided by the native vegetation. It is assumed stygofauna were present in the underlying aquifer.

Shallow estuarine wetlands (e.g. The Lake)

These wetlands filled with freshwater during the wet season floods and water levels fluctuated with the tide. The head of freshwater both on the surface and in the aquifer pushed the salt wedge in the aquifer closer to the coast however there was still significant ingress of tidal waters on the surface. Throughout the year the water within these wetlands was brackish through the balance of groundwater and tidal influx.

The combination of groundwater input and tidal influx created a boom of microcrustacean production which supported abundant fish and waterfowl populations (Fig. 9). During the wet season the fish community was diverse and included amphidromous (fish that migrate between freshwater and the sea) and freshwater species (Fig. 9). Many juvenile fish were also present utilising the boom of productivity. Aquatic plants and emergent vegetation such as *Eleocharis* flourished creating habitat for both aquatic and terrestrial animals (Fig. 9).

As the dry season progressed, groundwater input dropped off and the surface water became more saline from spring high tides (Fig. 9). Depending on the magnitude of the preceding wet season, some wetlands may have dried completely while others were maintained through tidal ingress. As the head of freshwater decreased, the salt wedge moved upstream. The fish community changed to one dominated by salt tolerant species. Less salt tolerant plants died back and more salt tolerant plants flourished. Marine invertebrates such as penaeid prawns may have invaded during this time utilising this highly productive environment for rapid growth.

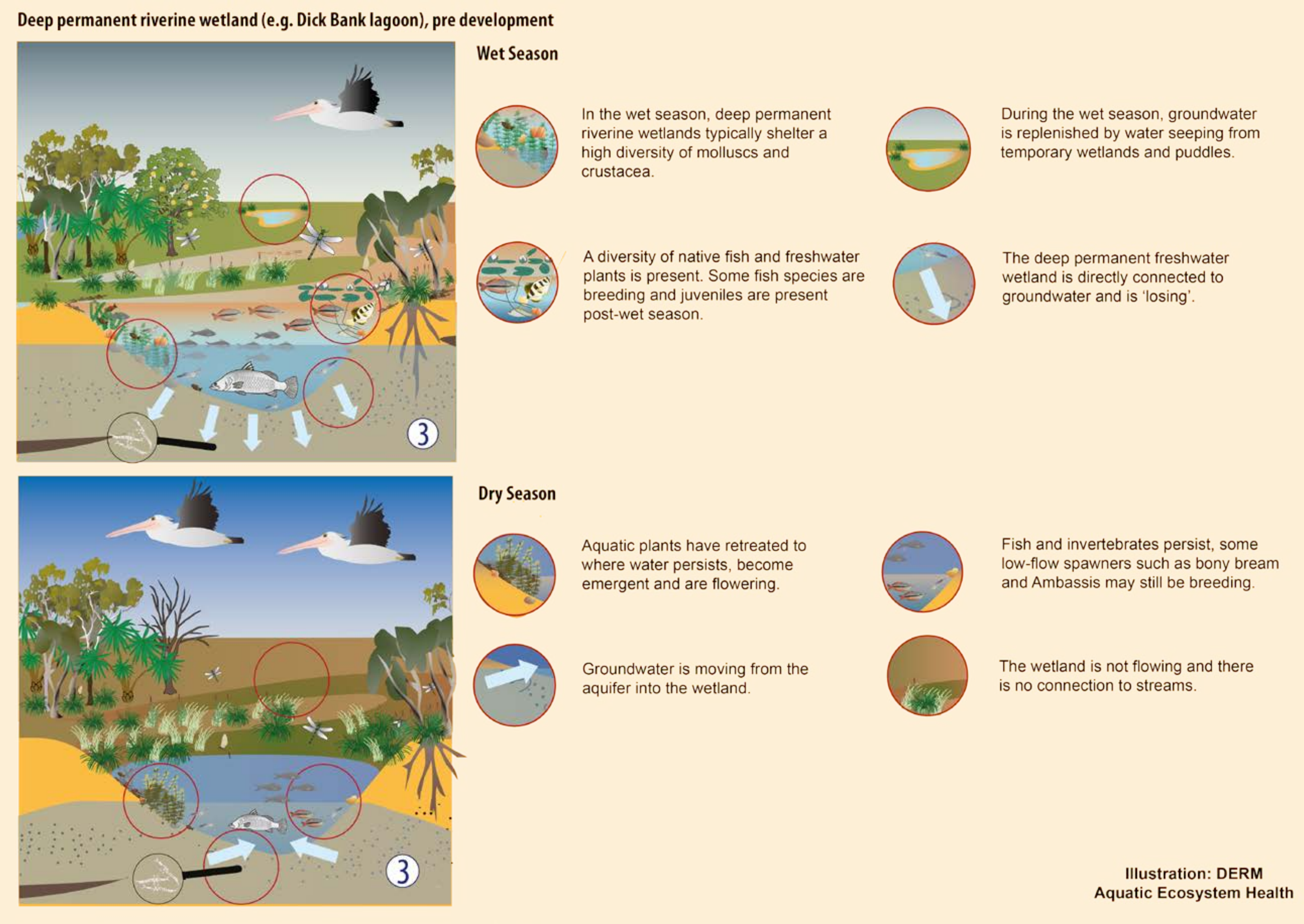


Figure 8 Conceptual model of pre-development wet and dry season conditions in deep permanent riverine wetlands. See Appendix B for legend.

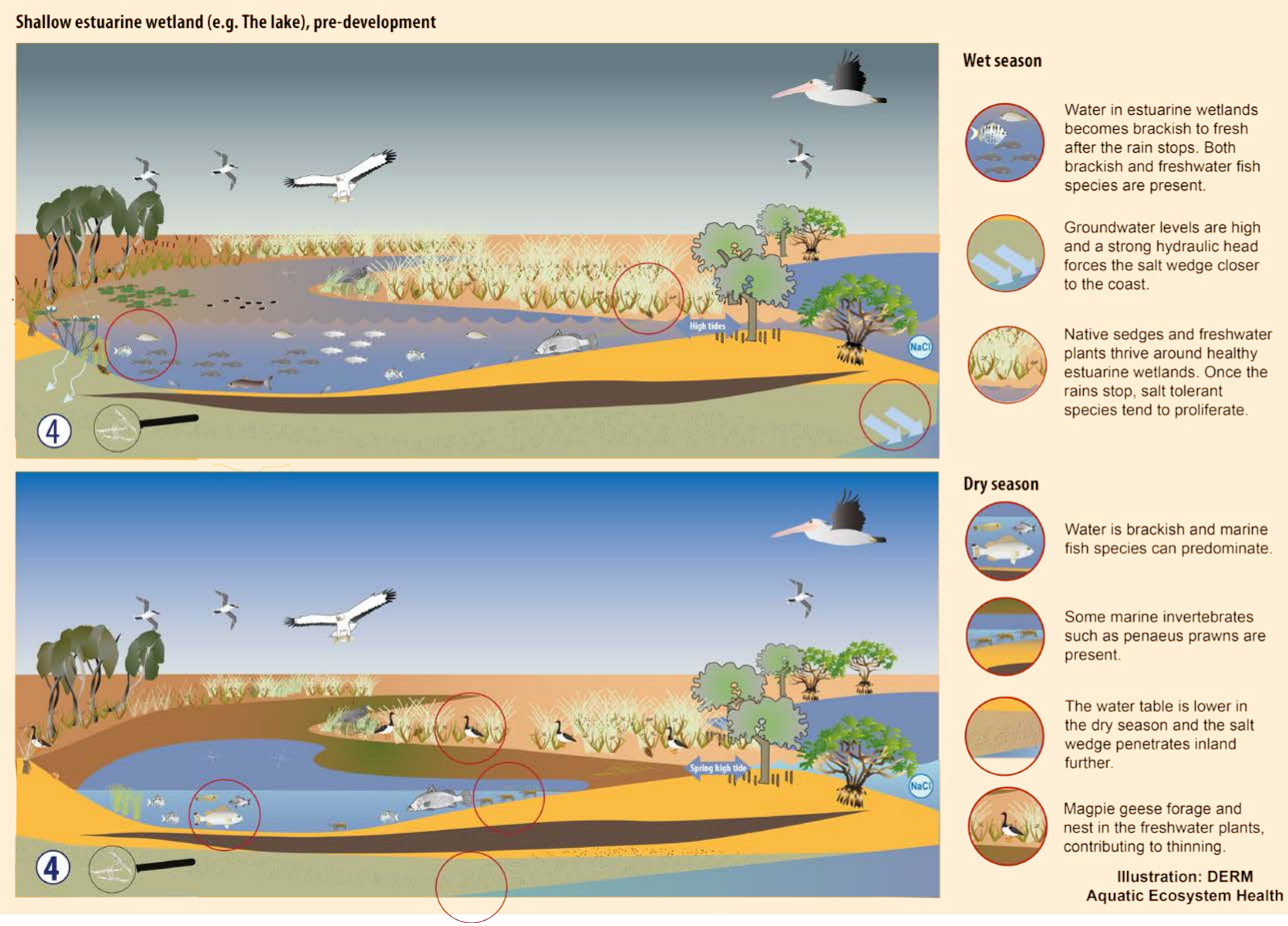


Figure 9 Conceptual model of pre-development wet and dry season conditions in shallow estuarine wetlands. See Appendix B for legend.

2.4.2 Conceptual models of current groundwater hydroecology

The following section refers primarily to impacts related to surface water supplementation rather than focussing solely on groundwater. This is because changes in groundwater are a direct result of surface water supplementation. Separating the two is not possible at the moment, but DERM are currently investigating the proportional contributions of groundwater and surface tailwater to the perennial flows of wetlands such as non-permanent riverine wetlands (B. Bennett pers comm).

Shallow non-permanent palustrine wetlands (current)

Shallow non-permanent palustrine wetlands have been greatly altered in hydrology and habitat in the post-development scenario (Fig. 10). Where there was an annual wet/dry cycle, there is now continuous inundation with seasonal high flows from wet season rains. In some areas the water table is higher than the base of the wetlands and provides continuous inputs of water with low dissolved oxygen levels (Fig. 10). Water levels are also maintained by tailwater (this may come from furrow irrigation and channel overflow) which may also be low in dissolved oxygen. However in some areas where constructed channels have bypassed wetlands the wetlands receive less water than in pre-development conditions (eg. Gilgai and Green swamps). In general the supplemental surface water flows have increased groundwater levels but it has yet to be determined the percentage contribution of each water source to the permanence of water in these wetlands.

Changes in hydrology and water quality have greatly reduced habitat quality for native aquatic fauna while benefiting alien plants and animals. Plants characteristic of non-permanent wetlands have been replaced by aquatic plants adapted to permanent inundation such as *Ceratophyllum*, or alien plants such as water hyacinth which prefer static water levels (Fig. 10). Static water levels have also caused waterlogging of riparian trees in some places which has benefited alien semi-aquatic grasses such as para grass and *Hymenachne* (Fig. 10) However other factors such as fire, weed invasion and herbicide spray drift may also be negatively impacting riparian vegetation communities.

"Black water" which is anoxic and nutrient rich can occur in some wetlands with consistently high water levels (Fig. 10). With the first high flows of the wet season this water can be flushed down streams causing ecosystem failure. This water also prevents upstream migration of amphidromous fish, delivers high nutrient loads into estuaries and the Great Barrier Reef lagoon and can cause large scale fish kills as it moves through the system. The fish communities remaining in these wetlands are dominated by alien species and a few natives that are tolerant of low oxygen conditions (Fig. 10). The macroinvertebrate community may be have lost species adapted to desiccation and sensitive to low oxygen levels. Stygofauna is present in at least some locations but the impacts of poor quality and raised water tables on diversity and abundance is unknown.

Non-permanent riverine wetlands (current)

Many of these wetlands have gone from being non-permanent to permanently inundated. The bulk of water present comes from irrigation tailwater but raised groundwater levels now mean the aquifer also contributes water to these wetlands (Fig. 10). The water entering the wetlands is now low in dissolved oxygen and high in oxygen demand. This anoxic water mobilizes nutrients which have increased in concentration due to adjacent agriculture. This oxygen poor and nutrient rich water is also percolating through the hyporehic zone into the aquifer reducing water quality in the aquifer.

Raised water tables combined with fire, weed infestation and grazing are negatively impacting the remnant vegetation along these streams (Fig.10). Lower canopy cover along the streams has aided the invasion of semi-aquatic grasses which have negatively impacted instream habitat quality. Permanent flows and increased nutrients also promote infestations of alien plants such as *Salvina* and *Hymenachne* which further reduce habitat quality by restricting water flow and blocking light passage through the water column (Fig. 10).

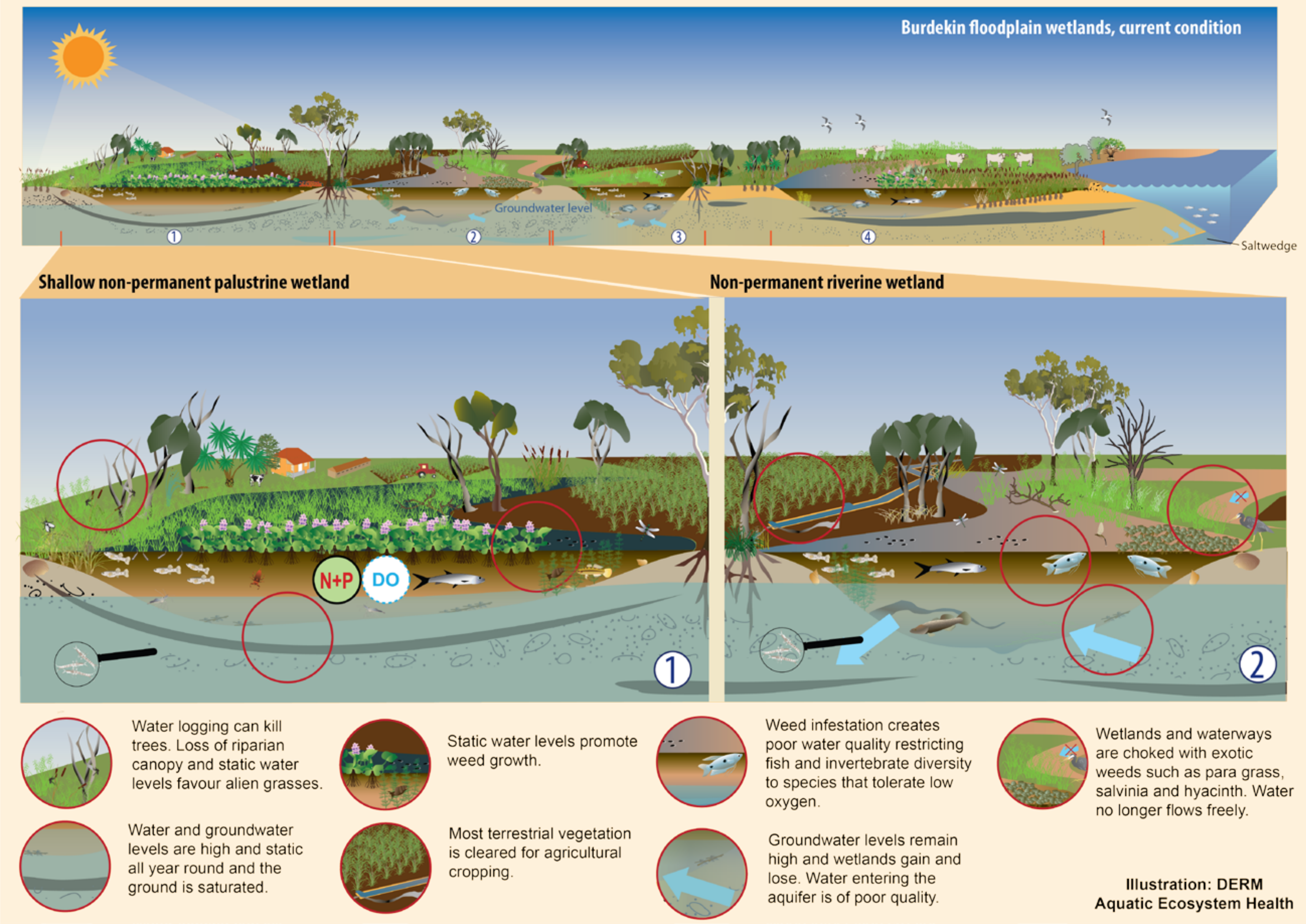


Figure 10 Conceptual model of current conditions in shallow non-permanent palustrine wetlands and non-permanent riverine wetlands. See Appendix B for legend.

The deterioration in water quality means diminished in-stream habitat quality for fish species that are sensitive to low oxygen. In the larger waterbodies many of the native fish species remain, but alien species are colonizing this disturbed habitat and impacting on native fish communities (Fig. 10). Changes in habitat such as large mats of *Salvinia* mean changes in community structure of the macroinvertebrates. Communities may also have lost species adapted to desiccation and sensitive to low oxygen levels. Stygofauna would be present in the underlying aquifer but their response to changed aquifer levels and water quality is not understood.

Deep permanent riverine wetlands (current)

The shallow overflow reaches of these streams have now been dug deeper to connect the deep lagoon reaches; water from the river is pumped through them turning them into distribution channels. There are now permanent flows in both the channels and lagoons. Water levels which prior to development fluctuated with season are now consistently high. The hydraulic head in the aquifer has risen above the stream bed in some places and in others the consistently high water levels are maintained artificially to create aquifer recharge (Fig. 11). These modifications have increased the flow of surface water into the aquifer largely to recharge groundwater for irrigation, however this increased connectivity may be greatly altering the water quality in the aquifer. Stygofauna would be present in the underlying aquifer but their response to changed aquifer levels and water quality is not understood.

Consistent, high water levels have promoted the invasion of alien plants such as water hyacinth, para grass and *Hymenachne (*Fig. 11). More organic matter is retained in the system as there is no opportunity for removal via decomposition on exposed banks in the dry season or early wet season flushes. These factors and the complete blanketing of some lagoons with floating plants results in chronically low oxygen levels. Irrigation water tends to flow through these lagoons with little or no mixing with resident anoxic water (Fig. 11). When larger wet season flows occur, this 'black water' is mobilized and creates a plug of anoxic water that flows down into the estuary greatly adding to the nutrient loads in the estuary and Great Barrier Reef lagoon.

The fish communities are dominated by alien species and those native species that can tolerate low oxygen levels (Fig. 11). The natural migration by fish is now limited by weed infested channels which form a barrier and the plugs of black water that come from lagoons with high wet season flows. The macroinvertebrate community would be dominated by anoxia tolerant species. Some lagoons remain with good habitat quality (such as Castalinalli's) and would be good targets for conservation through maintenance of migration pathways.

Shallow estuarine wetlands (current)

Shallow estuarine wetlands have now become predominantly freshwater palustrine (vegetated) wetlands often used as ponded pastures to graze cattle. (Fig. 11). These wetlands are being used by the Burdekin Water boards to maintain the freshwater head in the aquifer that keeps the salt wedge away from agricultural production areas (Fig. 11). This is done through bunding of wetlands so high tides can no longer reach them, as well as pumping in water and irrigation tailwater to maintain freshwater throughout the year. Maintenance of the freshwater head means groundwater level has increased and in some wetlands the aquifer is high enough to transect the wetland and create a direct connection. Water low in oxygen and high in nutrients is continuously seeping into the shallow aquifer changing its water chemistry.

The wetlands are now high in nutrients but low in oxygen and the loss of the previous fresh/salt cycles and marine connection means that the boom of microcrustacea that supported highly productive food webs no longer occurs. The maintenance of standing shallow water has created ideal conditions for the proliferation of the native plant *Typha* (or "cumbungi") as well as the alien water hyacinth and para grass. These have replaced most of the characteristic saltpan plants previously growing in these wetlands (Fig. 11). Poor water quality and the changed plant community have greatly reduced habitat quality favouring alien fish species. Fish diversity has been reduced by the bunding which prevents amphidromous species accessing the wetlands from the sea (Fig. 11). Bunding also stops the natural movement of other marine fauna such as penaeid prawns into these wetlands.

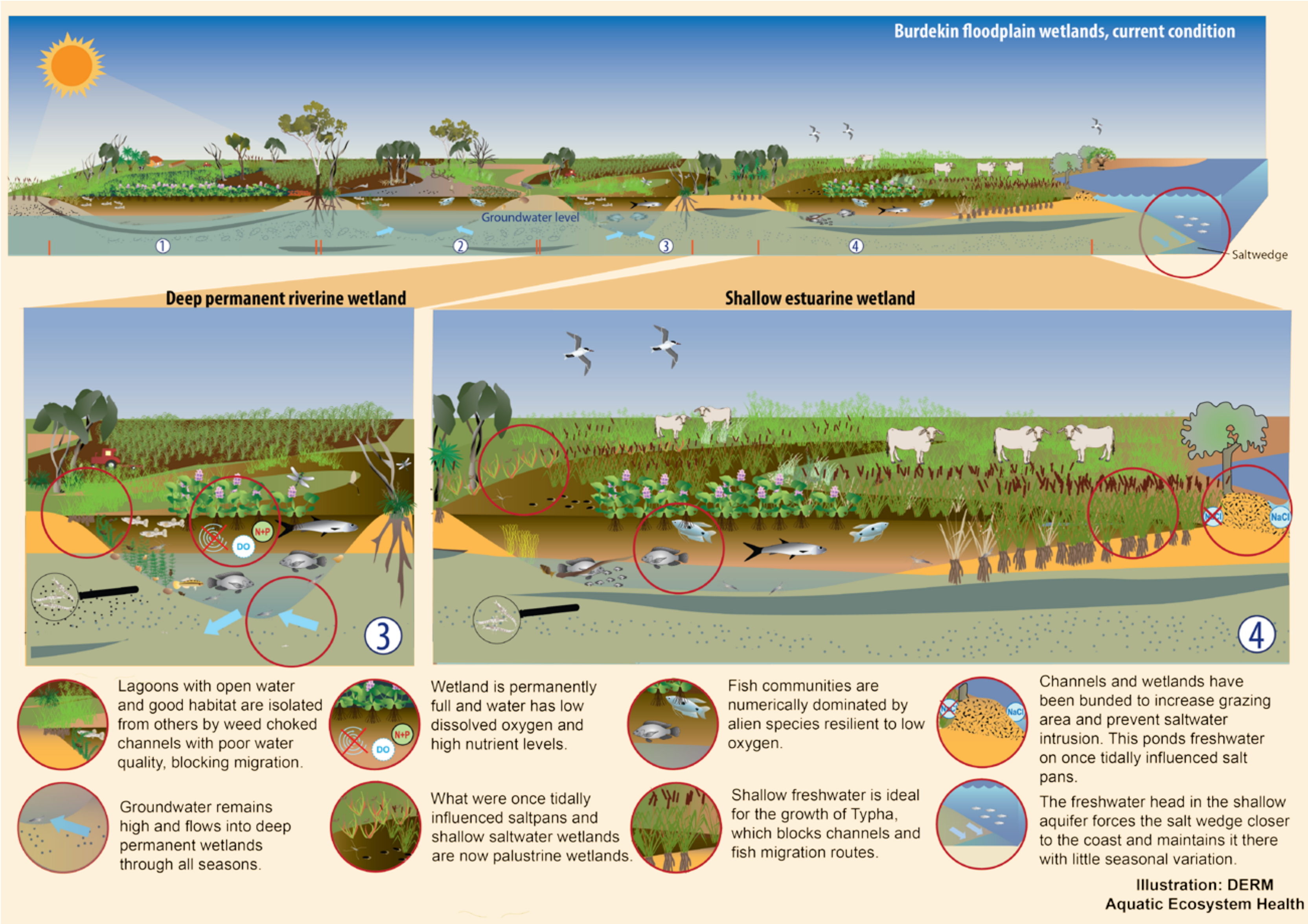


Figure 11 Conceptual model of current conditions in deep permanent riverine wetlands and shallow estuarine wetlands. See Appendix B for legend.

3 Characterisation of hydrological alteration to the lower Burdekin groundwater system

As described in the previous chapters, the lower Burdekin floodplain has undergone a wide range of hydrological modifications since intensive water resource development. Here we attempt to describe how the LBF aquifer levels have changed and how that may be impacting associated ecosystems. Most of the changes to the aquifer are a direct result of supplemental surface water flows and Managed Aquifer Recharge (MAR).

3.1 Indicators of hydrological alteration of the water table

Ecologically relevant characteristics of groundwater hydrologic regimes may include the magnitude, timing, frequency, duration and rate of change of a) water exchange processes between surface and subterranean ecosystems (i.e. recharge and discharge processes), and b) depth to water table, or subterranean zones of variable hydraulic permeability (SKM 2001; Eamus et al. 2006). Metrics associated with these characteristics can be used to establish the hydrological foundations for impact assessment studies (Richter et al. 1996; Kennard et al. 2010). We have used adjusted (ground level) depth to water table (DTWT) as an ecologically-relevant indicator of the *magnitude* of hydrological alteration to the lower Burdekin groundwater system (Table 1).

Table 1 Ecologically-relevant thresholds for depth to water table (DTWT) for the lower Burdekin system.

Water table threshold	DTWT (m)	Ecological relevance	Examples from the Burdekin
very shallow	<0.50	Likely that water table intersects root zone of shallow rooted phreatophytic species	the riparian species *Melaleuca fluviatilis* (Dowe 2004)
shallow	0.51 – 4.00	Likely water table intersect refugial pools within shallow non-permanent wetlands	Pelican Wetlands or The Lake
moderate	4.10 – 8.00	Likely that water table intersects root zone of all deep rooted phreatophytic species; Likely that water table does not intersect shallow non-permanent wetlands; thus they are dry; Likely that water table intersects refugial pools in deep wetlands	*Eucalyptus camaldulensis* (Dowe 2004); The Lake; Dick Bank Lagoon and deep pool on Barratta Creek
deep	8.10 – 20.00	Possible that water table intersects root zone of some deep rooted phreatophytic vegetation	*Eucalyptus camaldulensis* (Dowe 2004)
very deep	>20.00	Likely that surface GDEs are not supported	

Shallow-rooted vegetation can access groundwater if DTWT is <0.5 m and deep rooted vegetation can access groundwater to about 20 m DTWT (Table 1; Eamus et al. 2006). The deepest pools (i.e. extreme dry season refuges) of wetlands and lagoons in the Burdekin floodplain have a depth of between 4 - 5 m (e.g. Dick Bank lagoon; Perna, pers comm); thus, at 4 m DTWT the water table intersects wetlands and ensures persistence of water during the dry season (Table 1). There is also a suite of shallow non-permanent wetlands that are up to 4 m deep (Loong et al. 2001) Whilst groundwater can occur large distances below the surface, 20 m was considered as the ecologically-relevant threshold for very deep water table level because only subterranean GDEs would likely be supported below this level (Eamus et al. 2006).

Bore water levels are typically recorded less than ten times annually in Queensland so analyses of the timing of hydrological fluxes using daily data were not possible. Therefore, *timing* of water table fluxes was considered as either ‘wet season’ (i.e. 1st November the preceding year through to 30th April) or ‘dry season’ (i.e. 1st May to 30th October). This is typically seen as the wet season although we acknowledge that rainfall may be late therefore low DTWT may occur within the wet season, and cyclones can occur after May increasing DTWT into the dry season. *Frequency* was considered as the proportion of years in the time series satisfying a given criterion, and *duration* was broadly considered as the mode of recorded DTWT measurements in the time series. Finally, *rate of change* was considered on an inter-annual time scale for the time series data.

These facets of groundwater hydrology were used to identify 19 putative ecologically-relevant hydrological metrics (Table 2). These metrics were calculated for both the ‘pre-development’ and ‘post-development’ periods, with 1986 regarded as the final year of the pre-development time series and 1987 the first year of the post development time series. This cut-off was chosen as it corresponds with the completion of the Burdekin Falls Dam. The degree of hydrological alteration for each metric was calculated as per the Indicators of Hydrologic Alteration (IHA) approach (Richter et al. 1996):

$$\frac{\text{post} - \text{pre}}{\text{post}} * 100$$

post – is the calculated value of a given metric for the post-development time period, and

pre – is the calculated value of the same metric for the pre-development time period.

We used three arbitrary thresholds to interpret results of the IHA analyses. Less than 75% hydrological alteration was regarded as ‘noise’ or natural variability and was thus not interpreted as evidence for hydrological alteration. Between 75 % and 150 % was regarded as moderate hydrological alteration, and finally, over 150% was regarded as strong hydrological alteration. These thresholds are conservative and possibly underestimate the actual magnitude of hydrological alteration, but in the absence of more frequently sampled DTWT data, we felt it necessary to apply conservative criteria for interpreting results of these analyses.

We selected bores with high-quality long-term DTWT data, as robust analyses of hydrological alteration requires a minimum of 40 years data, with a minimum of 20 years pre-development data and 20 years post-development data (Richter et al. 1996). The criteria used for selecting bores were:

- Minimum 20 years of pre-dam data;
- Minimum 20 years of post dam data;
- Minimum of 6 records per year (average);
- Maximum of 1 year gap in data series;
- Existing adjusted DTWT data available.

Seven bores satisfied these criteria (Fig. 12).

Table 2 Putative ecologically-relevant metrics for setting hydrological foundations of groundwater systems.

#	Metric	Calculation of metric	Facet of flow regime	Relevant GDE
1	Proportion of years in time series with very shallow wet season water table (< 0.5 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Shallow rooted phreatophytic vegetation
2	Proportion of years in time series with shallow wet season water table (0.51 – 4.00m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Shallow wetlands[1], streams
3	Proportion of years in time series with moderate wet season water table (4.10 – 8.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Deep rooted phreatophytic vegetation[2]; Deep wetlands[3]
4	Proportion of years in time series with deep wet season water table (8.10 – 20.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Deep rooted phreatophytic vegetation
5	Proportion of years in time series with very deep wet season water table (>20.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Surface GDEs not supported
6	Proportion of years in time series with very shallow dry season water table (< 0.5 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Shallow rooted phreatophytic vegetation
7	Proportion of years in time series with shallow dry season water table (0.51 – 4.00m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Shallow wetlands[1]
8	Proportion of years in time series with moderate dry season water table (4.10 – 8.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Deep rooted phreatophytic vegetation
9	Proportion of years in time series with deep dry season water table (8.10 – 20.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Deep rooted phreatophytic vegetation
10	Proportion of years in time series with very deep dry season water table (>20.00 m)	Count of years in time series satisfying criteria/total number of years in time series	magnitude, timing, frequency	Surface GDEs not supported
11	Most commonly recorded wet season DTWT	Mode of wet season DTWT	Magnitude, timing, duration	All GDEs
12	Overall greatest wet season DTWT	10th percentile of wet season DTWT	Magnitude, timing	All GDEs
13	Overall least wet season DTWT	90th percentile of wet season DTWT	Magnitude, timing	All GDEs
14	Range of wet season water levels observed	Coefficient of variation of wet season DTWT	Rate of change	All GDEs
15	Most commonly recorded dry season DTWT	Mode of dry season DTWT	Magnitude, timing, duration	All GDEs
16	Overall greatest dry season DTWT	10th percentile of dry season DTWT	Magnitude, timing	All GDEs
17	Overall least dry season DTWT	90th percentile of dry season DTWT	Magnitude, timing	All GDEs
18	Range of dry season water levels observed	Coefficient of variation of dry season DTWT	Rate of change	All GDEs
19	Long-term rate of change in DTWT	R value calculated for time series	Rate of change	All GDEs

1 shallow wetlands: e.g. Pelican wetlands, The Lake 2 Deep rooted phreatophytic vegetation: e.g. Barratta Ck. 3 Deep wetlands: e.g. Dick Bank Lagoon

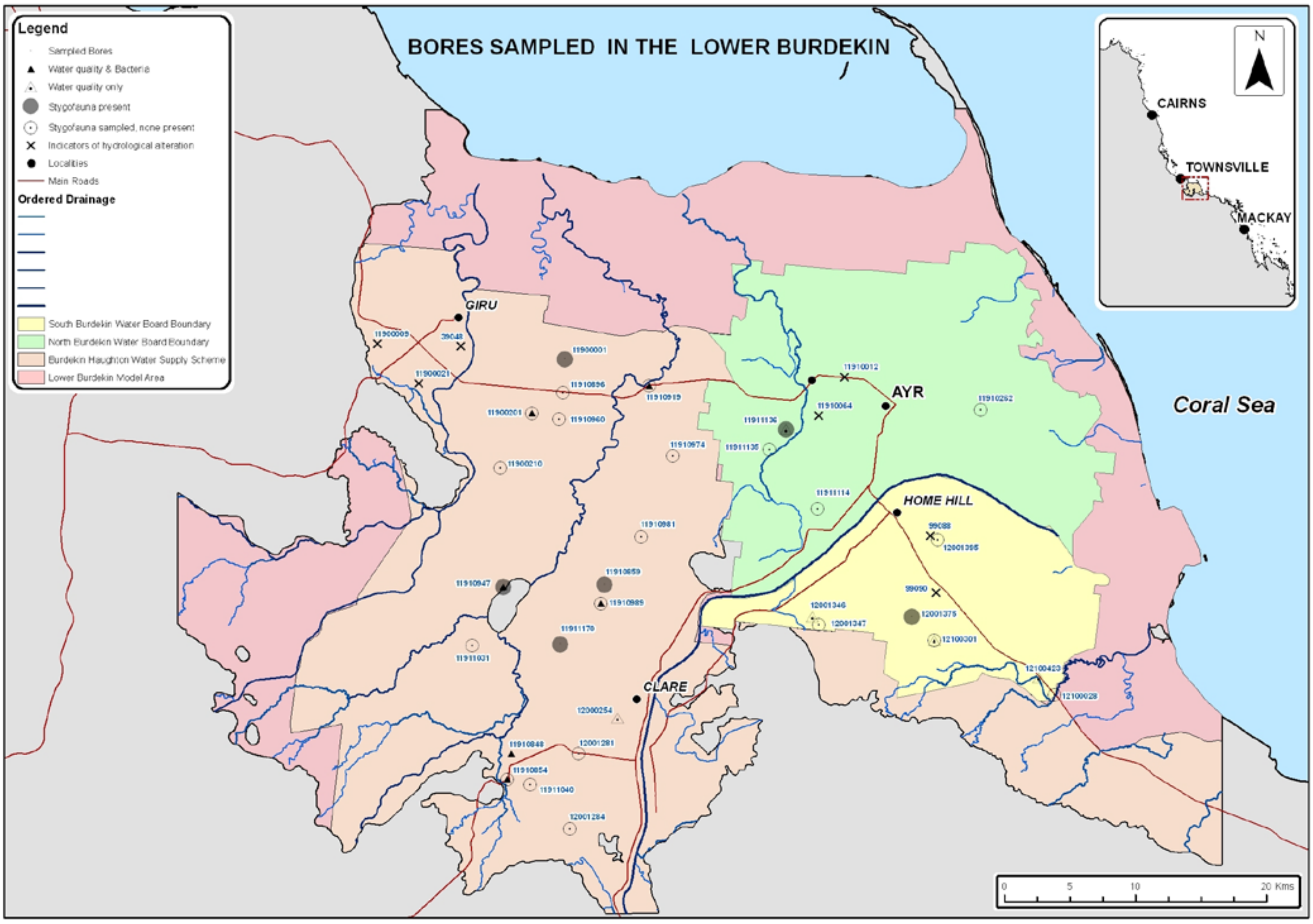


Figure 12 Map of the lower Burdekin floodplain showing location of bores used to assess hydrological alteration and sampled for stygofauna, water quality and bacteria.

Results for the IHA analyses, excluding results that were less than 75 %, are shown in Figure 13 (Appendix C shows all results). Three metrics (5, 6 and 10; Table 2) were not assessed for any bore because the water table was always shallower than 20 m, and the dry season water table was always deeper than 0.5 m. Of the other 16 metrics, three (i.e. metrics 13, 15, and 17) did not indicate hydrological alteration above the 75 % threshold. Thus, of the 19 metrics, 13 indicated moderate or strong hydrological alteration at one or more of the bores (Fig. 13).

No two bores had evidence for hydrological alteration for the same suits of metrics; rather, different sets of metrics were important for each bore, respectively (Fig. 13). However, where more than one bore was observed to have hydrological alteration for a given metric, the alteration was always in the same direction i.e. always increasing or decreasing for each bore (Fig. 13). This indicates localised variability in the degree of hydrological alteration to the aquifer but consistent overall trends.

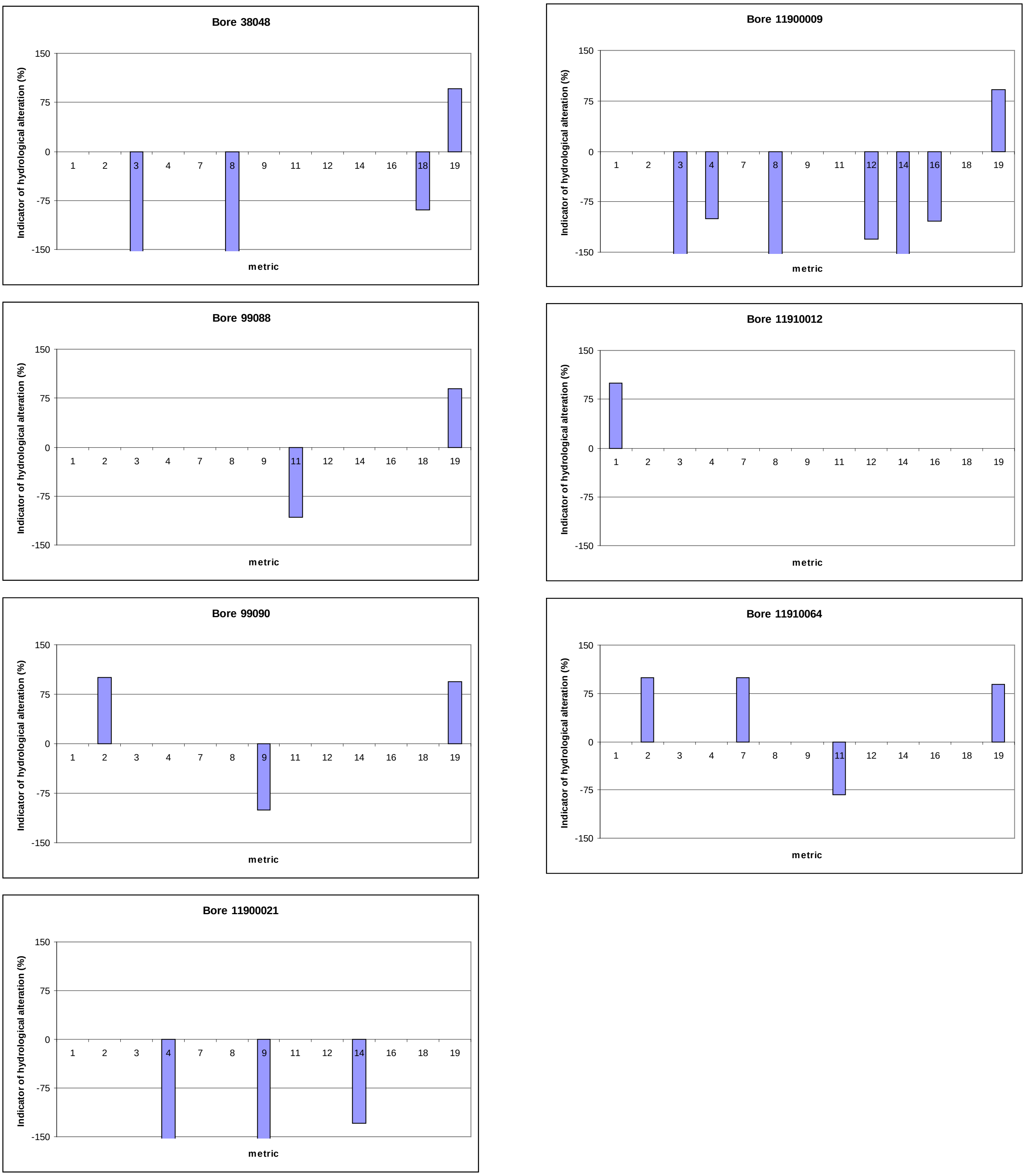


Figure 13 Results of analyses of Indicators of Hydrological Alteration of depth to water table for 7 bores in the lower Burdekin alluvial aquifer. The x-axis indicates the metrics analysed (see Table 2) and the y-axis indicates the percentage of hydrological alteration, ranging from 150 to -150 %. Bars that intersect 150 or -150 indicate strong hydrological alteration; bars that intersect 75 or -75 indicate moderate hydrological alteration. Results of IHA less than 75 % are not shown.

Strong hydrological alteration was observed for metrics associated with a decreasing proportion of moderate to deep dry and wet season DTWT observations (metrics 3, 4, 8 and 9 - Fig. 13). In other words, since the construction of the dam, there are fewer occasions (at three sites) where in both the wet and dry seasons water table levels are moderate to deep (4 - 20m). This trend is also illustrated by the hydrograph for bore 99090 as an example (Fig. 14). Strong hydrological alteration was also observed for the range of water levels experienced in the wet season at one site (metric 14 - Fig. 13). The range decreased meaning wet season DTWT has become more constant since dam construction.

A moderate decrease was also evident in metrics 11, 12, 16 and 18 (Fig. 13). These indicate the greatest wet and dry season DTWT levels are shallower (metrics 12 and 16) and the most common wet season DTWT (metric 11) is shallower. In other words water tables have risen. Also dry season water table levels have become less variable since dam construction (metric 18).

The analysis also showed a moderate increase in metrics 1, 2, 7 and 19 (Fig. 13). This indicates an increasing proportion of wet and dry seasons with very shallow and shallow DTWT (metrics 1, 2 and 7). Metric 19 which measures long term rate of change in DTWT was the most frequently altered metric (at five out of the seven bores; Fig. 13). This seems to indicate that even though water table levels have become less variable, the rate at which they change has increased and that this is quite common. As an example, this increased rate of change is evident in the hydrograph of bore 99090 with more rapid changes in DTWT happening more quickly giving steeper curves since commencement of irrigation (Fig. 14).

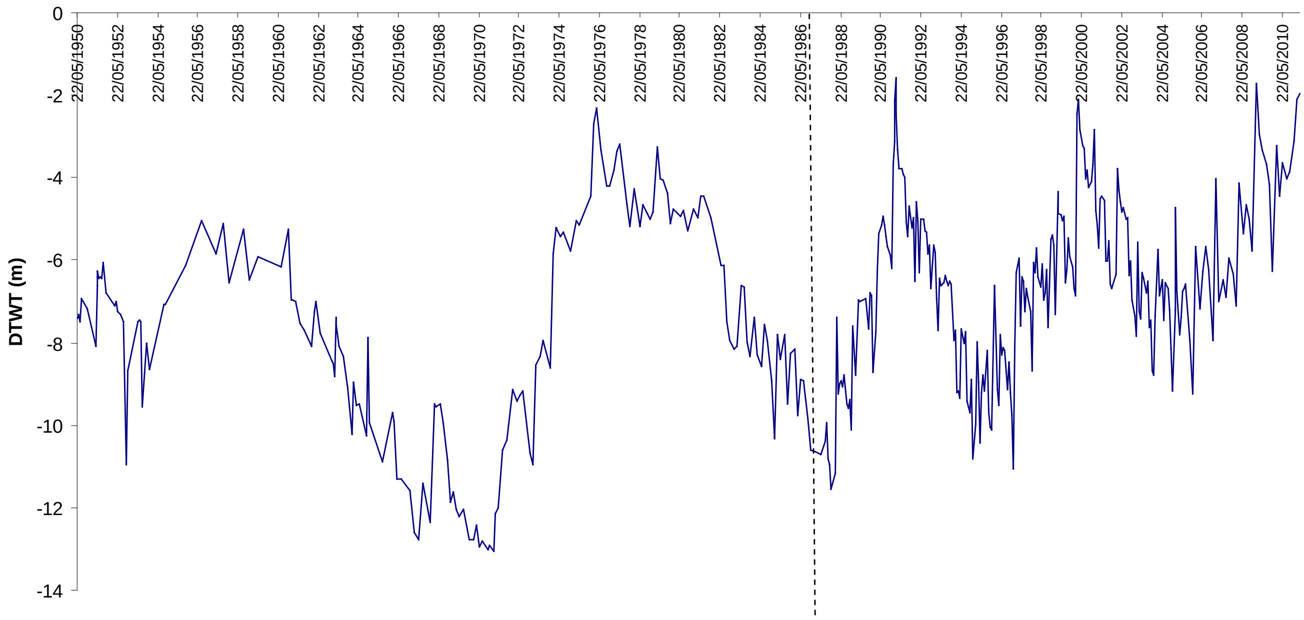


Figure 14 Hydrograph for bore 99090, showing increasing loss of proportion of time with deep DTWT. Dashed line delineates pre and post construction of the Burdekin Falls Dam.

Despite the conservative interpretation of the IHA analyses, the results demonstrate significant hydrological alteration to the lower Burdekin groundwater system. While the severity of hydrological alteration seems to vary geographically, water tables have generally risen and overall variability in DTWT has decreased. However, all bores have experienced rapid hydrological alteration since 1986; thus, the bores have all been modified rapidly at the same time, albeit in slightly different ways.

The ecological implications are that shallow rooted vegetation continues to have a dry root zone (< 0.5 m) in the dry season but is more often inundated during the wet season. Shallow non-permanent wetlands have become more permanent since dam construction with water tables more often 0.51 - 4 m in both seasons. The water table is less often at a depth of 4.1 - 20 m in both wet and dry seasons since dam construction. Given the other results this would appear to be because it is more often at the shallower depths. Ecologically this means phreatophytic vegetation on the floodplain would have their root zone inundated more often and deep refugial pools in wetlands would be full more often and display less seasonality in water level. How increased rates of change in DTWT translate to water levels in surface water GDEs is unclear but it may have implications for subterranean ecosystems.

A limitation with the current analyses is that we cannot tell whether the observed patterns would have occurred regardless of water resource development (i.e. we have no control for comparison). This situation can be remedied by repeating such analysis using modelled pre-development and current-development scenarios over the same simulation period when the groundwater model that is being created as part of this project is complete (McMahon et al. 2012).

3.2 Indicators of hydrological alteration to Barratta Creek

No-flow periods are important determinants of the ecology of non-permanent streams, with transitions from wet to dry periods being ‘temporal ecotones’ and dry stream beds providing critical habitat for specialised faunal groups (Steward et al. 2011). A shift in the flow regime from intermittent to permanent represents a disturbance to the hydroecology of seasonally dry streams, and is also indicative of hydrological alterations to the underlying groundwater system (i.e. a groundwater system that has shifted from a low discharge to a high discharge system).

In our analyses we have used the year 1986 as the point of hydrological alteration as in the previous section. The Index of Hydrological Alteration software (IHA, Richter et al. 1996) and the River Analysis Package software (RAP, Marsh 2004) are commonly used for analysis of flow time series but these techniques require continuous time series data. The IHA software can interpolate ‘data’ for days without real data but was deemed inappropriate because the arbitrary threshold of 10% missing data per year was exceeded in each year (Fig. 15).

Because of the considerable gaps in the daily flow time series between 1975 and 1988 (Fig. 15), we used the following metrics to quantify alteration to the naturally non-permanent flow regime of Barratta Creek:

1. the median and maximum flow for each month in each of the two time periods (i.e. pre- and post- 1986), and
2. the proportion of months, and a timeline of months for each year, within which zero flow days were recorded.

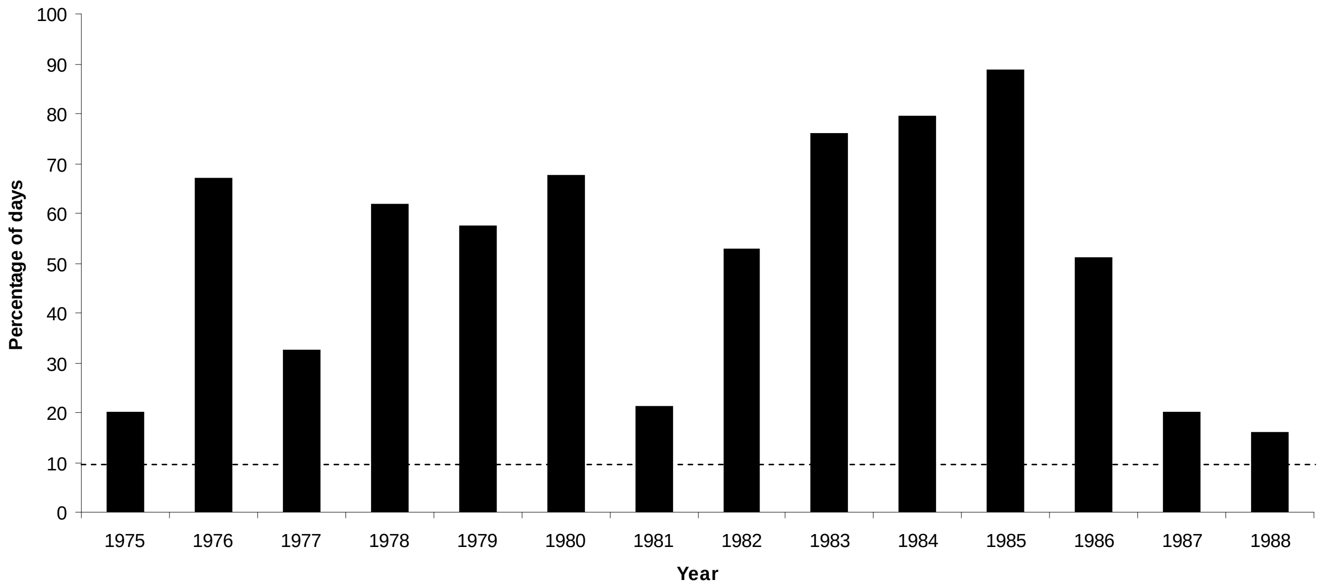


Figure 15 Percentage of days in years 1975 to 1988 without discharge data for Barratta Ck. The dashed line represents the arbitrary threshold of 10% missing data.

3.2.1 Median and maximum discharge

Hydrological analyses indicated an increase in the median and maximum dry-season discharge in Barratta Creek at gauging station 119101A (Figs 16 and 17). This hydrological alteration has been reported previously (Veitch et al. 2007) and is widely known by local landholders in the area. Indeed, for several months of the dry season (i.e. August and September), current magnitudes of discharge closely resemble the magnitude of former wet season flows, especially for maximum flow recorded (Fig. 17).

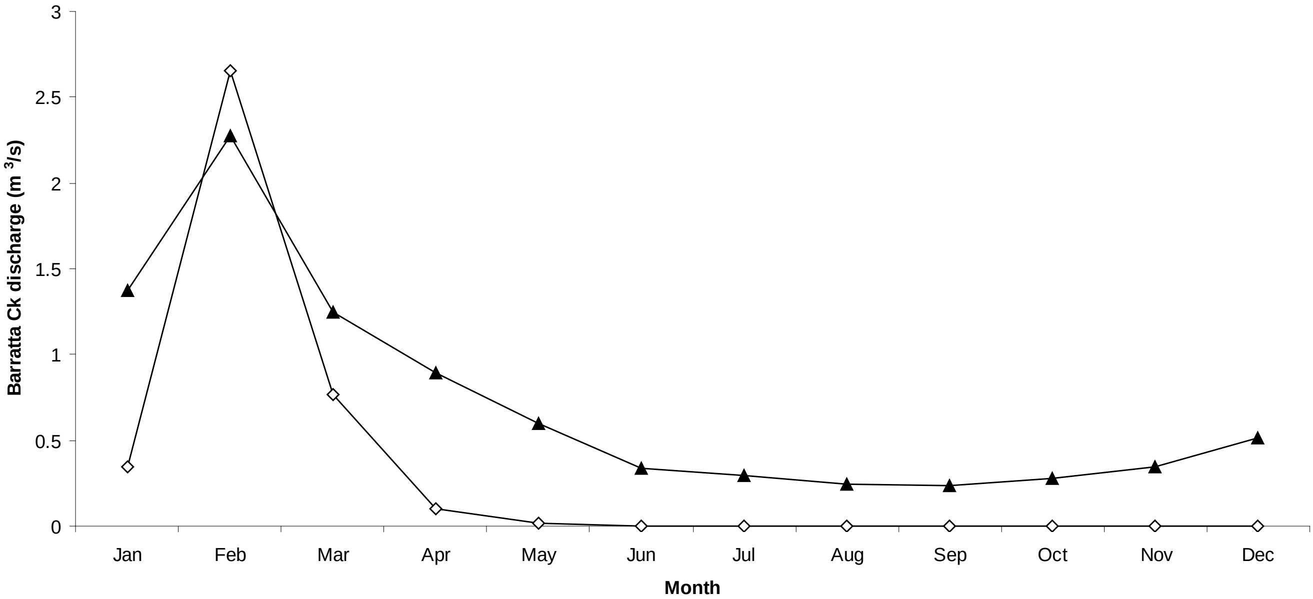


Figure 16 Median monthly discharge (m^3/s) at gauging station 119101A on Barratta Creek for pre- 1986 (open diamonds) and post- 1986 (black triangles).

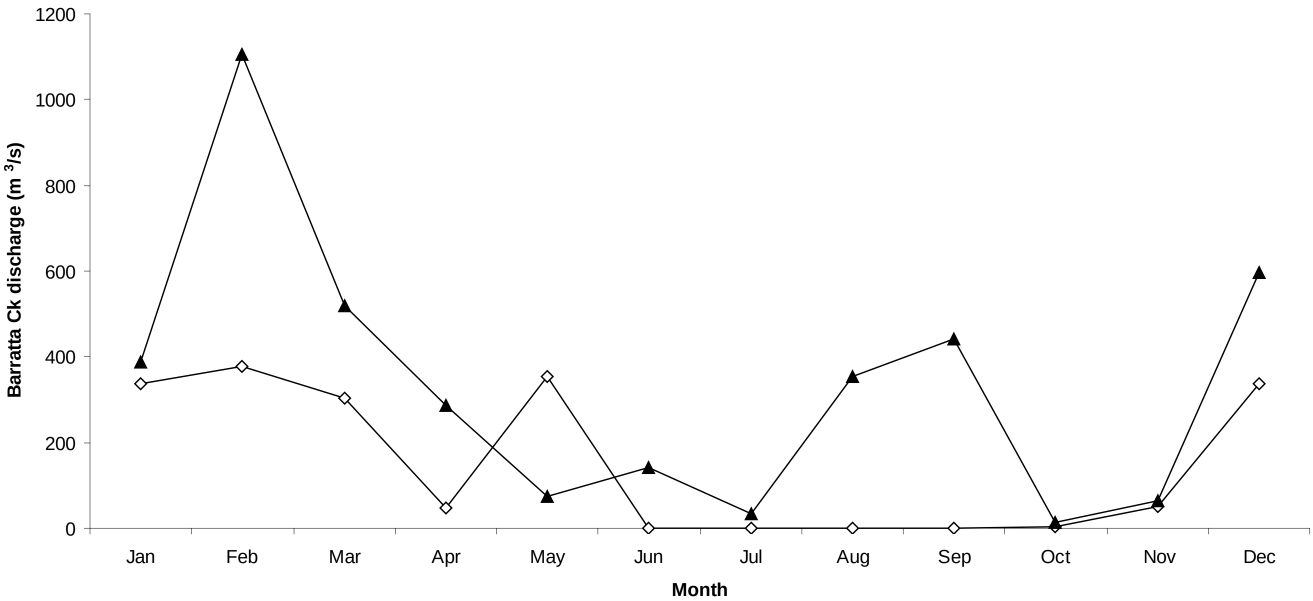


Figure 17 The maximum discharge (m^3/s) recorded for each month at gauging station 119101A on Barratta Creek for pre- 1986 (open diamonds) and post- 1986 (black triangles).

The maximum magnitude of wet season flows has also substantially increased, and anecdotal information suggests that Barratta Creek currently floods more frequently and severely than it did historically. The increased magnitude of discharge in Barratta Creek since dam completion is likely due to changes in drainage associated with the development of the BHWSS (Veitch et al. 2007).

3.2.2 Cease to flow

Data gaps are likely biasing some results (Fig. 18). Several dry season months (August to November) were especially under represented in the time period 1975-1988. The true range of zero flow months per year is likely in the range 0.4 to 0.7 as shown for the years 1975 and 1981, which had the greatest degree of completeness of flow data. In other words for 4 - 8 months in a year Barratta Ck ceased to flow for at least 1 day a month prior to completion of the Burdekin Falls Dam. Results from years with low data availability should be interpreted with caution.

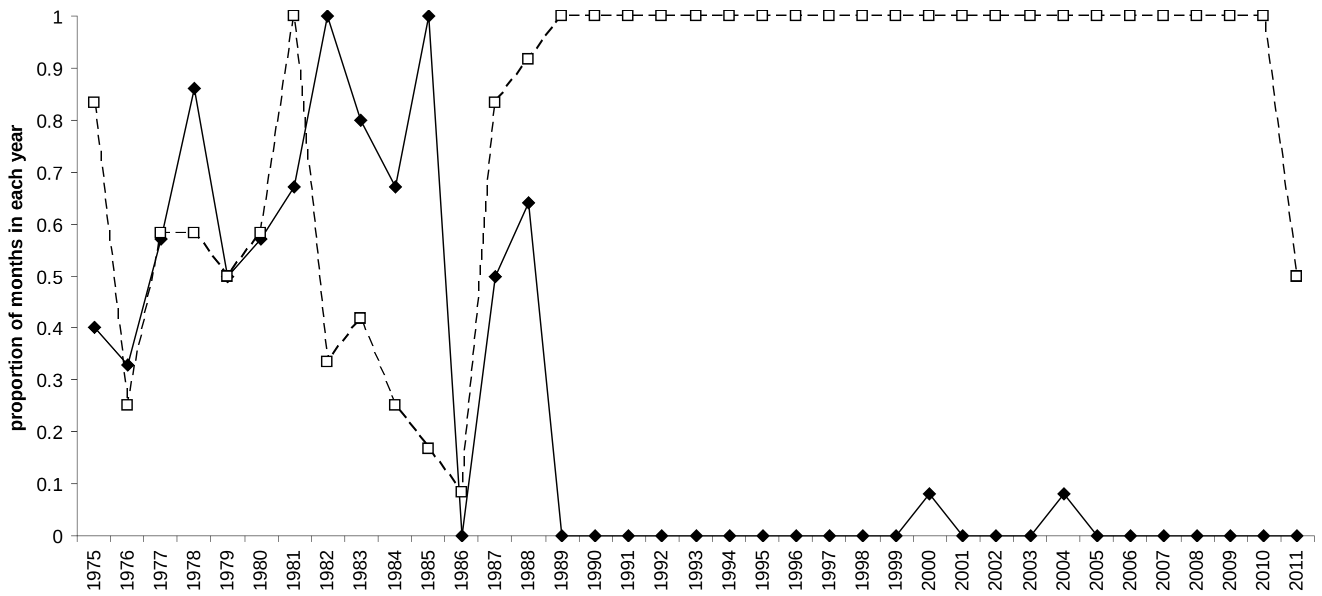


Figure 18 Proportion of months in each year within which zero flows were recorded (i.e. one or more days within the month with zero flow) at gauging station 119101A on Barratta Creek (black diamonds connected by think black lines). The proportion of months in each year for which flow data was available to calculate the proportion of zero flow months is also shown (open squares connected by dashed black lines).

However, even with data limitations it is still clear that Barratta Creek has ceased to flow much less frequently since 1986 (Fig. 18). For example, if we compare the percentage of months (with data available) that had no flow records in the period 1975 to 1986 with 1987 to 2011, we find that pre-development Barratta Ck was dry at least 1 day/month in 64% of months versus 5% of months in the post-development situation (Table 3). Even if the creek flowed every day in all the months with no data up to 1986, the percentage of months with at least 1 no flow day would still be 30%. This increasing permanence of Barratta Creek over the last 25 years is likely to be highly significant to the ecology of the creek (Perna 2003; Veitch et al. 2007; Perna et al. in press).

The likely causes of this hydrological change within Barratta Creek are land clearing and cropping, tailwater runoff, supplemental flow overflow, and groundwater infiltration. Presently the percentage contribution of each of these water sources in not fully documented, however the turbidity and conductivity of water in Barratta Creek is very similar to that of the irrigation supply channels, suggesting the majority of water is coming from irrigation water supply (Veitch et al. 2007).

Table 3 Time line of months with zero flow (black shading) and months with greater than zero flow (grey shading) in years 1975 to 2011. ND = no data.

Year	Jan	Feb	Mar	Apr	May	Jun	Jul	Aug	Sep	Oct	Nov	Dec
1975								ND	ND			
1976			ND	ND	ND	ND	ND	ND	ND	ND	ND	
1977									ND	ND	ND	ND
1978							ND	ND	ND	ND	ND	
1979						ND	ND	ND	ND	ND	ND	
1980							ND	ND	ND	ND	ND	
1981												
1982		ND				ND	ND	ND	ND	ND	ND	ND
1983			ND	ND			ND	ND	ND	ND	ND	ND
1984				ND	ND	ND	ND	ND	ND	ND	ND	ND
1985	ND	ND			ND	ND	ND	ND	ND			ND
1986	ND		ND	ND	ND	ND	ND	ND	ND	ND	ND	ND
1987	ND	ND										
1988								ND	ND			
1989-1999												
2000												
2001-2003												
2004												
2005-2010												
2011							ND	ND	ND	ND	ND	ND

4 Ecological assessments of the lower Burdekin groundwater system

4.1 Wetland assessments

The conceptual models (Figs 6 - 11) and analysis of changes to water table depth (section 3) illustrate major changes to the hydrology of wetlands on the LBF. One of the most significant changes appears to be reduced variability in the water levels in wetlands and, in some instances, a shift from non-permanent to permanent inundation. To assess ecosystem response to these changes we collected data on the aquatic macroinvertebrate communities of the wetlands and analysed the historic diatom record from wetland sediments. Macroinvertebrates are commonly used as indicators of ecosystem health (Rosenberg and Resh 1993) while diatoms are the most commonly used paleolimnological indicator of environmental change. The diatom record can provide us with information, therefore, about the hydroecology of wetlands prior to water resource development.

4.1.1 Assemblage structure of diatoms

Introduction

Diatoms are silicon based single celled algal organisms, between 5-200um (Sabater 2009) that are common in freshwater wetlands, estuaries and marine systems (Taffs et al. 2008; Smucker and Vis 2011). Diatoms have very short life cycles and a narrow range of preferred habitat, because of this they reproduce rapidly and their communities respond quickly to a change in conditions (Korhola 2007). Diatoms can be used to indicate past climate change over various timescales, and different resolutions (Paul et al. 2010) and to map changes in vegetation and complete changes in an ecosystems structure (Gasse and Campo 2001). Diatoms are a particularly good indicator of hydrological change including water quality and alterations to hydrological regimes (Chalié and Gasse 2002). Environmental changes can be inferred by looking at past diatom assemblages which have known pH, salinity and nutrient preferences and tolerances.

The primary aim of this study was to identify baseline conditions of three groundwater dependent floodplain wetlands and determine the degree to which, if at all, modern conditions deviate from that baseline. This study therefore directly informs Objective 2 of the report; assess and report responses by ecosystems to altered groundwater hydrology, as outlined in Section 1.1.

Methods

A full description of the methods and results can be found in the full report (Appendix D).

Three wetlands were sampled on the LBF: Swans Lagoon, Labatt Lagoon (also known as Payard's Lagoon) and EVT Inkerman. Swans Lagoon was selected as a reference site and conforms to the Queensland regional ecosystem (RE) classification of 11.3.27 and 11.3.27b. This corresponds to freshwater wetlands where vegetation is variable including open water with or without aquatic species. The b defines that it is a lacustrine or lake ecosystem with open water. Labatt Lagoons is consistent with classification 11.3.25g, riverine or fringing riverine wetland with no modification recorded. However, this current classification requires amendment as significant changes in hydrology exist. The EVT site at Inkermans is mostly classification 11.3.27x1c, palustrine wetlands dominated by *Shoenoplectus* and tending be seasonally inundated by freshwater tending to brackish as they dry. This site however has recently become more towards permanent and less saline due to increased bund height downstream.

Cores from the bottom of these wetlands were analysed for both their diatom communities and sediment characteristics (see Appendix D).

Results and discussion

The following is a summary of the key findings, the full findings are in Appendix D.

Assuming that the basal sediments were reached in all sites, then Swans and Labatt Lagoons are similar in age and are approximately 1300 years old, while EVT Inkerman, with sediments over 7500 years old, is considerably older.

Swans Lagoon

- It appears that for its entire existence Swans Lagoon has been a freshwater, circumneutral to alkaline, moderate to highly nutrient enriched wetland. The diatom taxon which dominates the lagoon today has been dominant for over 1300 years, suggesting that Swan’s Lagoon is within its natural regime of variability. In other words it is an appropriate reference site from which to compare the other two wetlands.
- Three phases were identified in the history of this wetland. Phase 1 (oldest): contains bands of coarse gravel that are indicative of sediment transport by a high energy stream. This may represent a time when there was still some connection to the main Burdekin River before a channel avulsion. Phase 2 (mid time period): In this phase, the proportion of benthic diatom taxa is at its highest. This, high levels of organic material, and sediment features indicate this period of time was one where there was less interaction between the Burdekin River and Swans Lagoon. Phase 3 (most recent): The most recent phase of the Swans Lagoon core record appears to represent a return to a greater influence of riverine inputs in the wetland.
- In comparison to marked and often wholesale changes observed in floodplain diatom records from southern Australia, there have been moderate changes in the diatom composition of Swans Lagoon.

Labatt Lagoon (also known as Payard's Lagoon)

- Three phases in the history of Labatt Lagoon were identified. The early portion of the record (phase 1) shows similarities to conditions at the Swans Lagoon reference site i.e. a non-permanent system with regular connection to the Burdekin River, albeit with lower energy conditions. Therefore, phase 1 may be considered to be representative of background reference conditions for this wetland.
- Environmental conditions clearly changed during phase 2, with increased sediment deposition in the wetland, possibly due to land clearance. Changes in the diatom community are suggestive of nutrient enrichment. There is insufficient chronological control to determine the timing of this change, however the transition to phase 1 most likely occurred in 1973.
- The final most recent phase marked a sudden and dramatic shift in environmental conditions. The rate of sedimentation increased from a combination of riverine and *in situ* sources. Increased sediment moisture was also recorded in this phase, possibly indicative of permanent inundation. Despite this, planktonic diatoms barely feature in the record. Instead, the diatom record marks a dramatic increase in the proportion of the epiphytic species which attach themselves to aquatic vegetation. The substantial representation of these taxa is clear evidence for a marked increase in aquatic vegetation in Labatt Lagoon.
- These data indicate that the Labatt Lagoon ecosystem is well outside its natural range of variability. This was likely through supplemental flows and the associated promotion of aquatic weed proliferation, as described in the conceptual models (section 2.4).

EVT Inkerman site:

- The sediment and diatom records suggest broad scale changes have occurred over much of this wetland. The lower sediments, dated at around 7,500 years old, are indicative of a low energy, estuarine environment. This means that the EVT wetland was likely permanently inundated with sea water for much of its history. This hypothesis is supported by the findings from another study about historic changes in sea level in the region (Woodroffe 2009) and the proximity of the study site to mangrove marshlands.

- Little information can be gained from the diatom record at this site due to the low abundance of valves in the majority of the record. Results from the uppermost 20 cm of sediment indicate a change away from non-permanent towards a permanently inundated site that is nutrient enriched.

This study has identified clear changes in the history of Burdekin River wetlands. In particular, it is clear that Labatt Lagoon is in a condition that is very different from the vast majority of its more than 1200 year long history. Labatt Lagoon has changed from a site that was non-permanent, but regularly flooded by the Burdekin River, to one which is permanently inundated and dominated by aquatic plants. It is likely that the ecosystem of EVT Inkerman wetland is also markedly different from that which existed naturally. However, because this information is mainly drawn from the nature of sediments that provide less precise information than diatoms, this conclusion is somewhat less certain. By contrast with the marked changes observed in the other sites, and in many other studies, the record from Swans Lagoon suggests that the site has undergone relatively little change.

4.1.2 Assemblage structure of macroinvertebrates

Introduction

Australia's guidelines for water quality monitoring and reporting (ANZECC 2000) state that bioassessment (assessment of biota) should be a vital part of assessing changes in aquatic ecosystems. DERM has developed a framework for indicator selection based on the principles in the ANZECC (2000) guidelines. Criteria include sensitivity to pressures, practicality of measurement and cost-effectiveness (Negus et al. 2011). Macroinvertebrates Aquatic macroinvertebrates (commonly insects but also crustaceans, molluscs, etc) are commonly used in bioassessment programs including the Stream and Estuary Assessment Program (SEAP) which is the Statewide assessment program for Queensland (Negus et al. 2011).

Changes to species richness, community composition and/or structure can indicate disturbance of the aquatic ecosystem. Macroinvertebrate groups have also been classified according to their tolerance to poor water quality (e.g. Chessman 2003) as well as flow and substrate preferences (Marshall et al. 2000). This means the presence or absence of particular groups provides a window onto the condition of the ecosystem. The guidelines for biological assessment are intended to detect important departures from a relatively natural, unpolluted or undisturbed state — the reference condition. The results of bioassessment may require interpretation using additional supporting information on water quality and physical conditions at site, etc.

Macroinvertebrate communities from wetlands on the Lower Burdekin Floodplain were sampled to determine the ecological implications of changes to hydrology. It was thought that data on fish communities would be more difficult to interpret because they are also affected by barriers to movement and the presence of exotic species (see section 2.4). Basic water quality variables were also measured to assist in the interpretation of the macroinvertebrate data.

Methods

Sixteen wetland sites on the Lower Burdekin Floodplain were sampled for aquatic macroinvertebrates in the dry season between August and September 2011 (Fig. 19). Sites were selected to assess changes in seasonality arising from flow supplementation for irrigation and wetlands were classified (Table 4) following the system used by the Queensland Wetlands program (EPA 2005). It was not possible to sample "natural" wetlands in pre-European condition on the LBF, because of ubiquitous pressures from agriculture and water resource development (including raised watertables). However, three sites with minimal localised impacts were selected for reference.

Three macroinvertebrate samples were taken at each site using a triangular dip net of 250µm mesh size and 25 x 25 cm dimensions. Each sample consisted of a 5m sweep to a depth of 30 cm through homogenous habitat. Habitat generally consisted of submerged, floating or emergent macrophyte. Each sample was then bagged and preserved

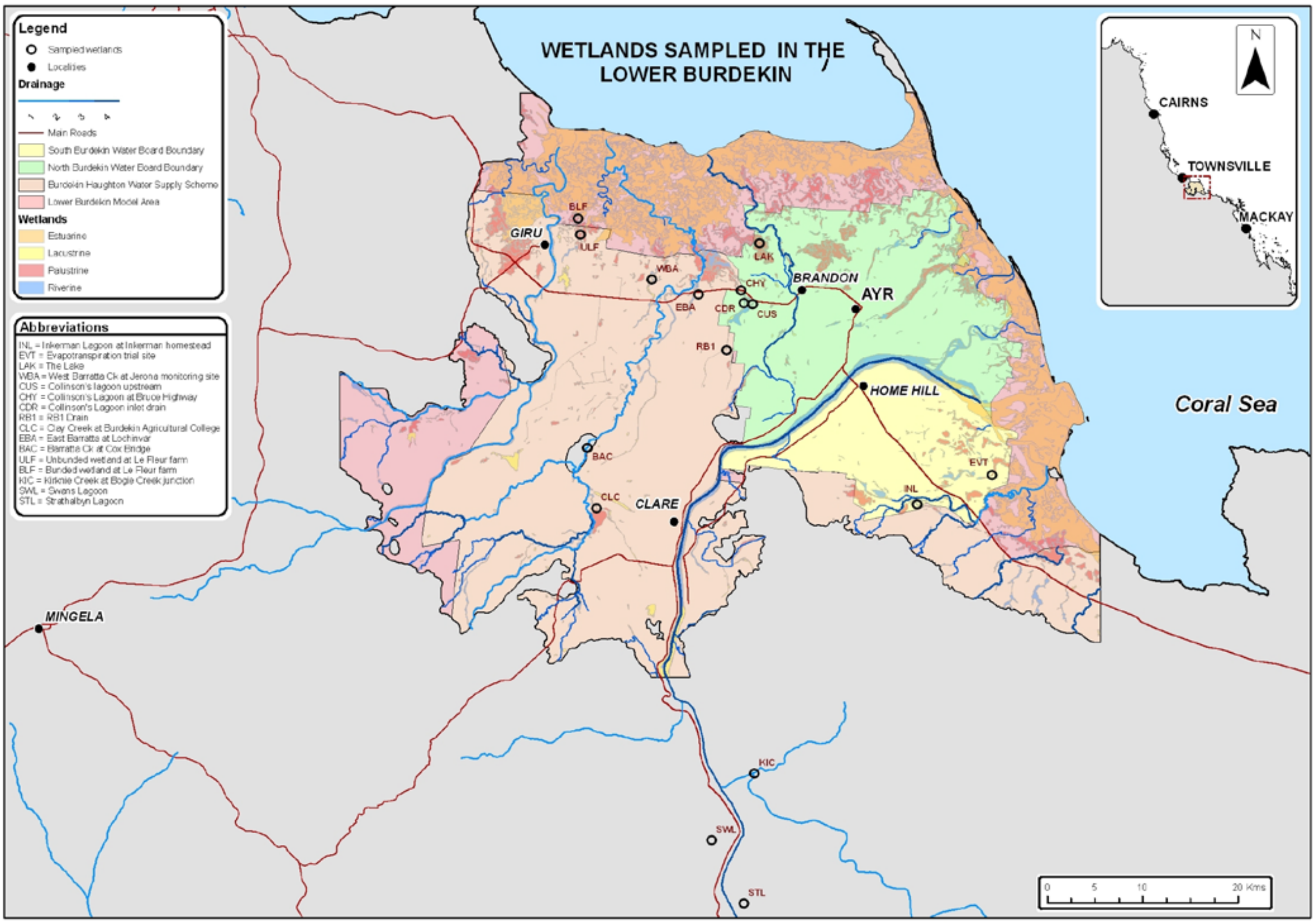


Figure 19 Map showing wetland macroinvertebrate sampling locations on the lower Burdekin floodplain.

in alcohol for later sorting in the laboratory. While an established Departmental methodology exists for wetland macroinvertebrate sampling (DERM 2011) we needed to modify the protocol because of the time constraints within which we had to work. In the laboratory samples were emptied into a tray and sorted by eye and using a stereo microscope until all specimens were recovered. Specimens were then identified by an expert according to DERM protocols (NRW 2008). Identification was generally to the family level except for some of the non-insects.

Physico-chemical water quality variables were recorded just below the water surface using a hydrolab datasonde.

Table 4 Details of wetland sites sampled for macroinvertebrates.

Site name	Site code	Ecological system	Natural or altered flow regime	Flow conditions
Swans Lagoon	SWL	Permanent lacustrine wetland	Natural	No flow
Strathalbyn Lagoon	STL	Permanent lacustrine wetland	Natural	No flow
Inkerman Lagoon at Inkerman homestead	INL	Deep permanent riverine wetland	Natural	No flow
Kirknie Creek at Bogie Creek junction	KIC	Deep permanent riverine wetland	Natural	No flow
Clay Creek at Burdekin Agricultural College	CLC	Non-permanent riverine wetland	Altered	flowing
Barratta Creek at Cox Bridge	BAC	Non-permanent riverine wetland	Altered	flowing
West Barratta Creek at Jerona monitoring site	WBA	Deep permanent riverine wetland	Altered	No flow
Collinson's lagoon upstream	CUS	Non-permanent palustrine wetland	Altered	No flow
Collinson's Lagoon at Bruce Highway	CHY	Non-permanent palustrine wetland	Altered	flowing
Collinson's Lagoon inlet drain	CDR	Non-permanent palustrine wetland	Altered	flowing
RB1 Drain	RB1	Artificial	Altered	flowing
East Barratta Creek at Lochinvar	EBA	Deep permanent riverine wetland	Altered	flowing
Unbunded wetland at Le Fleur farm	ULF	Shallow estuarine wetland	Altered	No flow
Evapotranspiration trial site	EVT	Shallow estuarine wetland	Altered	No flow
The Lake	LAK	Shallow estuarine wetland	Altered	No flow
Bunded wetland at Le Fleur farm	BLF	Shallow estuarine wetland	Altered	No flow

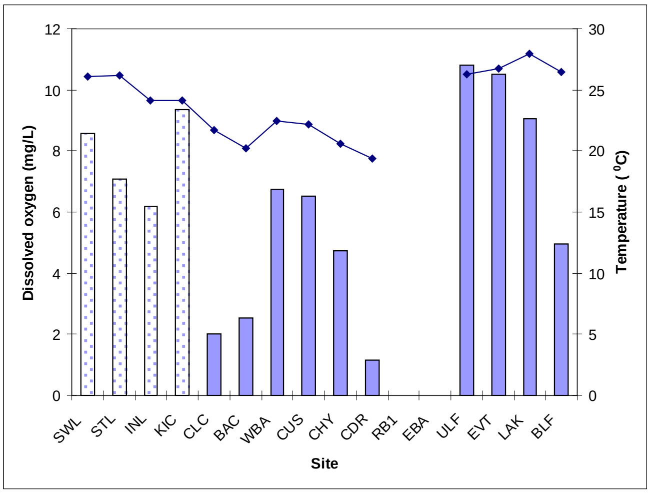
Results and discussion

Dissolved oxygen ranged quite widely from 1.1 mg/L (Collinson's Lagoon inlet drain/CDR) to 10.8 mg/L (Unbunded wetland at Le Fleur farm/ULF; Fig. 20). Variation was not related to flow conditions on the day (Table 4), and also doesn't appear to be related to temperature, as sites with the lowest temperature (Collinson's Lagoon inlet drain/ CDR and Barratta @ Cox Bridge/BAC) also had low dissolved oxygen which is the opposite of what would be expected (Fig. 20a). This may be due to the input of low-oxygen groundwater to these wetlands.

Electrical conductivity ranged from 47 µs/cm (West Barratta Ck/WBA) to 4901 µs/cm at the Evapotranspiration trial site/EVT (Fig. 20b). The Unbunded wetland @ Le Fleur farm/ULF and the Evapotranspiration trial site (EVT) were at least twice as saline as the other sites. These sites are on old saltpans which may mean salts are leaching into the wetlands from their substrata. Six of the sites (CLC, BAC, ULF, EVT, LAK, and BLF; Table 4) exceeded the guidelines for protection of aquatic ecosystems (Fig. 20b; DERM 2009).

Turbidity ranged from 2 NTU (Collinson's Lagoon @ Bruce Highway/CHY and RB1 drain/RB1) to 40 NTU (The Lake/LAK; Fig. 20c). These are all within guidelines for protection of aquatic ecosystems (DERM 2009). pH ranged from 7.1 (Inkerman Lagoon @ Inkerman homestead/INL) to 10.8 (Unbunded wetland @ Le Fleur farm/ULF). Sites were generally at the high end of pH for lowland streams in the region with five wetlands (SWL, KIC, ULF, EVT and LAK; Table 4) exceeding a pH of 8 which is the guideline for protection of aquatic ecosystems (Fig. 20d; DERM 2009) but it is unclear how relevant these riverine water quality guidelines are for wetlands.

a)



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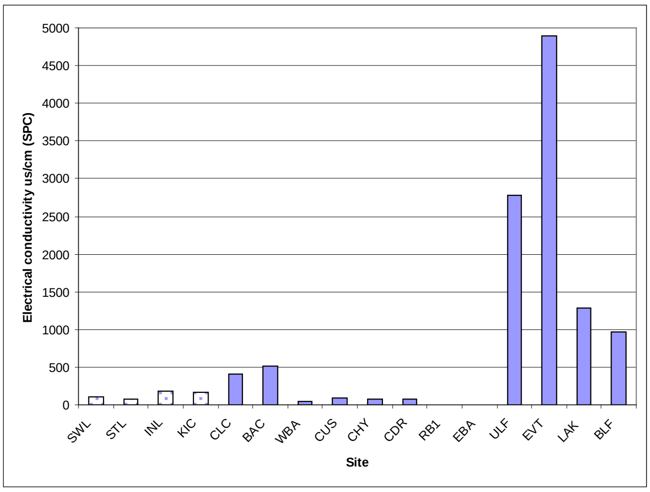
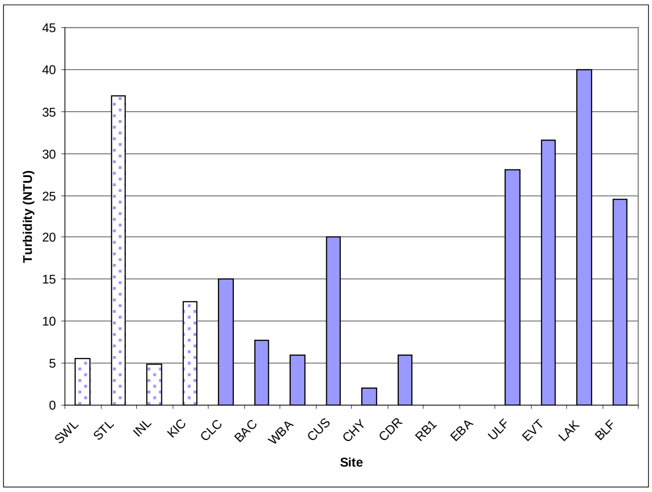


c)



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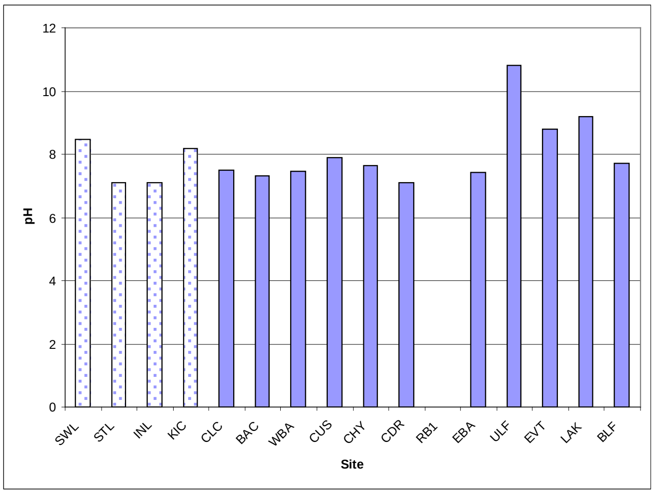
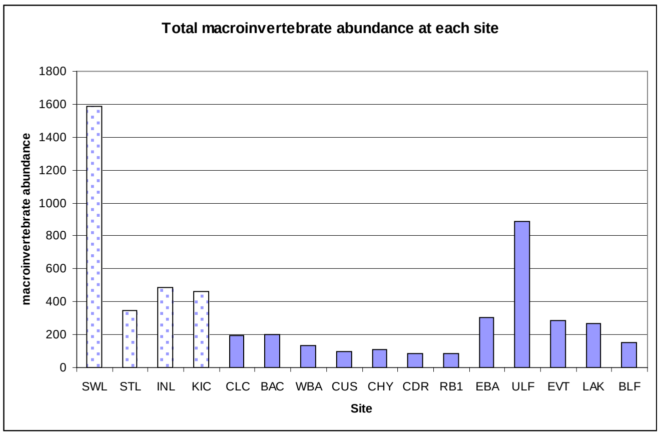


Figure 20 Water quality readings at the 16 wetlands sampled for macroinvertebrates. Wetlands with unregulated hydrology are indicated by stippled bars.

Total macroinverebrate abundance from sites (comprising 3 replicates) varied from 87 (Collinson’s Lagoon inlet Drain/CDR) to 1586 (Swans Lagoon/SWL; Fig. 21). Abundance sometimes varied widely between replicate samples, most notably at Unbunded wetland @ Le Fleur farm/ ULF (Fig. 21). The third replicate had a high number of Atyid shrimps along with 94 copepods (that weren’t present in other replicates). This was the only sample taken in nardoo (*Marsilea* sp.) a floating wetland macrophyte species. Swans Lagoon/SWL had significantly higher abundance than all sites except the Unbunded wetland @ Le Fleur farm/ULF (Fig. 21).

A total of 67 taxa (primarily families) were recorded from the 16 wetlands. The lowest total site richness of three taxa was at the same site that had the lowest overall abundance - Collinson’s lagoon inlet drain/CDR (Fig. 22) while the highest richness was 33 taxa (Kirknie Ck @ Bogie Ck junction/KIC; Fig. 22) a riverine site. The four unregulated wetlands had the highest total richness, although once within site variability is taken into account they are not significantly different from a number of the regulated sites (Fig. 22).

a)



b)

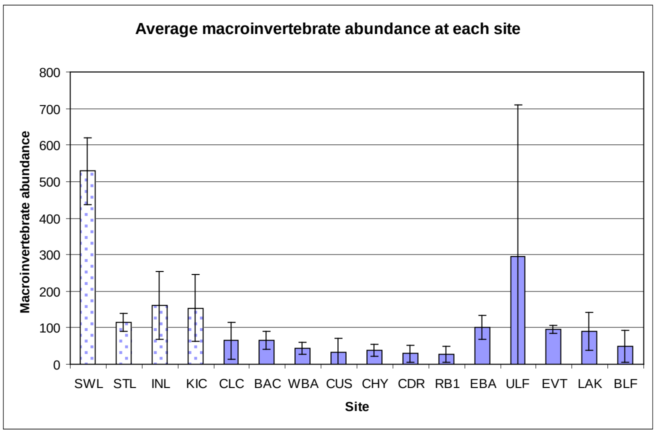
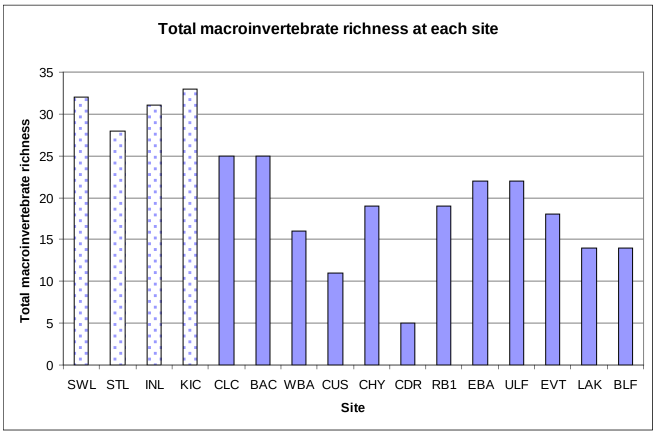


Figure 21 Total macroinvertebrate abundance from 3 replicate samples at 16 wetlands (a)) and average macroinvertebrate abundance at each site showing 1 standard deviation (b)). Wetlands with unregulated hydrology are indicated by stippled bars.

a)



b)

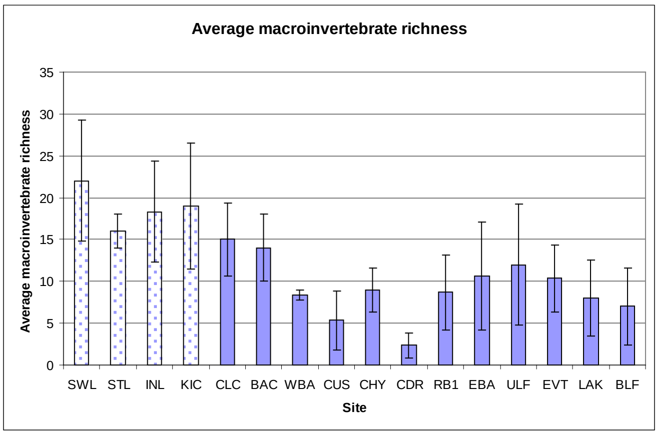


Figure 22 Total macroinvertebrate richness from 3 replicate samples at 16 wetlands (a)) and average macroinvertebrate richness at each site showing 1 standard deviation (b)). Wetlands with unregulated hydrology are indicated by stippled bars.

Multivariate analysis of the data was done to determine how similar sites and samples were to each other in terms of macroinvertebrate communities. Primer statistical software was used (Clarke and Gorley 2006). Macroinvertebrate count data was $\log_{10}$ (x+1) transformed to downweight the influence of abundant taxa and a pairwise Bray-Curtis similarity matrix between samples calculated (Clarke 1993). An analysis of similarity (Clarke 1993) on this matrix indicated that sites were significantly different from each other. In other words, the differences that we might see within a site or wetland (in our case represented by three sweep samples) shouldn't overshadow differences between the wetlands. This result was consistent with presence absence data indicating results weren't driven by invertebrate abundance alone.

Multi-dimensional scaling (MDS) ordination (Clarke 1993) was plotted to help interpret the results. This presents the relative dissimilarities of the sites in two-dimensional space so that points that are close together represent samples with similar macroinvertebrate composition, while those far away correspond to different communities. The plot showing sites with different symbols (Fig. 23) highlights that samples taken within a single wetland were generally similar (close together). Some outliers occur where samples had very low taxa richness (e.g. RB1 drain/RB1, Collinson's Lagoon at Bruce Highway/CHY and Collinson's Lagoon upstream/CUS).

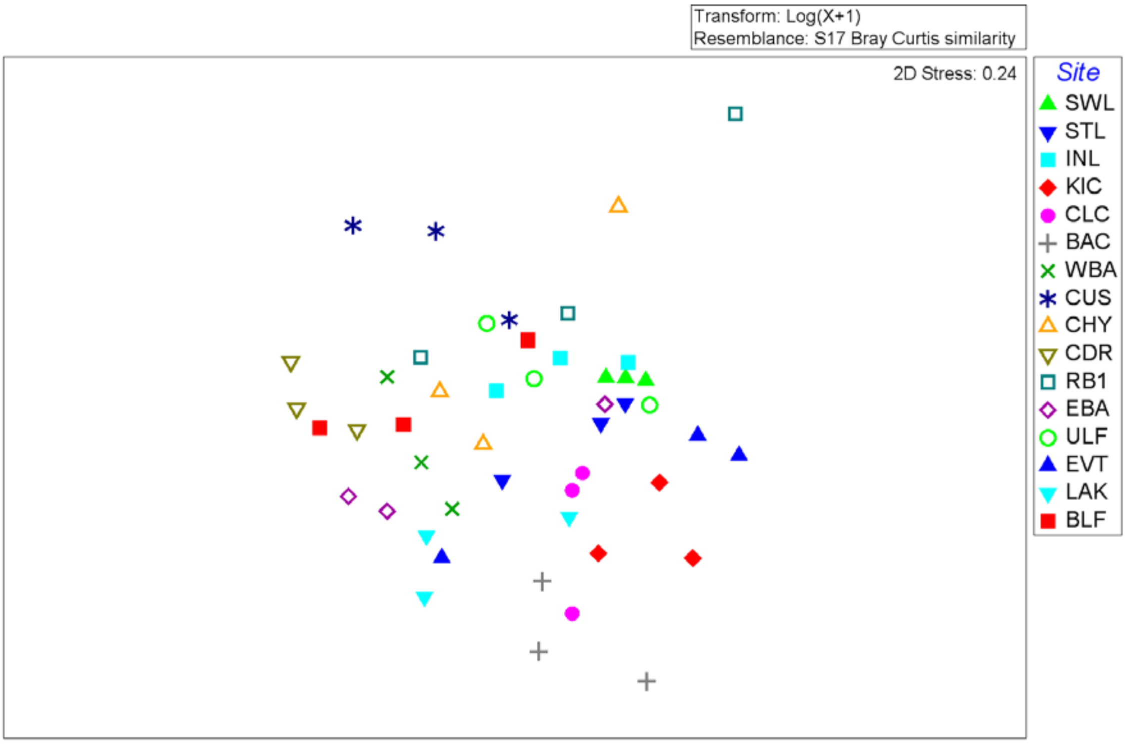


Figure 23 Two-dimensional MDS plot of wetland macroinvertebrate data showing sites.

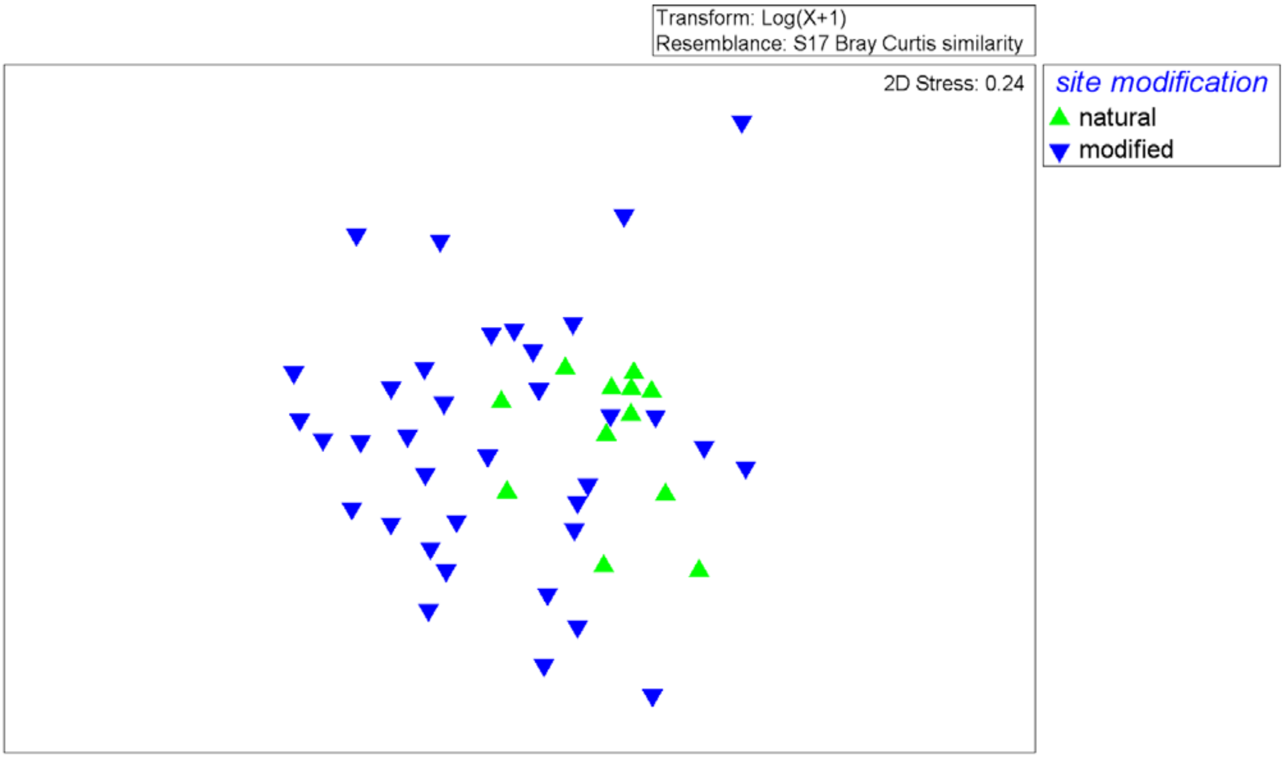


Figure 24 Two-dimensional MDS plot of wetland macroinvertebrate data showing site modification. "Natural" denotes a wetland with close to natural hydrology and "modified" denotes a wetland with altered hydrology.

One of the factors that may be contributing to differences between sites is their degree of hydrological modification. When sites are coded according to whether or not their hydrology has been modified (Fig. 24) the unmodified ("natural") sites cluster together but aren't statistically different from the modified sites (global r = -0.013). R values vary from 0 to 1 with 0 indicating two wetlands are indistinguishable while a value of 1 indicates two wetlands are extremely different. Pairwise r values from the ANOSIM analysis indicate the four unmodified sites were not always similar to each other which may indicate natural variability in macroinvertebrate communities over and above any anthropogenic impact. Kirknie Ck @ Bogie junction in particular was very different from most of the other sites with 11 out of the 15 other sites (including two of the other unmodified sites) having an r > 0.75 which indicates they are well separated. This may reflect a difference in flow regime (dries to isolated pools in the dry season) and substrate (sandy as opposite to vegetated) at this site.

Sites with unmodified flow regimes were most similar (r <0.25) to Unbunded wetland @ Le Fleur farm/ULF, East Barratta Ck @ Lochinvar/EBA, and Collinson's Lagoon at Bruce Highway/CHY. These similarities may be indicators of good macroinvertebrate habitat being a more important driver than changes in flow regime at these sites. Collinson's Lagoon has recently been the focus of habitat restoration works which may also be reflected in these results.

Macroinvertebrate community composition may also be driving differences and similarities between wetlands on the lower Burdekin floodplain. Several taxonomic groups seem to be found primarily at the unmodified (and similar) sites and may be what distinguishes them from the others. These include: mayflies (especially the family Baetidae) at KIC, EBA, SWL, STL and ULF; caddisflies (especially the family Leptoceridae) at STL, KIC and EBA; and damselflies (especially the family Coenagrionidae) at ULF, SWL, STL, INK and EBA. Mayflies and caddisflies are generally considered sensitive to impact (Karr and Chu 1999) which supports the idea that these sites are the least impacted of those sampled.

All unmodified sites were very different (r >0.9) from Collinson's Lagoon inlet Drain/CDR and Barratta Ck @ Cox Bridge/BAC - this is also evident in the ordination (Fig. 23). These wetlands may therefore represent the most impacted of those sampled. With Collinson's drain/CDR this is supported by the very low macroinvertebrate richness found (Fig. 22) and may reflect poor water quality, in particular low dissolved oxygen at this site (Fig. 20b). Barratta Creek @ Cox Bridge has relatively high sustained flows and a leaf litter substrate (as opposed to macrophyte). These may have been important factors determining what macroinvertebrate taxa were present and distinguishing it from other sites sampled.

In summary, there weren't clear differences in macroinvertebrate communities found in wetlands with different levels of flow regime modification, although unmodified sites tended to have higher overall richness. These ambiguous results may reflect the difficulties finding sites unaffected by groundwater modification or other pressures from agriculture in the lower Burdekin floodplain. Also, all unmodified sites were in permanently inundated wetlands. This may mean changes to hydrology were not as obvious as they may have been if communities from unmodified non-permanent systems had been sampled as a point of reference (the change from non-permanent to permanent inundation being more ecologically significant). Macroinvertebrate communities on floodplains may also be reasonably robust to some perturbation if water quality is not severely degraded and habitats are sufficient for feeding and reproduction, as has been found on weed-infested tropical floodplains (Douglas and O'Connor 2003).

4.2 Ecology of the aquifer environment

It was posited that subterranean ecosystems may also have been impacted by the changes to groundwater dynamics in the LBF described previously (Figs 6 - 11 and section 3). The two main biotic components of groundwater ecosystems are subterranean aquatic fauna (stygofauna) and microbes (Tomlinson and Boulton 2010). No information was available on these groups so we collected preliminary data to determine whether or not they may have future relevance to water resource planning, as assets for example.

4.2.1 Stygofauna diversity

Methods

Twenty-six pre-existing groundwater monitoring bores were sampled for stygofauna in September 2011 (Fig. 12). Bores had an internal diameter of 50 mm and were cased with PVC piping except for a length that was slotted to allow entry of groundwater and stygofauna. At each bore, 300 L of water was pumped using either a Waterra Powerpump 2 peristaltic pump or an airlift pump and passed through a 63 μm sieve. Equipment was decontaminated after each sample. Samples were preserved in 100 % ethanol. Within 24 hours of sampling, stygofauna was picked from each sample using a Nikon model 106333 stereo dissecting microscope and preserved in fresh 100 % ethanol.

Molecular analyses were also conducted on specimens from the Parabathynellidae and Bathynellidae families using methods described in Appendix E. Molecular methods have proven an effective means for describing patterns of connectivity among populations of aquatic fauna, as well as for identifying the factors that act to either constrain or enhance genetic connectivity in lotic systems (e.g. Hughes 2007; Cook et al. 2011). Furthermore, many of these molecular studies have revealed highly divergent genetic lineages within described species, some of which have turned out to represent reproductively isolated biological species (e.g. Cook et al. 2008). Molecular approaches have also been used to assess population genetic structure in subterranean aquatic fauna. These studies have invariably indicated high levels of cryptic endemism and localised population structuring across several types of groundwater system, such as karstic or calcrete aquifers in western Australia (e.g. Cooper et al. 2008; Finston et al. 2007; Guzik et al. 2008, 2011). We know of no studies that have used molecular methods to assess population genetic structure within stygofauna from alluvial aquifers in eastern Australia.

Results and discussion

Stygofauna were caught from six of the 26 bores that were sampled (Table 5). Six bores also had small numbers of terrestrial invertebrates collected, primarily from the family Coleoptera. These are not reported here as they are not representative of groundwater quality. Stygofauna samples consisted predominantly of syncarids from two families: Bathynellidae (4 individuals from bore 11911170) and Parabathynellidae (23 individuals from all six bores). Two copepods were also collected in this study, one cyclopoida and one harpacticoida. Thus, only crustacean species were sampled from the Burdekin River alluvial aquifer. This demonstrates that even substantially modified groundwater systems can be biologically active. However, the number of bores containing stygofauna (i.e. 23 %) is low compared to other studies in the Pioneer River alluvial aquifer (NRW 2007) and in the Bundaberg region (Hancock, unpublished data). In the absence of pre-development data, it can only be speculated whether hydrological alteration of the groundwater system has influenced the distribution and / or abundance of stygofauna. However, three of the six bores containing stygofauna were in areas with the most pronounced hydrological modifications; thus, it is possible that stygofauna are resilient to rising groundwater levels given the low likelihood that refugial pockets exist for stygofaunal recruitment.

Table 5 List of bathynellaceans and lineages found in the lower Burdekin River alluvial aquifer.

Bore registration number	**Taxa found**	**Bathynellaceans and lineages**
11900001	Parabathynellidae	
11910947	Parabathynellidae	Parabathynellidae P-B
11911170	Parabathynellidae Bathynellidae	Parabathynellidae P-A Bathynellidae B-A
11910859	Parabathynellidae	Parabathynellidae P-A
11911136	Parabathynellidae	Parabathynellidae P-A
12001375	Parabathynellidae	Parabathynellidae P-C Parabathynellidae P-D

The gene tree (Fig. 25) shows four highly divergent genetic lineages within the Parabathynellidae, with genetic differentiation among them ranging from 21 to 35 % (Appendix E). This suggests high cryptic species diversity within this family from the Burdekin River alluvial aquifer, and the large magnitude of genetic differentiation among them suggests the possibility for them to represent distinct genera. Three of the four lineages within the Parabathynellidae we report, and the single taxon within the Bathynellidae were found only in a single bore (Table 5). This pattern of apparent endemism is striking, although this result should be interpreted with caution as once-off sampling has been shown elsewhere to capture only about 46 % of the species present in a bore (Eberhard et al. 2009). Thus, it is possible that at least some of the species are more widely distributed within the lower Burdekin River alluvial aquifer than we report in this study; some may even be more widely distributed in eastern Australia. However, as more than 50 % of known species of stygofauna are short-range endemics (Eberhard et al 2009), it is possible that some of these species have very narrow distributions, even within an alluvial aquifer. Regional-scale sampling and repeated sampling of selected bores is needed to determine which of these taxa are truly short-range endemics.

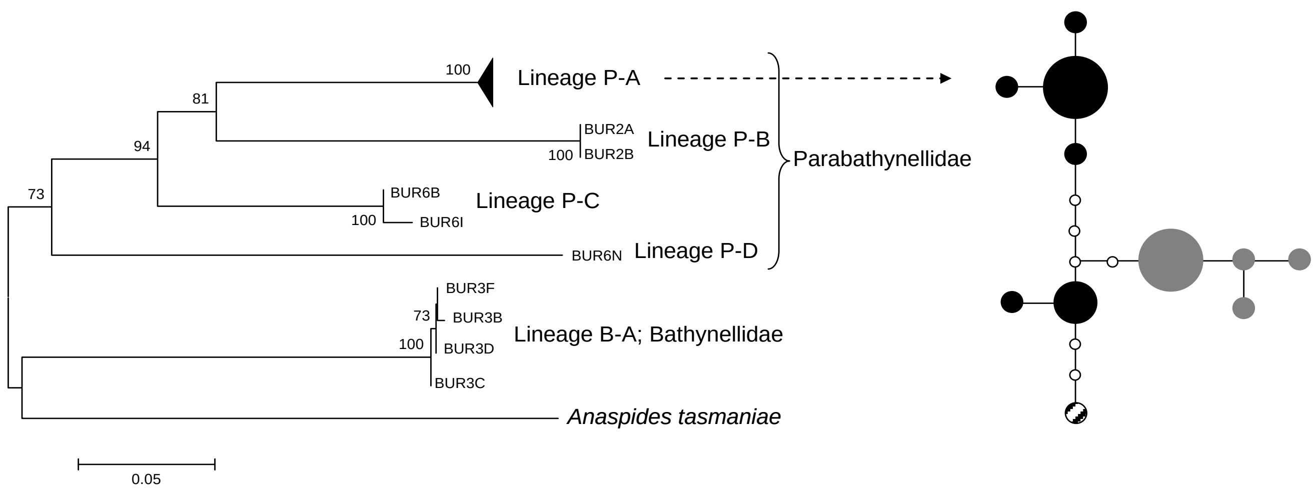


Figure 25 Mitochondrial DNA gene tree for stygofauna showing divergent lineages within Parabathynellidae and the single lineage within Bathynellidae. The genotype (haplotype) network for lineage PA is also shown and is colour coded by the bore sampled – black is 11910859, grey is 11911170 and cross-hatched is 11911136. Each colour circle represents a genotype and the way they are connected shows how the genotypes are related to each other.

Lineage P-A was distributed across three geographically proximate bores, and in sufficient numbers from two of these bores to enable population genetic analyses. These two populations of Lineage P-A had strong and significant population genetic differentiation ($\Phi_{ST} = 0.603$; $P < 0.001$), and this species did not share haplotypes among the three bores from which it was sampled. This demonstrates that even for the one species found at multiple bores, each bore represents a distinctive and isolated population.

This study has demonstrated high species diversity within the Parabathynellidae, but relatively low diversity of higher-order taxa in the lower Burdekin alluvial aquifer. While these species appear to be short-range endemics (i.e. restricted to a single bore), regional-scale sampling and repeated sampling of selected bores is needed to determine which of these taxa are truly short-range endemics. Pronounced population genetic subdivision was also found within the species demarcated by lineage P-A. This shows that alluvial aquifers can have strong barriers to dispersal by stygofauna. While stygofauna research is in it infancy in Queensland, this study has shown that stygofauna likely have strong long-standing barriers to connectivity in alluvial aquifers.

4.2.2 Groundwater microbial ecology

Introduction

Historically, characterisation of the microbial component of groundwater ecosystems has focused on assessing processes in contaminated systems meaning very little is known of microbial communities in pristine aquifers. Systematic studies in groundwater microbiology are required to develop an understanding of how microorganisms contribute to biogeochemical cycling and therefore, the food web and ecosystem maintenance. An understanding of microbial ecology is also necessary for a comprehensive understanding of groundwater biodiversity.

Methods

Microorganisms were collected from two groundwater samples taken from each of six bores in the upper, middle and lower reaches of the lower Burdekin floodplain . Samples were analysed targetting a selection of physiologically relevant groups of bacteria which are a function of the groundwater hydrochemistry and hydrology on the LBF (see Appendix F for full details).

Aquifer water quality was analysed at the six bores sampled for bacteria plus five other sites on the lower floodplain (Fig. 12) to provide context for the microbiological results. More detailed analysis of groundwater quality is presented in the hydrology report (McMahon et al. 2012). Sites were sampled between the 6th and 12th September 2011. Generally seven readings were made of standard water quality parameters (Fig. 26) at each bore using a Horiba water quality meter. Readings correspond to increasing volumes of water pumped up out of the bore ie 0-300 L. This ensured that the last reading was taken for water from the aquifer, rather than water from within the bore. Therefore, we have only referred to the last readings at each site (average of 250 and 300L sample results) or the last reading taken (bore 11910947: 200L reading only and bore 12001346: 250L reading only).

Results and discussion

The following is a summary of a more detailed report on microbes of the region that was commissioned for this study (see Appendix F for the full report).

Key findings were:

- All sites had high bacterial counts including microorganisms representative of surface water (allochthonous cyanobacteria). Aerobic, heterotrophic bacterial levels were similar to those found in contaminated or fouled organic-rich groundwater, and up to 5 orders-of-magnitude higher than those in pristine shallow unconsolidated aquifers. These results suggest that the groundwater aquifers are contaminated by high organic loads.

- The presence in all groundwater samples of cyanobacteria that require light for energy and CO_2 as a carbon source suggests there is widespread, dynamic surface and groundwater interaction. This is likely the result of managed aquifer recharge and deep drainage.
- The absence of faecal coliforms indicates that the groundwater sites sampled were not impacted by anthropogenic factors related to faecal contamination such as residential wastewater.
- The co-existence and ubiquity of both anaerobic as well as aerobic bacteria suggest the potential existence of micro-niches or physicochemical gradients such as redox zonation in the aquifers. The findings also suggest both aerobic and anaerobic respiration is of significance to organic matter mineralisation in the Burdekin aquifer system.
- Coastal groundwater contained actinomycete-like bacteria which are known for producing geosmins in potable water supplies, creating taste and odour problems.

Very little can be inferred about groundwater quality from a single sampling occasion. Basic variables are summarised in Figures 26 - 29. There was little variability in pH among sites (Fig. 28). Values ranged from 6.5 - 7.8 meaning they were generally within the DERM guidelines for protection of aquatic ecosystems (DERM 2009) but slightly more acidic than wetland results (Fig. 20d). This contrasts with other recent analysis which indicated pH of groundwater was below guidelines for the environment (McNeil and Raymond 2011).

In contrast, electrical conductivity (which is used as a surrogate for salinity) showed a high variation among sites (Fig. 27). Readings ranged from 251 - 16982 µs/cm which generally puts them above the guidelines for aquatic ecosystems which is 271 µs/cm for the Burdekin (DERM 2009). The sites with the highest salinity readings were in the upper part of the Burdekin Haughton Water Supply Scheme (Fig. 12). These results mirror more detailed groundwater assessments of the LBF that showed poor groundwater quality and excessive conductivity levels in particular (Barnes et al. 2005; McNeil and Raymond 2011). A number of theories have been proposed for high salinity in groundwaters further inland from the tidal zone (see McMahon et al. 2012).

Turbidity levels at the eleven sites showed a high degree of variability (Fig. 28) and varied from 9 - 3551 NTU. Four of the ten sites sampled had average turbidity readings above the guideline recommended for aquatic ecosystem protection (50 NTU - DERM 2009). Dissolved oxygen levels were generally low (in comparison with surface waters - Fig. 20) with readings varying from 0.8 - 4.5 mg/L but this may be a natural phenomenon for groundwaters.

These results broadly concur with the comprehensive analysis of groundwater done for the hydrology component of the project (McMahon et al. 2012) and the last State of the Environment report for the basin (McNeil and Raymond 2011) which noted water hard with occasional scale, electrical conductivity excessive for sensitive crops and environment, total nitrogen may also be excessive for environment or pH too low.

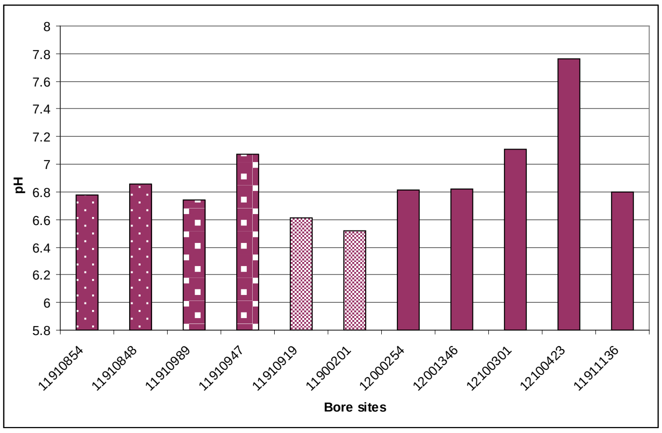


Figure 26 Average pH from 6 bores sampled for bacteria (patterned bars) and 5 other bores on the lower Burdekin floodplain. Upper, mid and lower flooplain bores are represented by different patterns from left to right.

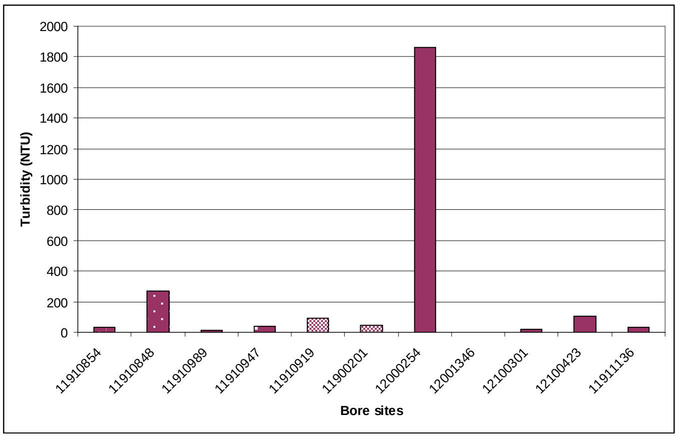


Figure 28 Average turbidity from 6 bores sampled for bacteria (patterned bars) and 5 other bores on the lower Burdekin floodplain. Upper, mid and lower flooplain bores are represented by different patterns from left to right.

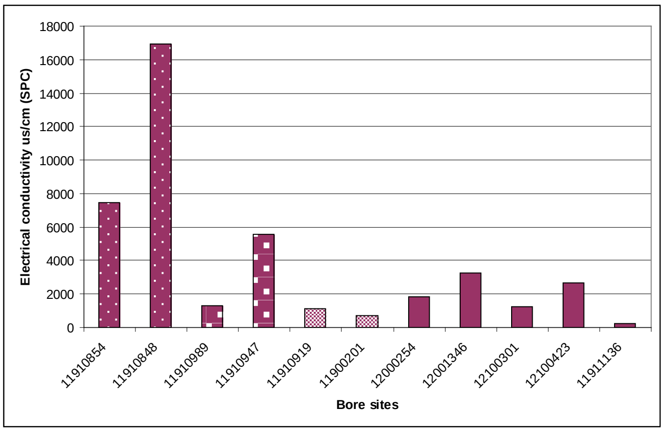


Figure 27 Average electrical conductivity from 6 bores sampled for bacteria (patterned bars) and 5 other bores on the lower Burdekin floodplain. Upper, mid and lower flooplain bores are represented by different patterns from left to right.

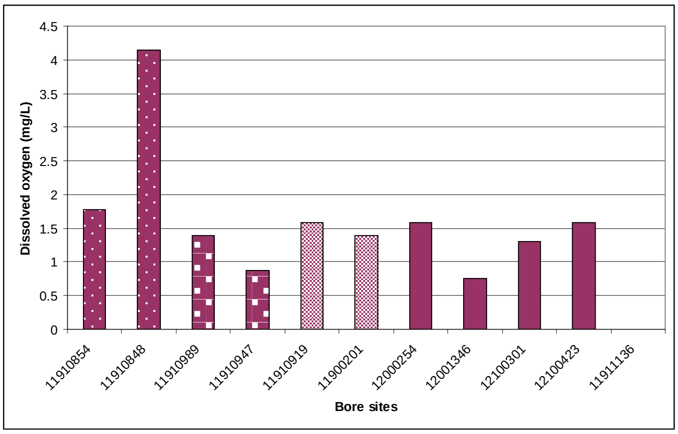


Figure 29 Figure 30 Average dissolved oxygen from 6 bores sampled for bacteria (patterned bars) and 5 other bores on the lower Burdekin floodplain. Upper, mid and lower flooplain bores are represented by different patterns from left to right.

5 Synthesis of findings: key hydrology - ecology relationships

5.1 Representation of flow and groundwater dependent ecosystem components and processes using ecological assets

Predicting potential ecological responses to altered flow and groundwater regimes is complex because of interactions between the hydrology and ecosystem components and processes at multiple scales. This is further compounded by confounding effects of non-hydrological stressors present in the system. Consequently general measures of ecological responses to managed surface and groundwater regimes are rarely observed (Kennard et al. 2010; Poff et al. 2010). A practical approach for managing hydrological regimes for specific ecological outcomes is to identify and partition the critical hydrological dependencies of ecosystem components and processes and consider their specific water requirements over space and time (Marshall and McGregor 2006). These components and processes are indicators of hydrological modification and are therefore broadly representative of the ecosystem response.

Known as ecological assets, they are highly valued components of the ecosystem for which aspects of the flow regime (i.e. duration, timing, variability, predictability, magnitude, rate of rise and fall) or groundwater regime (depth, pressure, quality) are critical to support their long term viability. Ecological assets are defined as: a species, a group of species, a biological function, an ecosystem or place of natural value for which water is known to be critical. The term 'critical' means water, and/or those conditions provided by specific flow or groundwater regime, is necessary to maintain the long-term viability of the asset. Ecological assets should occur naturally within the plan area and be linked to the ecological outcomes of the WRP. The process of ecological asset selection in the Burdekin WRP area is well underway, focussing on surface water linkages. Ecological asset selection is not part of this project; however the following section outlines information which will be useful to inform this asset selection process.

5.2 Ecological assets on the Burdekin floodplain

This report has highlighted that a range of groundwater dependent ecosystems exist on the lower Burdekin floodplain which are, within the limits of our current knowledge, all affected to varying degrees by the major changes in hydrology bought about by water resource development. The intensive agricultural development of the floodplain means that a range of threats exist in addition to changes in hydrology. We have identified potential ecological assets that we believe could be successfully monitored to assess the performance of water resource management over and above the other pressures on the floodplain. We have also focussed on those assets for which there is at least baseline information available.

Wetlands that were dependent on groundwater in their pre-developed state and wetlands that are now impacted by excess groundwater contain plants that may indicate relative wetland permanence. Two guilds of plants may be useful in understanding changes in wetland permanence; (i) plants that require full drying of wetlands (full drying guild), and (ii) plants that require permanent submersion (permanently wet guild). These guilds of plants are responsive to the hydrological changes of the LBF and can be valuable ecological assets in monitoring and understanding future changes in hydrology and how that may impact wetlands. With more data on the drying and depth variability requirements of theses two wetland plant guilds, it should be possible to model the ecological effects of alternative groundwater management scenarios. Such models could also be used to help devise solutions to the ecological problems associated with weed infestation. Establishing these guilds of plants, fish communities and wetlands as assets within the WRP may help to manage and model flows for ecological outcomes in the four wetland types we described in our conceptual models.

5.2.1 Wetland plants – drying guild (shallow non-permanent palustrine and shallow non-permanent estuarine wetlands)

Type of GDE: Surface water expression (wetlands)

Some wetland plant species require full drying of the wetland to complete their life cycles; usually with the reproductive stage occurring after drying (Cronk & Fennessy 2001). In the Burdekin, plants that may be in this guild include: *Eleocharis dulcis* and *Aponogeton queenslandicus* (most other *Aponogeton* spp. grow in permanent systems and are not in this guild). Both species are permanent that propagate by seeds and tubers. Seeds are set as water recedes. The tubers provide a way of rapidly re-establishing vegetative growth once water returns following the dry season, during which the above ground parts of the plants die off. This guild of plants has been heavily impacted by permanent water levels resulting from poor tailwater management and associated increase decrease in depth to water table. This is particularly so at the lower end of the system where small partially tidal basins were bare or sparsely covered by saltwater cooch and/or samphires during the dry and flooded with fresh water during the wet and supporting dense beds of *E. dulcis*. These are now permanently or semi-permanently flooded, saltwater is excluded by bunds and or groundwater pressure, and are dominated by *Typha* or worse, *Cabomba* and/or *Hymenachne*.

Value

Aponogeton queenslandicus is a wetland indicator species (listed as endangered in NSW, previously listed as Rare in Queensland). *Eleocharis dulcis* (commonly known as Bulkuru) beds are of great importance as nesting and feeding sites for waterbirds, particularly brolga and magpie geese which nest in the dense growth when the water is high, then feed on the tubers during the dry season when the swamps are dry (Figure 9).

Hydrological requirements

Populations of these plants can not be sustained in wetlands that are permanently inundated. This is most likely because a dry period is required to complete their reproductive cycle. *Aponogeton queenslandicus* does not occur in flowing streams, permanent water bodies or in water deeper than about 60 cm (Hellquist and Jacobs 1998). *Eleocharis dulcis* grows in dense stands in shallow seasonally flooded basins that dry out completely during the dry season. These may be marginally affected by tidal water and salinity will increase in the substrate as freshwater evaporates.

These plants are still found on the LBF but their distribution is limited to temporary wetlands on parts of the floodplain with limited flow modification. Most wetlands are now permanently inundated and no longer contain this guild of plant, thus reducing the diversity of individual wetlands and possibly creating other cascading ecological impacts. In permanent water, different species of plant have become dominant and in some cases alien plants have completely taken over.

5.2.2 Wetland plants – wetland guild (permanent riverine wetlands)

Type of GDE: Surface water expression (wetlands)

Plants that require constant submersion may have benefited or expanded their ranges on the LBF. Examples of species that require constant submersion include *Ceratophyllum demersum* and *Utricularia gibba*. *C. demserum* can grow to depths of over four metres and can flower and pollinate underwater or reproduce vegetatively (Aston 1973). *U. gibba* can also grow in water up to four metres deep as a rooted or floating plant (Aston 1973).

Therefore the transition of most wetlands on the LBF to permanent has benefited these plants. In some cases *C. demersum* may proliferate and choke a wetland or river reach.

Value

C. demersum is commonly known as the oxygen plant and is important when growing in sustainable densities for oxygen balance of the water column. *U. gibba* is known as bladder wart and provides habitat for small bodied native fish and would also contribute to oxygen content.

Hydrological requirements

The plants in this guild do not survive desiccation well and would become locally extinct if wetlands were to dry regularly, these plants are restricted to permanent waters.

5.2.3 Permanent riverine wetlands and fish community structure

Type of GDE: Surface water expression (river baseflow)

With the changes in water regimes, many wetlands have changed from non-permanent to permanent which has had a number of ecological impacts (see sections 2.3 and 2.4.2). Perna (2003) identified increased permanent flows as the major driver of decreased habitat quality and availability on the LBF, often indirectly through associated weed infestations, poor water quality and decreased biodiversity. We need to understand the links between hydrology and these impacts to effectively monitor wetland ecosystems on the LBF. Examining fish community structure may help establish benchmarks for water quality and habitat restoration goals within the WRP ecological outcomes. Through habitat restoration water quality and habitat condition may be improved therefore favouring a fish community rich in native diversity rather than dominated by alien fish species, indicating an overall improvement in ecological outcomes.

Once weeds have established on the water surface, water quality rapidly deteriorates, especially at the bottom end of the system where oxygen has been consumed by decomposing organic material. Perna and Burrows (2005) recorded sustained dissolved oxygen concentrations of less than 10% saturation effectively barring most native fish from migrating. Compounding this is the lack of open water where fish might move to extract more oxygen from the water/air interface. Physical barriers also exist where think stands of Typha have established in shallow standing water on the lower floodplain. Lastly once a waterbody is covered by weeds few native macrophytes remain as light is cut off from the water column. Many native fish species such as *Ambassis* spp. are closely associated with these plants and may become locally extinct without them (Perna and Pearson 2008).

Value

Fish communities contribute to the biodiversity of the region and also have recreational and commercial fishing values

Hydrological requirements

Some aspects of seasonality in water levels need to be restored through improved flow management. This needs to be done in conjunction with the control of alien plant infestations to create healthy habitats that favour native fish communities (Kennard et al. 2005; Perna et al. in press). The aim would be to transform the current fish communities that are dominated by alien fish and native species highly tolerant to poor water quality (Mosquito fish, Gudgeons, Tarpon, Tilapia, and Gourami) to a community more closely resembling the natural state including the more sensitive species of Bony bream, Archer fish and Longtom. Improving water quality in these wetlands will also directly benefit the groundwater by improving the quality of water entering the aquifer.

5.2.4 Specific wetlands with high conservation values

Types of GDEs: Surface water expression (wetlands), and phreatophytic vegetation

There are specific wetlands with high conservation values that should investigated and established as priority habitats for conservation and monitoring. These include the DIWA listed Barratta and Jerona Aggregations (Environment Australia 2001), Castalinalli's lagoon, Inkerman homestead lagoon, Didgeridoo complex, Healy's lagoon, Woodhouse lagoon, Upper floodplain wetlands including Swans and Strathalbyn lagoons and finally the non-permanent palustrine wetlands of Kelly's and Pelican Island. This represents an incomplete list of the highest quality wetlands left on the LBF. It covers the breadth of wetland types from permanent riverine wetlands to non-permanent palustrine wetlands. All provide specific ecosystem services that for the most part are gone from the rest of the LBF.

Value

Biodiversity (DIWA), water quality, habitat diversity and connectivity.

Hydrological requirements

The hydrological requirements vary according to the type of wetland; some require surface expression of groundwater (Healy's lagoon), and others require seasonal inundation and drying (Pelican Island wetlands). Most wetlands that have remained isolated from direct connection to the surface water distribution network are in the upper reaches of the floodplain, however there are also a few on the lower floodplain such as the Inkerman homestead lagoon. Each of these wetland types would need to be examined to assess their hydrological requirements.

5.2.5 Subterranean ecosystems

This study has provided some initial information about the fauna of the alluvial aquifer. However, we have little understanding of how subterranean communities respond to changes in groundwater quantity and quality and what values they represent (over and above biodiversity). For this reason we aren't recommending their inclusion as assets for LBF. There is a high likelihood, however, that these ecosystems are being affected by changes to groundwater in the LBF.

The results from bacterial sampling on the LBF indicate there is a strong influence of surface water on the subterranean bacterial community (Appendix F). This suggests that the flow through of water from the surface is happening at a high volume and that chemical changes in the water are not happening at similar rates to allow that water to become chemically typical of groundwater. How this may be affecting biota of the aquifer is unknown. Also unknown is how the quality of water entering the ground is affecting this biota.

High levels of nutrients are documented in the surface waters and given high volumes of this water are entering the aquifer it is likely nutrient loads are also high. Dissolved oxygen is generally low in surface waters on the LBF so this may also being translated in aquifer water quality. This deterioration in aquifer water may stress the biotic communities and some taxa may have already become locally rare or extinct. Changes in water levels may also be negatively impacting subterranean ecosystems. In section 3 we showed a generally increase in water table levels which may be linked to the increased influence of surface water in the subterranean ecosystem. Another impact may be the increased frequency and speed with which water levels rise and fall (Fig. 14). Stygofauna are not highly mobile and if water levels are rising and falling at more rapid rates and more often this may cause local extinctions due to desiccation if the animals cannot move fast enough to follow the falling water levels.

5.2.6 Phreatophytic vegetation

We did not examine impacts to phreatophytic vegetation of rising water tables however we strongly suggest that these GDEs are used as assets in the revised WRP. This is because of the limited quantity of remaining vegetation patches on the floodplain and the likelihood that these ecosystems are being negatively impacted by changes in the groundwater hydrology. To do this would require some separation of impacts as mentioned in section 2.2.2 from the potential impacts of waterlogging. Specific areas that should be considered are the Barratta Creek corridor and specific wetlands identified by Brizga et al. 2006.

6 Knowledge gaps

6.1 Knowledge gaps and recommendations for future research

We focussed on four key wetland types to investigate groundwater related issues through sampling, conceptualisation and analysis of historical data. Below is a list of recommendations and knowledge gaps that if filled will inform the amendment of the Burdekin WRP to incorporate groundwater. This was generated based on the findings of this project and the recommendations of an expert panel (Appendix A).

1. Document the hydrological requirements for wetland plants (drying guild)

Aponogeton queenslandicus and *Eleocharis dulcis* do not occur in permanent wetlands so it is assumed that they need to be dry every year. The timing and duration of drying required for these species is currently unknown. These two aspects would need to be investigated to be able to effectively model in relation to flow. This could potentially be done using manipulated experimental conditions.

2. Document the hydrological requirements for wetland plants (wetland guild)

We need to know the duration and frequency of drying in a wetland that causes plants such as *Ceratophyllum demersum* and *Utricularia gibba* to die off. In other words what are the thresholds of concern for water level fluctuation in regard to this plant guild. This could potentially be done using manipulated experimental conditions.

3. Permanent riverine wetlands and fish community structure

There are several knowledge gaps relating to how water management affects aspects of fish habitat quality:

a) What are the limits of seasonal water level variation before causing major water quality decrease? This could be determined by using pumps and the distribution networks to artificially manipulate flows while measuring changes in water quality with meters and data loggers.

b) What is the optimal timing of flows to provide best improvement in water quality and possibly weed control? This needs to be done in the context of land management activities that may negatively impact water quality For example monitor water quality at harvest times when there is a drop in irrigation water demand, increase in temperature and rain showers causing runoff of cane trash (generally October - December).

c) Can weeds be controlled using turbid river water? Experimentation at some sites could determine the effectiveness of this approach. For example by removing weeds systematically from the top of the system to the bottom, is turbidity maintained further down the system and does this aid in controlling the growth of submerged macrophytes that can cause oxygen crashes overnight through respiration?

4. Specific wetlands with high conservation values

A systematic review of current knowledge is required as a first step to identifying priority wetlands. Then we need to determine what level of seasonality would be required to be maintained in these wetlands to retain habitat values. Operationally this means identifying where regulated flow needs to be diverted away from a wetland or into a wetland to maintain environmental integrity.

One approach may be comparison of physically similar lagoons with different current management regime and different ecological values. For example Inkerman Homestead and Munro's Lagoon. In this case Munro's lagoon receives river water, has high water hyacinth cover, low diel oxygen cycling and low fish diversity. Inkerman homestead on the other hand does not receive river water, does dry back somewhat seasonally, has no hyacinth, maintains healthy diel oxygen cycles and has a diverse native fish community. Comparing these sites in more detail may reveal what some of the structuring forces are and how we may be able to manipulate them in Munro's for better ecological outcomes.

5. Determine the contribution of different water sources to the flows in non-permanent riverine wetlands such as Barratta Creek.

Quantify the relative contribution of water from different sources including tailwater from cane drills, over flow from both distribution and tailwater channels, inflows from the aquifer and natural stream flow. One approach may be to develop a water budget for the system based on field measurements and models that estimates inputs, pathways and outputs as has been done for the aquifer (McMahon et al 2012). Once this has occurred management approaches can be introduced to attempt to ameliorate the negative impacts of excess water. However, current supplemented flows dilute the high nutrient loads and biological oxygen demands associated with changed land use and this needs to be taken into account in flow management. This means that monitoring and adaptive management need to be implemented if any changes to flows are planned to avoid the negative impacts of both persistent flow and low flows in these systems.

6. Develop a better understanding of the stressors impacting on phreatophytic vegetation

Much of the remaining non-riparian terrestrial floodplain vegetation in the LBF occurs in the Barratta Creek catchment. This vegetation currently appears in poor condition and may be experiencing waterlogging from continuously high groundwater levels. However, vegetation is likely being negatively impacted by an insidious suite of stressors such as grazing, fire, woody weeds and herbicide spray drift (Tait and Veitch 2007). A better understanding of how this suite of stressors are impacting phreatophytic vegetation fragments would facilitate the development of a holistic management strategy that may include groundwater management. Analysis of current groundwater levels in the context of historical pre-development levels would be an essential first step and the groundwater model being produced in the other component of this project will provide the tool to do so.

7. Develop a better understanding of stygofauna communities and drivers of their diversity and distribution

We now know stygofauna exist in the lower Burdekin aquifer so we recommend further investigation of subterranean ecosystems to describe diversity, range and regional context of the species present. Further research should also establish the appropriate sampling protocols to address particular questions, for example sampling techniques to map biodiversity will likely be different than those to assess impacts. From this foundational knowledge key questions can be addressed such as how changes in water table and ground water quality affect the diversity and function of groundwater ecosystems. Further clarification is needed on the role stygofauna and bacteria play in the aquifer outside of just biodiversity rather linking them to ecosystem services such as; nutrient cycling, hydraulic connectivity or pesticide breakdown. Use of stable isotope in studies to better understand the food webs within subterranean ecosystems may be one approach to this.

8. Describe and map estuarine and near shore marine ecosystems

Groundwater upwellings are known along the Great Barrier Reef and off the coast of the Burdekin catchment, however their ecological function, fauna and flora are not well documented. Anecdotal evidence suggests they may be important for fish production as they are favourite fishing spots. Also unknown is how they are affected by changes to aquifer discharge. Water quality monitoring of selected wonky holes and baited underwater video to examine fauna are possible approaches to understanding these ecosystems better. Such techniques may reveal if fauna changes are associated with changes in aquifer discharge through the year.

9. Consider and investigate the implications of sea level rise due to climate change for saltwater intrusion

It is likely sea-level rise will become a significant pressure in the future. The scale of this potential threat needs to be assessed so it can be effectively planned for. For example modelling of potential sea level rises to investigate the distance inland which may be affected by saltwater intrusion. The new groundwater model (McMahon et al. 2012) will be able to address these questions.

6.2 Recommendations for groundwater management

The material presented in this report describes an interwoven suite of stressors impacting on the groundwater dependent ecosystems of the LBF. Any changes in groundwater management must be considered in the context of the other ongoing pressures that exist on the floodplain. For this reason we do not advocate a groundwater management framework that attempts to return the floodplain and its GDEs to the pre-development or "natural" state described in our conceptual models (Figs 6 - 9). Instead we propose changes that would result somewhere between the two states described in the conceptual models.

This idealised future water management scenario would include a range of approaches to maintain ecosystem integrity (Fig. 30). Regulated surface flows could be used to reduce low oxygen "black water" events during the early wet season when irrigation needs drop off, this could be achieved by maintaining flow during typical fish kill periods. This measure, incorporated with mechanical weed removal to ameliorate some of the water quality and connectivity issues throughout the distribution streams on the LBF, would benefit fish species currently unable to survive low oxygen conditions. Some wetlands that have not been connected to the surface water distribution schemes and have not had the weed or water quality issues seen in other lagoons (e.g. Castelanelli's and Inkerman Homestead) this suggests that keeping supplemental flows out of wetlands may be of benefit.. Bunding and levee banks could be used to protect more wetlands from supplemental flows. This would reduce the incursion of Burdekin river irrigation water and promote some seasonality in water levels, promoting guilds of wetland plants that need drying as part of their lifecycle. Installation of tidal gates at the bottom end of the system to allow some saltwater inundation would reintroduce some tidal influence promoting fish migration and reducing weed infestation in shallow estuarine wetlands.

Specific actions that may directly impact the aquifer levels are depicted in Figure 30 and include:

- Improved water delivery and recycling within the irrigation areas
- Decreased application of gypsum on the clay soils which increases water transport to the aquifer
- Increased use of water from the aquifer for irrigation
- Removal of drop boards allowing water levels to drop in wetlands and streams
- Examining alternatives to furrow irrigation

Our specific recommendations are:

1. Re-introduce seasonality to flow regimes

Our results and the synthesis of previous findings on wetlands generally suggest the loss of seasonality in water levels is a key driver of a range of impacts. It is strongly recommended to examine how to re-establish some seasonality and inter-annual variability in wetland water levels. The end point is not to return to 'natural' but to establish some water level fluctuation which may lead to controlling some alien weeds as well as help re-establish some native flora and fauna. Until we understand the relative contributions from surface waters and groundwater it is difficult to say how this may be most effectively achieved, however the most likely approach will be through surface water management which is directly linked to all the impacts identified throughout this report. Links between the new groundwater model (McMahon et al. 2012) and the surface water hydrological models can help tease this apart and then inform management for ecological outcomes.

2. Include Barratta and other distributary streams in the Resource Operations Plan (ROP)

The results presented in this report show Barratta Creek and other distributary streams are being negatively impacted by current water management practices (in combination with other pressures). While the health of Baratta Creek is already an outcome under the existing WRP, it is not part of the operational plan. Inclusion would allow better understanding and management of Barratta Creek through allocation of water modelling nodes and development of water sharing rules and ecological outcomes.

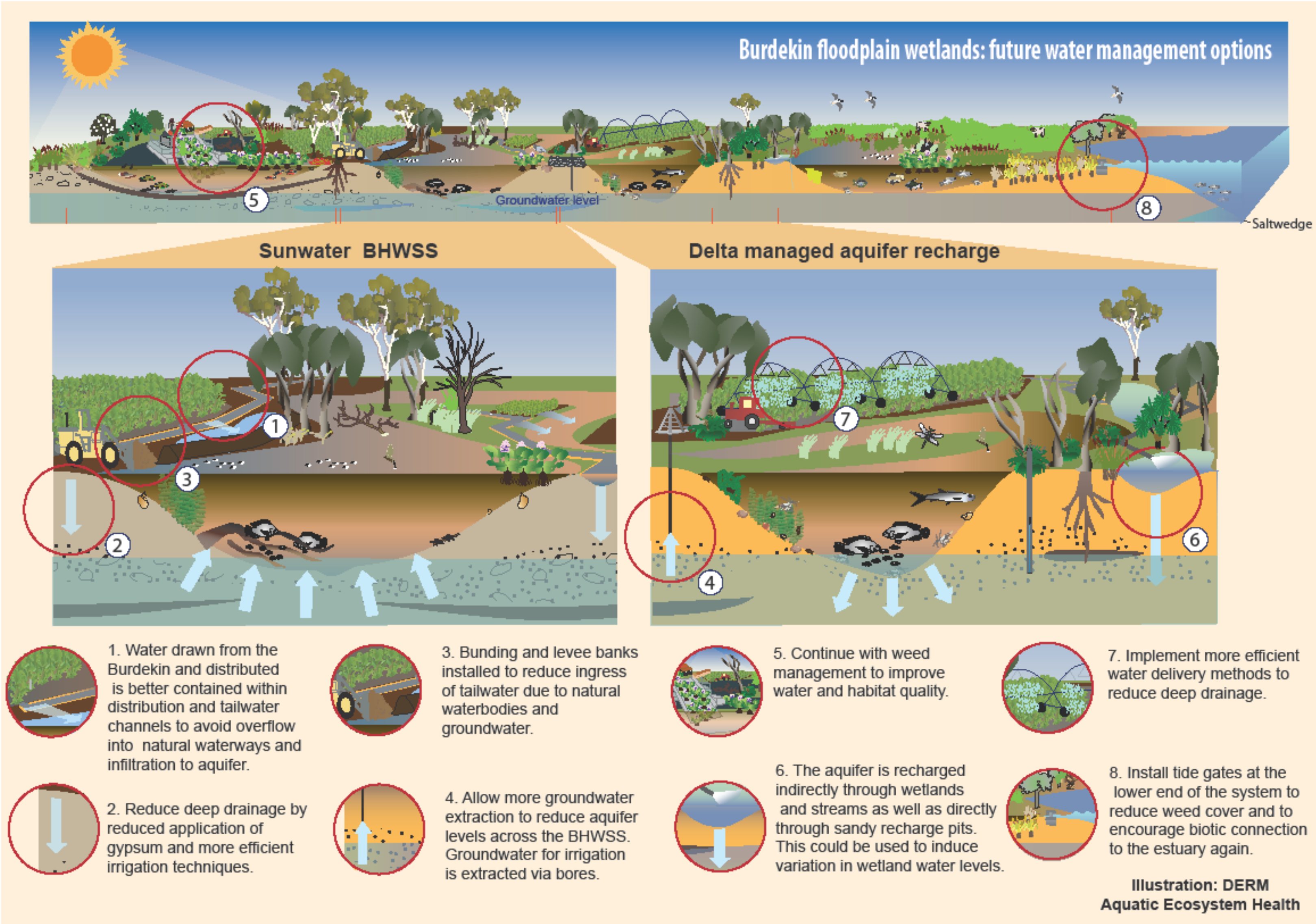


Figure 30 Burdekin floodplain wetlands: future water management options.

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Appendix A Details of expert panel workshop to review conceptual models and management recommendations

An expert panel was convened to provide feedback on draft conceptual models and report recommendations. The stated aims of the workshop were:

1. To review conceptual models of Burdekin floodplain wetlands
2. To review a draft set of recommendations for the amended Water Resource Plan

The workshop was held on January 25th 2012 in Townsville. Attendees are detailed in Table 1. After giving some background information about the project and the aims of workshop, participants were asked to give feedback on draft versions of the conceptual models. Discussion was captured on a whiteboard so participants could ensure their comments were captured accurately. Comments arising that weren't of direct relevance to the conceptual models were recorded on butchers paper for later follow-up or incorporation into the management recommendations.

Name	Organisation	Role/speciality
Bob Bennett	DERM	groundwater hydrologist
Amy Becke	DERM	hydrologist
Moya Tomlinson	DERM	groundwater ecology
Damien Burrows	James Cook University	tropical freshwater ecology
Charlie Papale	Lower Burdekin Water	Lower Burdekin water management
David Brooks	Lower Burdekin Water	Lower Burdekin water management
Peter Hancock	Eco Logical Australia Pty Ltd	groundwater ecology
Jim Tait	Econcern	Freshwater ecology (fish)
Stefan Eberhard	Subterranean Ecology Pty Ltd	subterranean fauna,
Diana O'Donnell	North Queensland Dry Tropics NRM Group	sustainable coasts/ healthy waterways
Jason Williams	DERM	irrigation management
Niall Connolly	DERM	Freshwater ecology / local water management
Anthony Morrison	DERM Townsville	wetlands/ botany
Chris Kahler	DERM Townsville	wetlands/ botany
Colton Perna	DERM	Project representative
Ruth O'Connor	DERM	Project representative & facilitator
Peter Negus	DERM	Project representative

Note: We subsequently met with Mike Ronan from the Queensland Wetlands Program (DERM)

Summary of comments pertaining to conceptual models

Generic comments

Modifications to incorporate models into the WetlandInfo website:

- Convert a satellite or lidar image into an overall real life representation of the Burdekin floodplain then have the zoom on the 4 selected wetland types bringing them into reference with what the area looks like.
- Include dynamic nature of GW level prior to development and show flow direction, eg mounding, capillary function

Consider re-naming of wetlands to make consistent with the Qld wetlands program classification:

- Shallow non-permanent wetland to "Shallow non-permanent palustrine wetland" (e.g. Pelican Island wetlands)
- Barratta Creek to "Non-permanent riverine wetland" (e.g. Barratta Creek)
- Deep permanent lagoon to "Deep permanent riverine wetland" (e.g. Dick Bank Lagoon)
- Shallow coastal wetland to "Shallow estuarine wetland" (e.g. The Lake)

- One suggestion to have the wet dry phases of each wetland type on the same page – side by side to highlight changes. Need to think how this affects the panorama (perhaps presented on a separate page?)
- To highlight how basic hydrological processes differ (between wetlands, seasons and post development), consider developing basic box and arrow hydrological models. Might be covered by the hydrological components of the project
- Have the models animated for display on wetland info
- Create pamphlet style report similar to Fitzroy for distribution to stakeholders on request
- Diagrams quite complex for those unfamiliar with the area. Should look to highlight the key processes better:
    - Maybe consider animations to highlight dynamism (e.g. wet to dry transition)
    - Key processes labelled in the diagram?
    - Exclude elements that aren’t essential to simplify
    - Magnifiers help (possibly code these so each element/process has its own shape that is consistent across diagrams

Wet season panorama diagram

- Should slope down a bit from left to right (needs to be applied to all panoramas)
- Groundwater levels (written in) need to be higher than the stream levels. Needs to possibly be more bell shaped showing mounding and capillary function
- Represent the dynamism of groundwater flows – maybe add dotted lines to diagram?
- Ask Ged for information on flow directionality
- On overall diagram need to separate 2 and 3

1. Pre-development wet season conditions in shallow non-permanent palustrine wetlands

To be noted in text

- non permanent wetlands can be highly productive

- There is an overall dynamism between whether the groundwater is flowing into or out of wetlands and distributary streams. This is linked to antecedent conditions (among other things)

Changes to conceptual models

- Change title to "1. Shallow non-permanent palustrine wetland (e.g. Pelican Island wetlands)"
- Arrows to be removed
- Groundwater level lowered
- Clay to expand underlaying all of the wetland

2. Pre-development wet season conditions in Non permanent riverine wetlands

To be noted in text

- Barratta Creek is an important example of distributary streams. Broad ecological processes are similar among distributary streams but specific differences arise between streams such as the Barratta on a clay layer versus Deep permanent riverine wetlands that sit on top of younger alluvium (e.g. Sheep station, Plantation and Saltwater creeks)
- Include a definition of distributary stream (geomorphic reference).
- May need to make the distinction between tributary distributary creeks with examples

Changes to conceptual models

- Change title to “2. Non-permanent riverine wetland (e.g. Barratta Creek)
- Show hyporheic flow in the wet season (ceases in dry) - inflows and outflows through point bars which can change the water quality in-stream
- Include stygofauna in diagram
- Groundwater extended into riparian lens - more in wet less in dry
- Change to magnifier text “… clears after the monsoon” to “clears after high flows”

3. Pre-development dry season conditions in Shallow non-permanent palustrine wetlands.

To be noted in text

- For phreatophytes, even if the roots are no longer in the saturated zone, they are still groundwater dependent. In the dry season they often use water from the capillary zone

Changes to conceptual models

- Change title to "1. Shallow non-permanent palustrine wetland (e.g. Pelican Island wetlands)"
- The clay layer (dark brown) should underlay the entire wetland
- Some trees without leaves in dry (especially Leichardt trees as they are deciduous)
- Remove the no mixing symbol from the water quality symbols

4. Pre-development dry season conditions in Non-permanent riverine wetland (e.g. Barratta Ck.).

To be noted in text

- Hyporheic flows become less and cease during dry seasons. Also very little in the clay bed areas
- There is a big difference in water quality during dry season between distributary streams (non permanent riverine wetlands) and deep permanent riverine wetlands. Wetlands on the older clays (Barratta) would have

high organic loads and high levels of tannins, while deep permanent wetlands on delta distributary streams would be clear from groundwater flowing through sandy soils.

- Groundwater would not be supplying running flows but would have maintained waterholes and their levels. Groundwater contribution in the pre-development stage would have also been highly influenced by native vegetation cover.
- Aquatic plants proliferate on the exposed bank where not shaded by riparian vegetation
- Catchment location will influence the degree of groundwater connection with less in the upper catchment
- Need to note that there are different land systems naturally across the floodplain that many of the wetlands cross and this is what brings in much of the variability to the system. In particular wetlands in the delta vs floodplain alluvials are different (see Christensons 1963)
- Brief description of water quality (talk about what it is like post development but what has it changed from?). Possibly: clear, low nutrients, pulses of organic matter, variability in groundwater contribution

Changes to conceptual models

- Change title to “2. Non-permanent riverine wetland (e.g. Barratta Creek)
- Location in the catchment is an important factor for groundwater dynamics, for example in upstream areas creeks not linked to groundwater in the dry season. Opposite in downstream sections due to soil types and depth of stream bed/height of water table.
- Include stygofauna in diagram
- Groundwater extended into riparian lens – more in wet less in dry
- Ecological processes occurring in the dry stream banks should be shown (for consistency with non-permanent wetlands)
- Try and highlight in the diagram that this is a series of pools in dry (cracked mud?)
- Some trees without leaves in dry (especially Leichardt trees as they are deciduous)
- Move clay layer underneath pool, water table lower and underneath

5. Pre-development wet season conditions in deep permanent riverine wetland

To be noted in text

Changes to conceptual models

- Change title to "3. Deep permanent riverine wetland (e.g. Dick Bank Lagoon)"
- More submerged vegetation shown in diagram in wet and dry
- Groundwater arrows can go either way (more uni-directional in dry)
- Make more of a contrast in the depth profile between this and the shallow costal wetland ie deep lagoon should look noticeably deeper than the coastal wetland which has a flatter profile
- For both shallow coastal and deep lagoons should note somehow that we are depicting the end of the wet season as this is when the groundwater becomes important
- Raise watertable so tree roots are well within it
- Need to check consistency in colours used to depict different types of water eg surface water vs water seen through the cross section, etc. May clarify if transitions between water types in cross sections (e.g. turbid to clear) aren’t depicted as distinct lines but rather gradations

6. Pre-development wet season conditions in shallow estuarine wetlands

To be noted in text

- Tidal connections will cause water level fluctuations
- Key point overall is that the balance between groundwater contribution and tidal influence keeps these systems brackish
- Productivity hotspot attracting waterfowl
- Nursery habitat for fish

Changes to conceptual models

- Change title to "4. Shallow estuarine wetland (e.g. The Lake)"
- For both shallow coastal and deep lagoons should note somehow that we are depicting the end of the wet season as this is when the groundwater becomes important
- Make more of a contrast in the depth profile between this and the deep lagoon ie deep lagoon should look noticeably deeper than the coastal wetland which has a flatter profile
- Include stygofauna in diagram
- Arrows going up from groundwater need to be changed to go down up ie the wetland is losing water (arrow styles should indicate high vs low hydraulic conductance)
- The sliver above the clay pan should be saturated (grey colour)
- Include magpie geese foraging
- Change text for sedges magnifier: remove “Once the rains …proliferate” and add to the dry season diagram.
- Salt wedge maybe needs be included here as well as the panorama
- In all individual diagrams – watertable needs to be lensed or mounded instead of flat

7. Pre-development dry season conditions in deep permanent Riverine wetlands

To be noted in text

Changes to conceptual models

- Change title to "3. Deep permanent riverine wetland (e.g. Dick Bank Lagoon)"
- Some trees without leaves in dry (especially Leichardt trees as they are deciduous)
- More submerged vegetation shown in diagram in wet and dry
- Make more of a contrast in the depth profile between this and the shallow costal wetland ie deep lagoon should look noticeably deeper than the coastal wetland which has a flatter profile
- Water table level should be the same as water level in the wetland
- Circle and magnify the small waterhole in the background – this is an example of an non-permanent wetland on sandy soil with higher connectivity

8. Pre-development dry season conditions in shallow estuarine wetland.

To be noted in text

- Tidal inflows are keeping these wetlands wet during the dry season
- Most of these wetlands have tidal ingress but not all

Changes to conceptual models

- Change title to "4. Shallow estuarine wetland (e.g. The Lake)"
- Make more of a contrast in the depth profile between this and the shallow estuarine wetland ie deep permanent riverine wetland should look noticeably deeper than the estuarine wetland which has a flatter profile
- Include stygofauna in diagram
- Include samphire plants in the saltbasin
- Sedges should be depicted as dead stems lying flat. Magnifier saying "Dead plants provide a rich food source for a boom in productivity when water levels rise again"
- Currently have a pandanus, should be a mangrove but not Rhizofer – one with pneumatophores (roots poking up)
- Some trees without leaves in dry (especially Leichardt trees as they are deciduous)
- Clay layer should extend completely underneath wetland
- Need to illustrate tidal inflows that influence wetland depth
- Change text in magnifier from "water is saline and marine fish species predominate" to "Water is brackish and marine fish species can predominate"
- Typo in magnifier: change *Ambasis* to *Ambassis*

9. Water management in the Burdekin floodplain

To be noted in text

- Definitions – we need to explain difference between distribution channels and distributary streams.
- While the delta predominantly uses pre-existing streams to distribute water these can no longer be considered "natural streams" now "regulated" streams
- MAR predominantly uses streams for distribution (but some constructed channels) and vice versa: BHWSS predominantly uses constructed channels abut also some streams including the Haughton River
- MAR has higher proportion of exotic fish
- Photos may help to illustrate difference
- Add in discussion groundwater vs surface water allocation is used as a management tool

Changes to conceptual models

- Better show sandy alluvial in the delta MAR vs clays in BHWSS

a) Sunwater BHWSS

- Burdekin river water normally of higher turbidity
- Need to show tailwater going back into streams on the diagram
    - Show all sources of tailwater. Through bottom of channels, over flow, deep drainage
    - Both distribution channels and tailwater channels lose to groundwater
- Show use of gypsum increases water loss to aquifer increasing hydrological flow through older clays
- Show supply channels and tailwater drains losing water. They also lose water through channel overflow
- More native and catadromous fish need to be included
- In magnifier text (2nd from left at bottom) replace "Barratta" with "distributary streams"
- First magnifier: insert the word "River" after "Burdekin"

- Add Hymenachne (exotic plant)
- Less hyacinth (mostly in Delta)

b) Delta managed aquifer recharge

- Change spray to furrow irrigation
- Change text next to magnifier from “groundwater level is monitored” to “The delta is managed according to groundwater level”. Currently the diagram here is of tree roots but this should be a piezometer
- Add text in magnifier: Water is extracted from the Burdekin River and pumped into distributary streams
- Channels are generally constructed in sand to maximize groundwater recharge
- Change magnifier text: from “groundwater is indirectly topped up from the wetland” to “groundwater is recharged from the wetland”
- Delta has much less catadromous but should be more exotics while Barratta may not have Tilapia – ie Barratta has better fish
- Bottom right magnifier text “…..water piped overland from the Burdekin River”
- Add water hyacinth and Hymenachne

10. Post-development conditions in shallow non-permanent palustrine wetlands.

To be noted in text

- Water may be flowing into or out of wetlands into the groundwater depending on their location in the landscape.
- Water present during the dry season comes from tailwater and groundwater
- Phreaophytic vegetation is also adversely affected by non-hydrological issues such as fire
- While most non-permanent wetlands have become more permanent, some such as Gilgai and Green Swamp are drying up because constructed drainage infrastructure leaves them cut-off except during high flow events
- The outflows of the shallow non-permanent wetlands ie black waters – can create impacts to the stream ecosystems – eg barrier to migration of catadromous fish
- Stygofauna present but diversity may have been negatively impacted by static watertables, anoxia or hypoxia and increased levels of nutrients

Changes to conceptual models

- Change title to "1. Shallow non-permanent palustrine wetland (e.g. Pelican Island wetlands)"
- The clay layer (dark brown) should underlay the entire wetland
- Change text in magnifier 2nd from left at bottom: “Wetland is permanently full of pumped water characterised…” to “Wetland is permanently full of water and is characterised…” Then “…The water is still and black.” to “…The water is still and dark”
- Include a tailwater drain entering the wetland
- Add arrows going both ways from the groundwater
- Other aquatic weeds include Hymenachne, Aleman grass and Para grass, reduce amount of hyacinth
- Include stygofauna Water quality in distributary streams has changed – generally more turbid water for longer periods than pre-development. More colloidal material and associated nutrients originating from the Burdekin Falls Dam and agriculture. Tailwater also contains inorganic nutrients

11. Post-development conditions in Non-permanent riverine wetland.

To be noted in text

- Anoxic water mobilises nutrients
- Surface water is having a greater contribution to distributary creeks but the relative contribution with groundwater is unknown
- “Stygofauna present but diversity may have been negatively impacted by static watertables, anoxia or hypoxia and increased levels of nutrients”
- Weed chokes restrict water flow, block light passage through the water column and cause loss of submerged macrophytes. Weeds also add to the fine organic matter load

Changes to conceptual models

- Change title to “2. Non-permanent riverine wetland (e.g. Barratta Creek)
- Change text in bottom right magnifier: from “…Water no longer flows.” To “…Water no longer flows freely.”
- Groundwater level needs to be depicted more as a cone
- Need to increase fish diversity (add tarpon, gudgeon, barramundi)
- Include stygofauna
- Weed chokes restrict water flow, block light passage through the water column and cause loss of submerged macrophytes. Weeds also add to the fine organic matter load
- Show hyporheic flows

Post development 3 and 4 panorama

- Move saltwater wedge further inland

12. Post-development conditions in deep permanent riverine wetland.

To be noted in text

- Note that some of these waterbodies are still in reasonable condition and these may be a priority for conservation
- In the pre-development scenario organic material from macrophytes was left to dry on the banks and much was taken out of the system. Now with constantly high water levels more organic matter stays in the system creating flows of anoxic black water in early wet season

Changes to conceptual models

- Change title to "3. Deep permanent riverine wetland (e.g. Dick Bank Lagoon)"
- Make more of a contrast in the depth profile between this and the shallow estuarine wetland ie deep lagoon should look noticeably deeper than the coastal wetland which has a flatter profile
- Water quality symbols that appear in the shallow non-permanent wetland should also appear here. Need some description of water quality, show how water from the Burdekin flows through in “tube” not mixing
- Include management actions required to maintain open water rather than weed infestation - harvesters
- Macroinvertebrates and stygofauna need representation – what impacts on diversity and distributions

13. Post-development conditions in shallow estuarine wetlands.

To be noted in text

- Need to define palustrine wetlands. More self-explanatory text for diagram might be “freshwater vegetated swamp” Describe how hydrological modification has altered these from estuarine to palustrine
- Lower productivity in terms of boom and bust species such as the microcrustacea which could have flow-on effects in the food web such as reduced waterfowl numbers

Changes to conceptual models

- Change title to "4. Shallow estuarine wetland (e.g. The Lake)"
- Make more of a contrast in the depth profile between this and the shallow costal wetland ie deep lagoon should look noticeably deeper than the coastal wetland which has a flatter profile
- Bund wall (magnified at bottom right) needs to look more like a bund wall. Have it curved and with one side obviously fresh and one side salt
- Need to show the saltwater wedge in diagram 4 (post development)
- Change text of magnifier at bottom right: add “This ponds freshwater on once tidally influenced salt pans.”
- Macroinvertebrates and stygofauna need representation – what impacts on diversity and distributions

Recommendations

Shallow non-permanent palustrine wetlands (e.g. Pelican Island wetlands)

1. Attempt to increase seasonal variation in water levels without aim of return to natural:
    - Reintroduction of some seasonality in water levels may be most effectively achieved through changed management of surface water rather than groundwater
    - Investigate and promote options for reducing the dumping of tailwater into these ecosystems. This could include tailwater re-use and recycling.
    - It was noted that a number of farmers already do this
    - Bunding some wetland areas may restore seasonality by reducing connectivity e.g. downstream of Bruce Hwy, Lochinvar

2. Monitor impacts of increased seasonality in water levels to prevent negative impacts

Non-permanent riverine wetlands (e.g. Barratta Creek)

1. Determine volume of water in streams from ground water versus surface runoff
    - Also what proportion is from tailwater?
    - How much water is being recycled?
    - Need this information to determine whether it is better to increase bore water or surface water pumping to deliver ecological outcomes
    - Determine what is the minimum amount of water needed in the Barratta to keep it healthy
    - Put a gauging station and node on sheep station creek, look to get DERM accreditation for existing gauging stations

2. Seek to increase seasonality and inter-annual variability in stream flow
    - Use regulation structures to direct flows down different systems in different years
    - Reduce channel overflows and tailwater into Barratta Creek
    - Promote groundwater extraction as was planned as part of the irrigation system in the Barratta

3. Include Barratta and other distributary streams in the Resource Operations Plan (ROP)
    - Needs water modelling nodes allocated. Following from this would come water sharing rules and ecological outcomes
    - Need a monitoring framework that directly assess flow impacts (current ROP doesn't include indicators of ecological condition)

4. Mitigate blackwater flow events
    - Run artificial flows through the distributary streams to flush

5. Reduce fresh water flows especially to the estuary
    - Consider issuing water permits for use of surface water (usual only for 1 year duration and then reapply)

6. Condition assessment of vegetation corridors along Barratta ck.
    - Define impacts, fire, high ground water, pesticide, fragmentation

7. Remove weed chokes
    - This is not likely to be part of the WRP but through other management mechanisms

Deep permanent riverine wetlands (e.g. Dick Bank Lagoon)

1. Modify flows to reduce occurrence of fish kills in the early wet season
    - The mechanisms of fish kills need further investigation
    - This is particularly relevant where the wet season was small
    - Pumping regimes to dilute blackwater events should be investigated

2. Increase seasonality and inter-annual variability in water levels
    - This could be difficulty due to clashes between optimal drawdown times and water supply demands
    - Investigate bunding to direct flows for ecological outcomes

3. Target some high value deep lagoons to reduce inflows
    - Lagoons on the periphery of the regulated network and with minimal current impact (such as Inkerman, Swanns and Woodhouse) may be most feasible for management actions
    - Minimize inflows through infrastructure devices

4. Use turbid river water to control in-stream plant growth
    - This approach relies on source water from the Burdekin being turbid which it isn't always
    - Another limitation is it only works for submerged weeds

Shallow estuarine wetlands (e.g. The Lake)

1. Trial removal of bunds to control weeds
   - Salt levels will not always remove typha – combined with spraying and drying and burning may be a good approach
   - Issue of accountability for any loss of agricultural productivity
   - May not be considered appropriate or applicable by water planners

2. Create some seasonality in levels without altering saltwater intrusion to production areas
   - Construct tidal gates so that the freshwater can get out of the wetlands but saltwater can't get in – could be beneficial for drying out of the system

3. Improve connection/migration between estuary and stream/wetlands
   - Use of temporary diversion walls at appropriate locations to increase connectivity for fish

4. Consider and investigate the implications of sea level rise due to climate change for saltwater intrusion

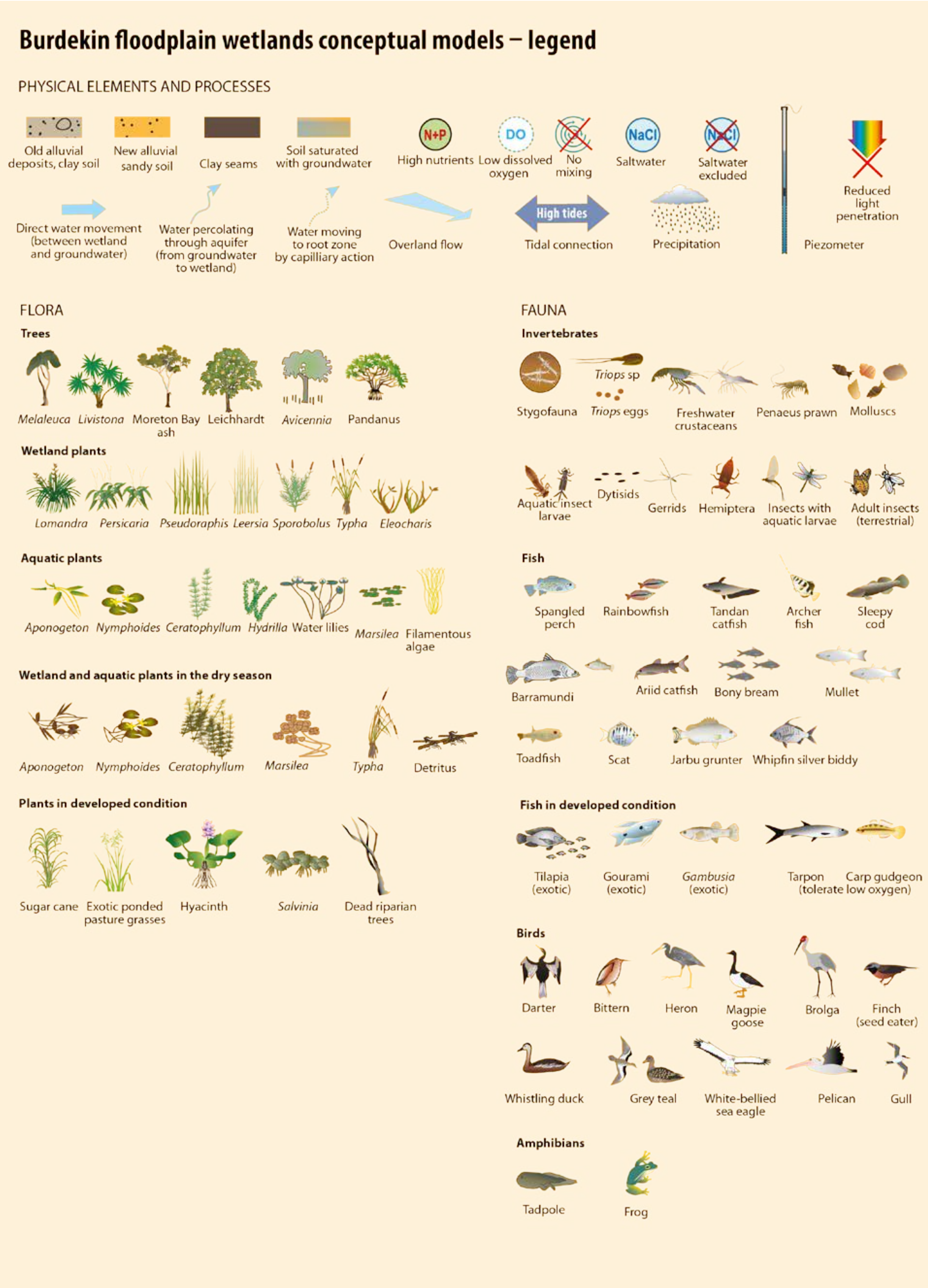
Subterranean ecosystems

- Before subterranean ecosystems can be considered in a WRP context, a hypothesis linking groundwater level to stygofauna communities needs to be tested.
- The value of stygofauna communities and their role in groundwater ecosystems also needs clarification. This could be biodiversity value or ecosystem services such as nutrient cycling, pesticide breakdown or hydraulic conductivity. If this is shown they could be treated as an asset
- Comments to not just have them related to another measure of health (indicator) but need to link to function or services
- Very little data currently exists. Data collection needs to consider:
- Sampling protocols appropriate for the question being asked (measuring diversity versus spatial analysis of variation)
- What constitutes a baseline state in a regional context?
- What are the drivers of biodiversity?

Water management approaches

1. Investigate pricing as a mechanism to encourage more efficient use of water
   - Farmers currently pay a flat rate for infrastructure costs on top of water use. If infrastructure costs were reduced so that a greater proportion of bills related to water consumption this might act as an incentive to use less
   - Increase fees to discharge to tailwater drains
2. A single manager with responsibility for the entire lower Burdekin water is needed (DERM?)
   - There are currently 4 managers with some responsibility
3. Needs to be extraction of water as well as salt from under the whole floodplain
4. Reduction in surface water application via irrigation practices and improved water infrastructure

Appendix B Legend for conceptual models



Appendix C Complete results for indicators of hydrological alteration

	Bores						
Metrics	**38048**	**99088**	**99090**	**11900009**	**11900021**	**11910012**	**11910064**
Proportion of years in time series with very shallow wet season water table (< 0.5 m)	0	0	0	0	0	100	0
Proportion of years in time series with shallow wet season watertable (0.51 – 4.00m)	57	0	100	44	0	-6	100
Proportion of years in time series with moderate wet season watertable (4.10 – 8.00 m)	-578	-21	35	-700	32	-3	-8
Proportion of years in time series with deep wet season watertable (8.10 – 20.00 m)	0	8	-67	-100	-169	0	5
Proportion of years in time series with very deep wet season watertable (>20.00 m)	0	0	0	0	0	0	0
Proportion of years in time series with very shallow dry season water table (< 0.5 m)	0	0	0	0	0	0	0
Proportion of years in time series with shallow dry season watertable (0.51 – 4.00m)	25	0	0	29	0	-2	100
Proportion of years in time series with moderate dry season watertable (4.10 – 8.00 m)	-256	-23	27	-560	13	2	-51
Proportion of years in time series with deep dry season watertable (8.10 – 20.00 m)	0	15	-100	0	-300	0	15
Proportion of years in time series with very deep dry season watertable (>20.00 m)	0	0	0	0	0	0	0
Mode of wet season DTWT	57	-108	12	11	-7	32	82
10th percentile of wet season DTWT	56	5	-27	-131	-35	2	-6
90th percentile of wet season DTWT	39	20	-24	-6	-5	-64	-48
coefficient of variation of wet season DTWT	-53	-3	-36	-190	-129	24	-22
Mode of dry season DTWT	70	35	-49	-33	9	47	-59
10th percentile of dry season DTWT	55	1	-20	-104	-22	-9	-10
90th percentile of dry season DTWT	35	13	-16	-10	-3	-43	-3
coefficient of variation of dry season DTWT	-90	-33	-64	-71	-50	-4	-41
rate of change over full time period	96	89	94	92	60	65	89

Appendix D The environmental history of Burdekin River wetlands as inferred from sediment and fossil diatom records (consultant's report)

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Introduction

This study examines sediment cores to document environmental change in three wetlands: Swan’s Lagoon, Labatt Lagoon and EVT Inkerman on the Burdekin River floodplain. These systems are classified as Ecosystems dependent on surface expression of groundwater, as outlined in Section 2.2.2 of the report.

Wetland sedimentary records contain numerous indicators of past environments (Battarbee, R W 2000) that can be utilised to provide long-term records of changing conditions. The main indicators used in this study are diatoms.

Diatoms are microscopic algae of the class Bacillariophyceae. They are unicellular with siliceous cell walls, consisting of two valves that, when joined together, form a frustule (Stoermer & Smol 1999). The morphological intricacy of the valves forms the basis of taxonomy. Diatoms can be identified to species level and beyond (Round, Crawford & Mann 1990) and numerous taxonomic texts are available to guide identification. Diatoms are sensitive to a number of water quality parameters (Reid, M. A. et al. 1995), of which pH (Tibby, J. et al. 2003), salinity (Gell, PA 1997) and nutrient status (Tibby, J. 2004) are among the most important. Many species have had their pH, salinity and total phosphorus optima and tolerance quantitatively defined through the use of modern calibration sets (e.g., Gell, PA 1997; Tibby, J. 2004). Qualitative inferences of changing conditions can also be gained through the known species preferences for different habitats. For instance, the ratio of open-water forms (planktonic species) to bottom-dwelling forms (benthic species) has been used to infer changes in lake depth (Brugam, McKeever & Kolesa 1998). Similarly, the occurrence of marine species in fresh coastal wetlands (Tibby, J, Lane & Gell 2007) or riverine species in sediments of floodplain wetlands (Reid, M.A. & Ogden 2009), may infer connectivity to an external source of water.

The primary aim of this study was to identify baseline conditions of three floodplain wetlands and determine the degree to which, if at all, modern conditions deviate from that baseline. This study therefore directly informs Objective 2 of the report; assess and report responses by ecosystems to altered groundwater hydrology, as outlined in Section 1.1.

Site descriptions

In order to provide some context for interpretation of results, brief descriptions of the three study sites are provided, including any relevant background information regarding known impacts on the wetlands.

Swan’s Lagoon

(20°7'9.66"S, 147°15'30.29"E).

Swan’s lagoon is at the western edge of the Burdekin River floodplain, approximately 1500 metres from the main river channel. Although some sugar cane is grown nearby, the lagoon is located on a grazing property and is used as a water supply for stock. Historically, water levels fell during the dry season with only the deepest sections retaining permanent water. The site receives no direct supplemental flows, however some tailwater from nearby cane irrigation does enter the lagoon, possibly leading to higher water levels during the dry season. Despite this, Swan's Lagoon remains one of the least impacted wetlands on the Burdekin River floodplain and may still be within its natural regime of variability. For this reason, Swan's Lagoon is the reference site for this study.

Labatt Lagoon
(19°36'44.22"S, 147°19'3.73"E)
Labatt Lagoon is located in the Sheep Station Creek palaeochannel approximately five kilometres from the Burdekin River channel. Historically, the lagoon was an ephemeral wetland, drying to small pools in the dry season. Since the inception of the water boards (1960s) the wetland has been receiving occasional supplemental flows, increasing the permanency of water in the system. Full supplemental flows commenced in 1989 after completion of the Burdekin Falls Dam. This lead to static water levels in the wetland and provided ideal conditions for major weed invasions by Water Hyacinth (Eichhornia crassipes) and Para Grass (Urochloa mutica), which began in late 1980s and eventually formed dense weed mats at the expense of open water. Labatt Lagoon, along with several wetlands along the Sheep Station Creek channel, formed part of a study investigating the effect of removing aquatic weeds on fish communities (Perna et al. 2011). Invasive weeds were mechanically removed from Labatt Lagoon in August 2000 and significant improvements were noted in both the richness and diversity of the fish community following a flood event in November 2000 which enabled fish to recruit in the lagoon from remnant habitat refuges (Perna et al. 2011).

EVT Inkerman
(19°44'4.70"S 147°31'28.33"E)
This site is a perennial freshwater wetland primarily used for watering stock and irrigating sugar cane. The wetland is located on the eastern edge of the floodplain, approximately 2 km upstream from Saltwater Creek and the mangrove marshlands of the Coral Sea. Hence, EVT Inkerman would naturally receive tidal inflows. In the historical period, a small bund wall protected the site from most tidal influence, though spring high tides did inundate the site, which would dry to a cracking clay saltpan. Recently, a landowner increased the height of the bund wall, removing all tidal influence from the wetland which, in conjunction with inflows of irrigation tailwater, has caused the site to become perennially fresh.

Field and laboratory methods

Core collection

Collection of sediment cores from the wetlands described above was undertaken from a Kawhaw coring platform lashed between two aluminium dinghies and tied to the shore to ensure stability. Sediment cores were collected using a Livingstone corer (Livingstone 1955). This coring method collects individual one metre lengths of sediment which are extruded into a storage tray, wrapped in cling film and stored for transport to the laboratory. The benefits of this method are that cores are taken from a single hole and are therefore contiguous, thus avoiding problems with aligning adjacent core sections. The disadvantages are that very soft sediments may lose stratigraphic integrity during extrusion, while more solid sediments may become compacted. The sediments from the Burdekin floodplain wetlands generally fell into the latter category. As a result, correction factors were determined for each core section and applied to sample depths. Two sediment cores were collected from each of the three floodplain wetlands.

Sediment magnetic susceptibility
The magnetic susceptibility of the sediment cores from each of the study sites was determined to assess both broad changes in the type of material being deposited at each site and, also, to assess the degree of uniformity in the sediment composition. This approach has been adopted in a number of Australian studies to allow intensive investigations to be focussed on a single (representative) core (see for example Leahy et al., 2005). Magnetic susceptibility of the sediments was measured at 1 cm intervals using a Bartington MS2E core logging sensor.

Sediment description, moisture and organic content estimation

The sediment cores were split lengthways in the laboratory and the main features and colours of the core were described using a modified version of the Troels-Smith system of sediment description (Kershaw 1997), with the aid of Munsell soil colour charts.

Estimates of moisture and organic content were made on regular centimetre thick samples from all cores. Wetland sediment organic matter, for the most part, reflects a balance between internal production and preservation of plant and, to a lesser extent, animal remains and the delivery of inorganic material from outside the wetland. In riverine settings, the dominant source of inorganic material is sediment from the catchment. Sediment moisture content often reflects the proportion of organic matter, in combination with a variety of factors such as changing particle size. While both measures provide rudimentary information, in combination they provide a means to assess the validity of hypotheses generated from other data. Moisture content was determined by the amount of mass lost after heating at 105ºC for 24 h. Organic content was determined as the amount of dried sediment lost after ignition at 550ºC for 4 h (Heiri, Lotter & Lemcke 2001).

Sediment dating

Radiocarbon (14C) dating was used to provide ages for sediments deposited before European settlement. To provide age estimates for younger sediments, we have submitted samples from the upper sections of the cores for 210Pb and 137Cs dating (Appleby 2001). However, these dates are not yet available.

Radiocarbon dating utilises the known half life of 14C (5730 years), which is incorporated into living organisms and commences decay when these organisms die, to estimate the ages of organic matter preserved in lake sediments. However, the concentration of 14C in the atmosphere, once thought to be static, is now known to have varied. Hence, 14C ages need to be calibrated to calendar ages using known calibration curves derived from 14C dating of material (in particular tree rings) of known age.

14C dating was undertaken by Beta Analytical Radiocarbon Dating Laboratory using standard techniques and reported as year before present (BP) where 'present' = 1950. The calibrated ages quoted in this study are the median of the two sigma distribution of calendar ages determined using the Southern Hemisphere calibration curve (McCormac et al. 2004) and implemented in computer program CALIB (www.*calib.qub.ac.uk*). These results are reported as calibrated years before 2000 AD (cal yr B2K).

Diatom sample preparation and counting

Diatom samples were taken as 1 cm slices at regular, but differing, intervals in each of the three study cores to eventually provide approximately 80 samples per record. In this report, results from initial analyses of 20 samples per record are reported. Diatoms were prepared using a modified version of Battarbee et al. (2001) with 2-3 h treatments in 10% HCl and 10% H202, respectively, to remove carbonate and organic matter. Following each of these steps, samples were washed (three times) in distilled water and allowed to settle for 12 h between washes. Prepared slurries were dried on coverslips which were then inverted and mounted on permanent slides using Naphrax® mounting medium. Diatoms were identified at 1000× magnification, using either a Nikon Eclipse E600 with differential interference contrast optics or with an Olympus BH2 microscope with bright field illumination. Taxa were identified with reference to a variety of sources (in particular Krammer & Lange-Bertalot 1986; 1988, 1991a, 1991b; and Sonneman et al. 1999). A number of diatom taxa could not be identified as known species in which case these were given descriptive epithets or the descriptor "aff." where they had strong affinities to known species, but did not completely fit the description. Diatom counting was undertaken along transects across the coverslip. Samples were considered 'barren' (i.e. no diatoms were preserved in the sediment) if no diatom valves were encountered in one transect. Raw data was converted to relative abundance prior to illustration. Diatoms were classified into four habitat groupings. Planktonic diatoms are those which spend all or the majority of their life cycle in the water column. Aerial diatoms can survive periods of exposure to the atmosphere or drying (Johansen 1999) and, as a result, they are often found on moist soils or other surfaces (e.g. peat bogs). Epiphytic diatoms are those species which attach to plants. The remaining diatoms are classified as "other" as they can persist in a variety of (non-planktonic) habitats.

The ecology of dominant diatoms in the records

Three key species occur in the diatom records from the study sites; *Aulacoseira granulata*, *Diadesmis confervacea* and *Cyclotella meneghiniana*. Given that these dominate the records at times, a brief discussion of the ecology of these species is provided.

Aulacoseira granulata is a freshwater planktonic diatom species found in lakes, reservoirs and rivers in many parts of Australia and the world (Tibby, J. 2004; Tibby, J. & Reid 2004). Notably, *A. granulata* is the dominant phytoplankton in the River Murray (Hötzel & Croome 1996; Tibby, J. & Reid 2004) and is generally found in abundance in water with moderate to high concentrations of nutrients. For example, in south-east Australian reservoirs this taxon has an total phosphorus optimum of approximately 50 µg l-1 (Tibby, J. 2004). *Aulacoseira granulata* is chain-forming and, like all diatom species, has no means of regulating its own buoyancy. As such, *A. granulata* has an obligate requirement for turbulent water to remain suspended in the water column (Bormans &

Webster 1999). This observation, in combination with the relatively small size of the Burdekin floodplain wetlands, suggests that the abundance of *A. granulata* is likely to reflect connection to the Burdekin River. This situation is similar to a number of River Murray wetlands where the abundance of *A. granulata* in surface sediments is positively correlated with the level of connection to the River Murray (Reid, MA, Ogden, R. 2009). As a result, in wetland palaeolimnological records from the River Murray, the abundance of this taxon is interpreted as indicative of wetland connectivity (Gell, P, Bulpin, et al. 2005; Gell, P, Tibby, et al. 2005). Similarly, we interpret the abundance of *A. granulata* in the Burdekin wetlands as reflective of connection to the Burdekin River.

Diadesmis confervacea is an aerophilic diatom, meaning that it can survive periods of exposure to the atmosphere or drying (Johansen 1999). As a result this diatom is often found on moist soils, peat bogs or in springs (Johansen 1999). The appearance of this species in the diatom records may therefore be indicative of 'low-flow' periods in which the lagoons experienced very low water levels and dried out. Alternately, the increased abundance of aerial taxa can represent periods when increased deposition of sediments has occurred, such as erosion after land clearance.

Cyclotella meneghiniana is a planktonic species that is very commonly encountered in water bodies studies in Australia and around the world. It is an alkaline species with a pH optimum of 8.2 pH units (from data in Tibby et al. 2003). *Cyclotella meneghiniana* is tolerant of high nutrients (Kelly, Penny & Whitton 1995). Tibby (2004) found that, consistent with other studies, *C. meneghiniana* is abundant in high nutrient waters and has a TP optimum of 67µg l-1. It has been found to be tolerant of a wide range of salinity (e.g., Davies et al. 2002; Gell, PA 1997) and, as such, its value as a salinity indicator is poor. In general, the appearance of this species in diatom records is interpreted as representing elevated nutrient levels.

Results and Discussion

Age of the sediments

Assuming that the basal sediments were reached in all sites, then Swan's and Labatt Lagoons are similar in age and are approximately 1300 years old, while EVT Inkerman, with sediments over 7500 years old, is considerably older (Table 1). At Labatt Lagoon, sediments at considerable depth (66-67 cm) were deposited recently, as indicated by the presence of modern carbon in the sample, which emanates from the atmospheric thermo-nuclear bomb testing that commenced in the 1950s. Since the concentration of atmospheric radiocarbon peaked in approximately 1966, any modern radiocarbon value could, theoretically, have occurred on either the rising or falling limb of the peak, providing two possible ages (Hua 2009). Calibration of the 66-67 cm sample indicates that, at its earliest, it was deposited in 1963. However, based on the probability distribution of the possible ages, the sediments are more likely to have been deposited in 1973 or later. The more recent sediments have been submitted for 210Pb and 137Cs dating which will provide more precise estimates of the timing of events over the last 150 years (Appleby 2001). However, these results are not yet available.

Table 1. Site ages. Note that by convention radiocarbon ages are presented as years "before present" (BP) where 'present' =1950. The uppermost sample age for Labatt Lagoon is not expressed using this convention since it is younger than 1950.

Site	Depth (cm)	Age (14C years BP)	13C/12C	Ages (years before 2000)
Swan's Lagoon	150-152	730 +/- 30 BP	-24.8 ‰	690
Swan's Lagoon	228-229	1260 +/- 30 BP	-22.7 ‰	1170
Labatt Lagoon	66-67	143.3 +/- 0.4 pMC	-29.0 ‰	37 (i.e. 1963 at the earliest)
Labatt Lagoon	142-143	1280 +/- 30 BP	-19.7 ‰	1200
EVT Inkerman	193-194	6510 +/- 40 BP	-27.0 ‰	7420
EVT Inkerman	270-271	6610 +/- 50 BP	-27.5 ‰	7510

Given the clear differences between study sites, the remainder of the results will be presented and discussed individually.

Swan's Lagoon

Sediment composition

Two cores were retrieved from Swan's Lagoon with Core 1 (the primary core that was used for all analyses) measuring 277 cm (uncorrected for compaction). The upper 2.35 m of sediments were predominantly homogenous (i.e. not layered or stratified) organic very dark greyish brown muds (Munsell colour 2.5 Y 3/2). See plate 1 for an example. Below 2.35 m, there are a number of layers of gravel (Plate 2) of varying size and of a similar composition and appearance to the stream bed gravels in the main Burdekin River channel.

Plate 1. Representative photograph of the upper 2.35 m of sediment from Swan's Lagoon. Approximately 24 cm of sediment from 139-163 cm is shown in this photo.

Plate 2. Photo of gravels near to the bottom of the Swan's Lagoon core. The depths shown in this photo are from 238-262 cm

Sediment magnetic susceptibility

The relationship between the magnetic susceptibility profiles of the two Swan's Lagoon cores (Figure 1) suggests that there was an overall good correspondence between these records. As such, reasonable confidence can be attributed to the representativeness of Core 1.

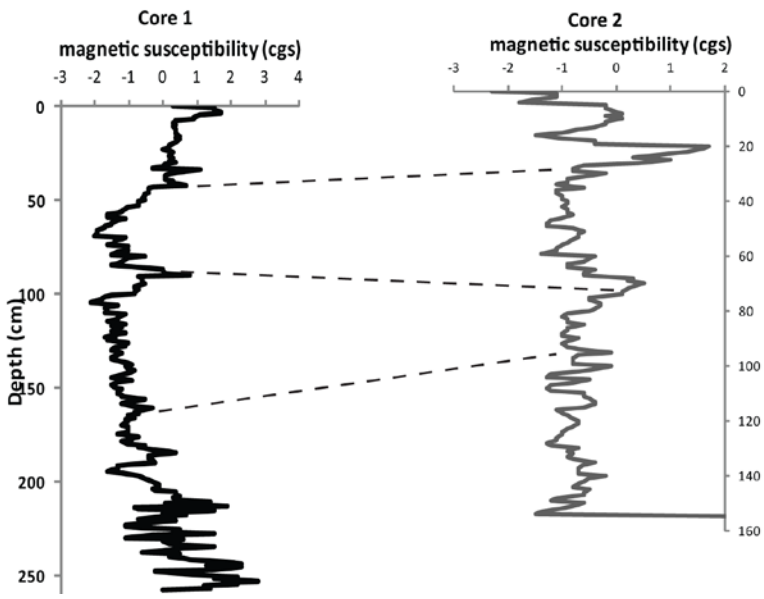

Figure 1. Magnetic susceptibility profiles of the two Swan's Lagoon cores. Likely points of correspondence are indicated by the dashed lines. Note the difference in length of the two cores.

History of Swan's Lagoon

It appears that the level of connection between the Burdekin River and Swan's Lagoon has varied on a variety of timescales. On shorter timescales (represented by individual samples such as at 125 cm, see Figure 2) there are very high relative abundances of Diadesmis confervacea that may reflect times when Swan's Lagoon was isolated from the Burdekin River for a prolonged period of time and the exposed wetland sediments were colonised by D. confervacea. On longer time scales, there is consistent evidence for the changing influence of the Burdekin River on Swan's Lagoon.

The interpreted history of Swan's Lagoon is based primarily on the diatom data, with complementary information being provided by interpretation of the nature of the sediments. Three phases have been identified in the history of this wetland, based on differences in the diatom assemblages and sediment characteristics.

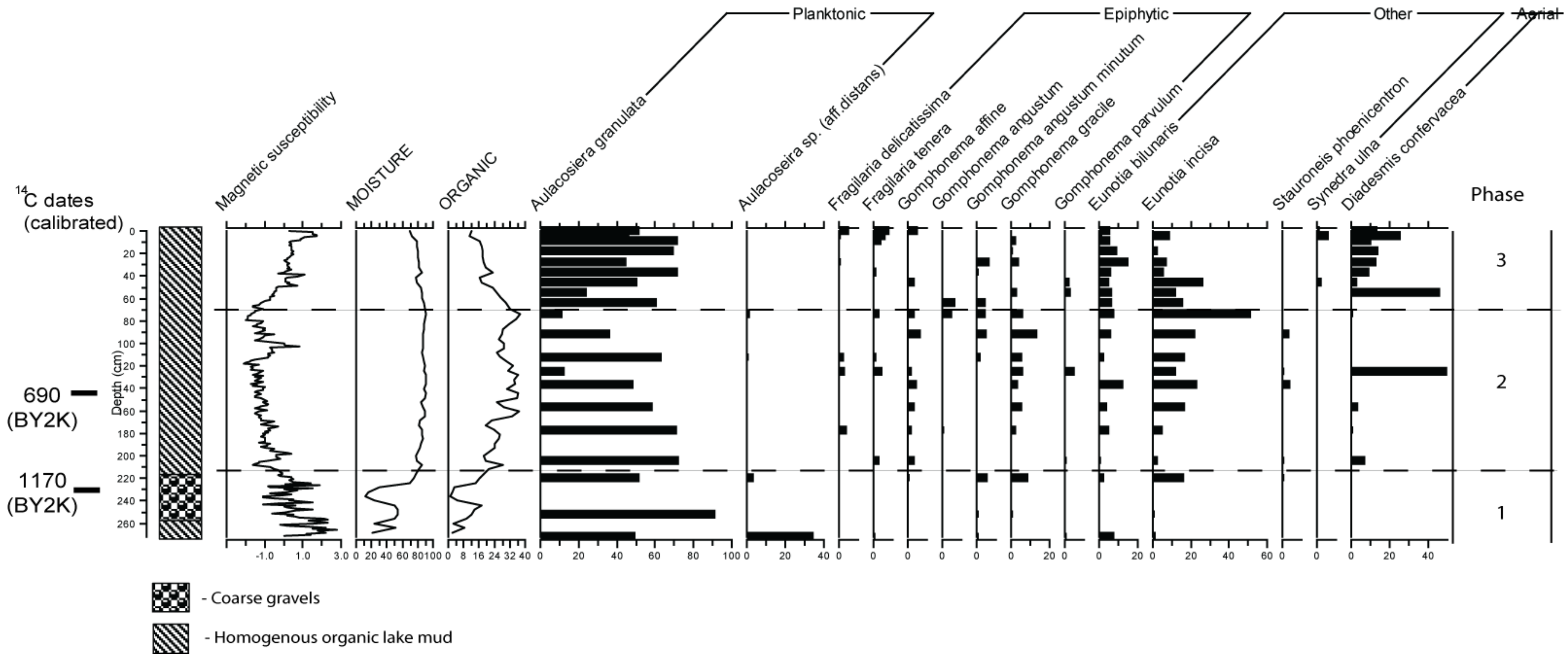

Figure 2: Summary diatom stratigraphy from Swan's Lagoon. Note that only taxa that exceeded 5% in any sample are displayed. Also shown is the sediment lithology magnetic susceptibility and organic and moisture content of the sediment.

Phase 1 (270-230 cm)
The most distinctive feature of the sediments deposited in phase 1 of Swan's Lagoon's development are the coarse gravel bands shown in plate 2. These gravels are indicative of sediment transport by a high energy stream and may represent a time when there was still some connection to the main Burdekin River before a channel avulsion. Diatoms were not sampled from the gravels themselves since preservation in such sediments is generally very poor. However, the samples either side of this gravel band are dominated by freshwater planktonic diatoms from the genus Aulacoseira (Figure 2) which are likely to be of riverine origin. In addition, magnetic susceptibility values are highest in this phase.

Phase 2 (230-65 cm)
In this phase, the proportion of benthic taxa is at its highest, with Eunotia incisa a particularly important part of the flora. During this phase, the proportion of organic matter is also maximal, although there are clear fluctuations within the values. These factors, in combination with the lower magnetic susceptibility and lower values for diatoms derived from riverine environments, suggests that this period of time was one where there was less interaction between the Burdekin River and Swan's Lagoon.

Phase 3 (65-0 cm)
The dating of the latest phase of the Swan's Lagoon record is somewhat imprecise as it relies on extrapolation of radiocarbon ages from sediments well below this point of change. This situation will be remedied when results of 210Pb dating are available. The most recent phase of the Swan's Lagoon core record appears to represent a return to a greater influence of riverine inputs in the wetland. There is an increased (though variable) abundance of Aulacoseira granulata in combination with increases in magnetic susceptibility, while the proportion of sediment organic content decreases to the top of the record. Contrasting somewhat to the interpreted increased riverine influence is the more consistent representation of Diadesmis confervacea in this section. However, in southern Australian wetland systems, aerial taxa often record increases following European settlement (Reid, M et al. 2007). These increases have generally been interpreted as representing increased soil erosion. Given these observations and the imprecision of the dating, it is possible that all of the sediments from phase 3 were deposited following European settlement of the region.

Conclusion

In comparison to marked and often wholesale changes observed in floodplain diatom records from southern Australia, in particular the Murray and the Yarra catchments (Gell, P, Tibby, et al. 2005; Leahy et al. 2005; Reid, M et al. 2007; Reid, M et al. 2001), there have been moderate changes in the diatom composition of Swan's Lagoon. It appears that for its entire existence, Swan's Lagoon has been a freshwater, circumneutral to alkaline, moderate to highly enriched wetland. The diatom taxon which dominates the lagoon today has been dominant for over 1300 years, suggesting that Swan's Lagoon is within its natural regime of variability. This suggests that the choice of Swan's Lagoon as a reference site within the Burdekin system is justified.

Labatt Lagoon

Sediment composition

The sediments from both Labatt Lagoon cores show clear evidence of environmental change. Apart from a small section of homogenous organic lake mud (90-105 cm – corrected depth, see Figure 4), much of the sediment below a depth of 75 cm consists of grey, cracking muds, indicative of an ephemeral environment (Plate 3). From 75 cm to the surface of the sediments, the sediment consists of soft, laminated lake muds (Plate 4) with high moisture content.

Plate 3: Example of the dry cracking muds representative of the majority of sediments below 75 cm (corrected depth) from Labatt Lagoon. The corrected depths shown in this photo are from 137-175 cm of Core 1.

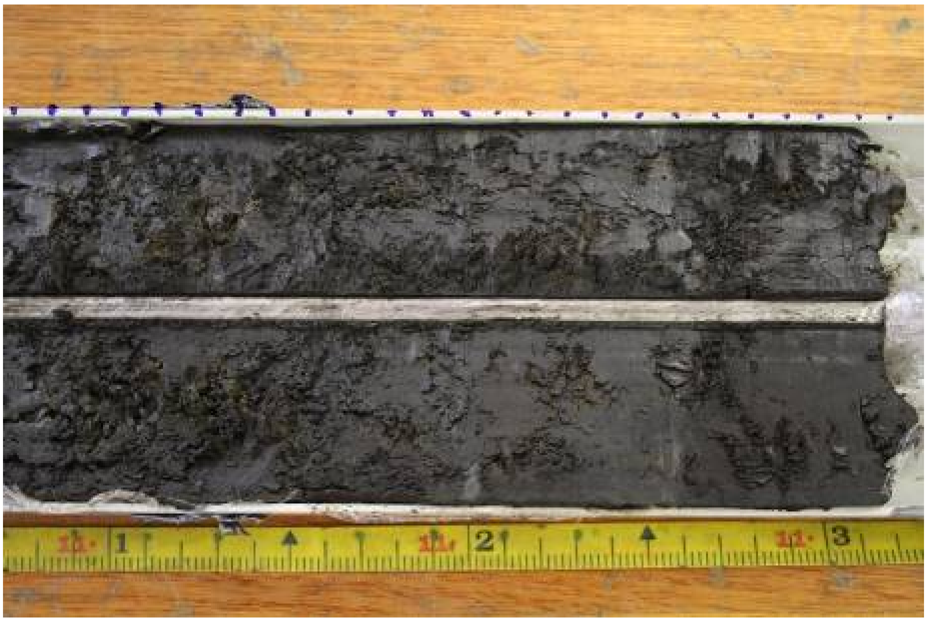

Plate 4: Example of the laminated lake muds above a depth of 75 cm (corrected depth) from Labatt Lagoon. The corrected depths shown in this photo are from 48-78 cm of Core 1.

Sediment magnetic susceptibility

As illustrated in Figure 3, there is an excellent relationship between the magnetic susceptibility of two cores extracted from Labatt Lagoon. The sedimentary record from Core 1, which was used for more detailed analyses, can be considered to be representative of the wetland.

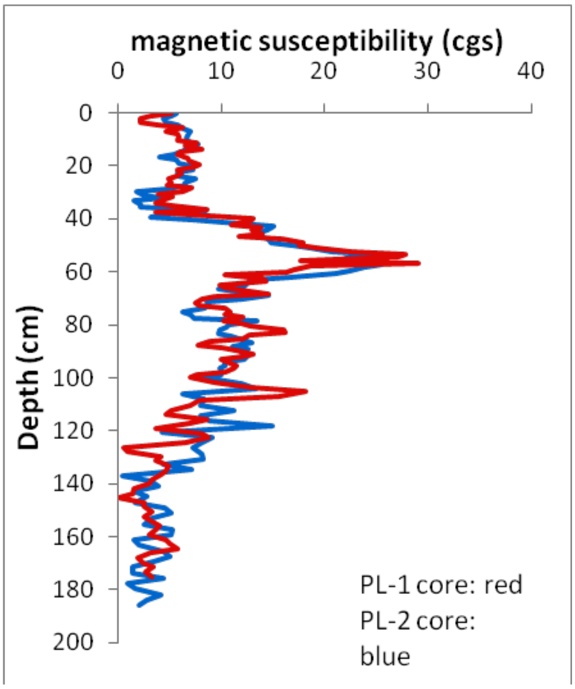

Figure 3: Magnetic susceptibility from the two Labatt Lagoon cores.

History of Labatt Lagoon

As was the case with the Swan's Lagoon reference site, Aulacoseira granulata and Diadesmis confervacea are key components of the diatom record for Labatt Lagoon (Figure 4), particularly in the bottom half of the core. However, in contrast to Swan's Lagoon, the relative abundance of these species has varied markedly over time. Three phases in the history of Labatt Lagoon have been identified from a combination of the diatom record and sediment characteristics.

Phase 1 (177-115 cm)

This phase is dominated by the freshwater planktonic diatom A. granulata and, as was the case for Swan's Lagoon, this is interpreted as being indicative of regular connection to the Burdekin River. In contrast to the Swan's Lagoon record, there was no deposition of sand or gravel during this phase and the magnetic susceptibility of the sediment remains relatively low and stable (Figure 4). This suggests that, while river connectivity occurred, it was of a lower energy than that experienced in the early phase of the Swan's Lagoon record. This is most likely due to the greater distance of Labatt Lagoon from the main river channel, than Swan's Lagoon. The sediments during this phase are dry cracking muds, indicative of ephemeral environments. The constant, though relatively low, presence of the aerial diatom D. confervacea supports this hypothesis.

Radiocarbon dating of the sediments during this phase indicate a calibrated age of 1200 yrB2K (ca. AD 800) at a depth of 143 cm (Table 1). However, in the absence of further dates in this section of the core, it is not possible to precisely determine the period of time represented in phase 1.

Phase 2 (115-65 cm)

While the sediments at the beginning of phase 2 remain unchanged from phase 1, later peaks in the magnetic susceptibility suggest they may have originated from a different source than those in phase 1. A corresponding sudden increase in the relative abundance of D. confervacea (at 107 cm, Figure 4) suggests that this may have been due to an increase in catchment erosion, possibly due to agricultural settlement and land clearance. Aulacoseira granulata maintains a constant presence in the record, indicating that river connectivity continued to play a key role in the hydrology of this wetland.

At a depth of 84 cm, Cyclotella meneghiniana first appears in the record, accounting for 11% of the species assemblage, suggestive of nutrient enrichment of the wetland. The sudden disappearance of A. granulata and rapid increase in the proportion of D. confervacea in the record marks the end to this phase of wetland history. This occurs in conjunction with the appearance of laminations within the sediment. Radiocarbon dating of sediments at 66 cm indicates that this shift in conditions occurred in 1963 at the earliest, but that it was more likely to have occurred in 1973 or later.

Phase 3 (65-0 cm). Post 1963 AD

The beginning of this phase marks a sudden and dramatic shift in environmental conditions, not least of which is a massive increase in the rate of sedimentation. The laminated sediments that are evident during this phase (Plate 4) are indicative of differing sources of sediment at varying times. The lighter coloured inorganic clay sediments are likely to be riverine in origin, deposited by supplemental flows, with darker sediments reflecting standing water and internal production. This hypothesis is, as yet, untested, although fine resolution diatom sampling of the various laminations may provide greater insight.

Phase 3 is also marked by increased sediment moisture content, possibly indicative of perennial conditions, which would correspond with the known history of the site. Interestingly, despite permanent water levels being maintained in Labatt Lagoon, planktonic diatoms barely feature in the record. Instead, the diatom record marks a dramatic increase in the proportion of the epiphytic species Cocconeis placentula and Epithemia adnata, which attach themselves to aquatic vegetation. The substantial representation of these taxa is clear evidence for a marked increase in aquatic vegetation in Labatt Lagoon. The proportions of D. confervacea in phase 3 are similar to those in phase 1 of the record and increase until a depth of 10 cm, at which point this taxon accounts for 49% of the assemblage. The cause of this is unclear, however thick weed mats had colonised Labatt Lagoon by this stage (Perna et al. 2011) and the moist surface of the weed mats would provide an ideal habitat for D. confervacea and may explain the prevalence of this species in this part of the record.

Conclusions

Labatt Lagoon has clearly undergone substantial changes over the period covered by this record. It is interesting to note that the early portion of the record (phase 1) shows similarities to conditions at the Swan's Lagoon reference site, with regular riverine connectivity, albeit with lower energy conditions. Therefore, phase 1 may be considered to be representative of background reference conditions for this wetland.

Environmental conditions clearly changed during phase 2, with increased sediment deposition into the wetland, possibly due to land clearance. There is insufficient chronological control to determine the timing of this change, however the transition to phase 1 is well dated and most likely occurred in 1973. Accepting this date, this would indicate a sedimentation rate for phase 3 of the record of approximately 1.7 cm yr-1. If the sediments at 66 cm date to 1963 AD, then the sedimentation rate for this phase is still 1.3 cm yr-1.This sedimentation contrasts to the an average sedimentation rate between 142 and 66 cm of just 0.06 cm yr-1.

These data indicate that the Labatt Lagoon ecosystem is well outside its natural range of variability. Perna et al. (2011) outline the degree to which supplemental flows altered the environment of the wetland by encouraging weed invasion. These changes are reflected in the diatom record, while the nature of the sediments and rate of sedimentation also significantly differ from background conditions.

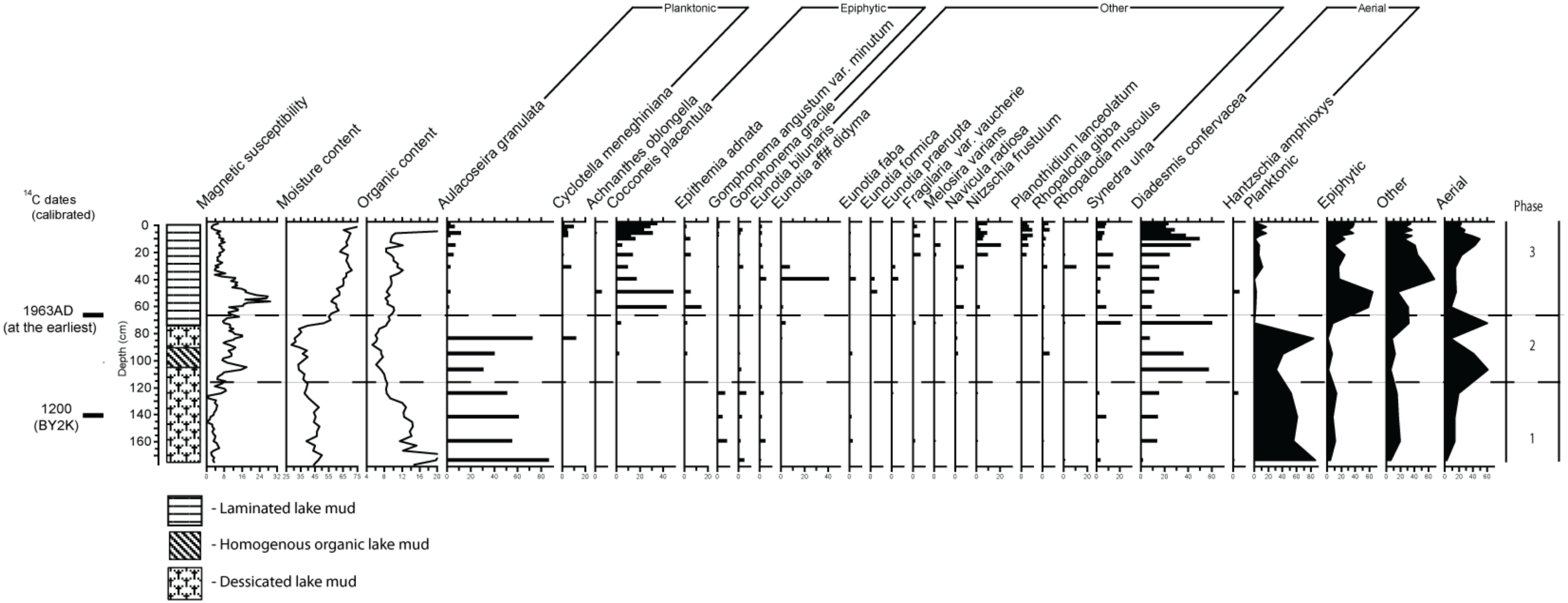

Figure 4: Summary diatom stratigraphy from Labatt Lagoon. Note that only taxa that exceeded 5% in any sample are displayed. Also shown is the sediment lithology, magnetic susceptibility and organic and moisture content of the sediment.

EVT INKERMAN site

Sediment composition

Two cores were retrieved from the EVT site. Core 1 was the longest, at 285 cm (corrected depth). Core 2 was collected approximately 400 metres downstream of core 1 in slightly shallower water. The corer could only penetrate 56 cm into the sediment at this site, before striking extremely hard sediments. The shallower water depth suggests that this site would dry out more readily than the core 1 site, which may explain the shorter sediment record and the hard nature of the underlying sediments. Core 2 is therefore not discussed here. All analyses were undertaken on core 1, and the sediments provide evidence of significant changes in the environment of the EVT site.

From the base of the core to 112 cm, the sediments are very well consolidated, grey clays with a fine sandy texture (Plate 5) possibly indicative of low energy estuarine environments. The sediment is homogenous in colour, with occasional bands of slightly coarser sand being the only stratigraphic change. Extrusion of these sediments was extremely difficult due to their consolidated nature. At 112 cm, there is a transition to coarse sand sediments (see Plate 6) which remain until a depth of 51 cm. Above this point, the sediments consist of grey, cracking muds, indicative of an ephemeral environment (Plate 7).

Plate 5: Example of the homogenous grey clays representative of the sediments between 285 cm and 112 cm in Core 1. The corrected depths shown in this image are from 162-185 cm.

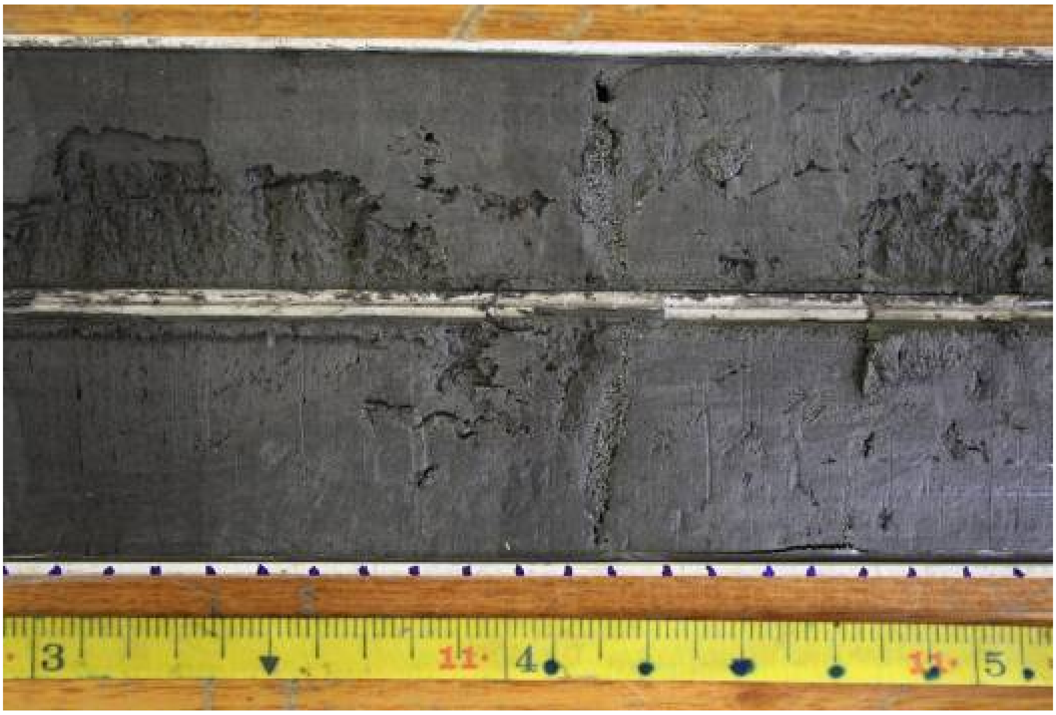

Plate 6: The transition from grey clay to coarse sand at a corrected depth of 112 cm. The corrected depths shown in this image are from 101-121 cm.

Plate 7: Example of the dry cracking clays representative of the upper 50 cm of sediments in Core 1. The corrected depths shown in this image are from 0-30 cm.

Sediment magnetic susceptibility

The magnetic susceptibility record from core 1 is presented in Figure 5. Substantial variability is evident, especially below 150 cm. A marked shift in magnetic susceptibility occurs around 50 cm, in line with the transition in the sediment lithology from coarse sands to dry cracking muds.

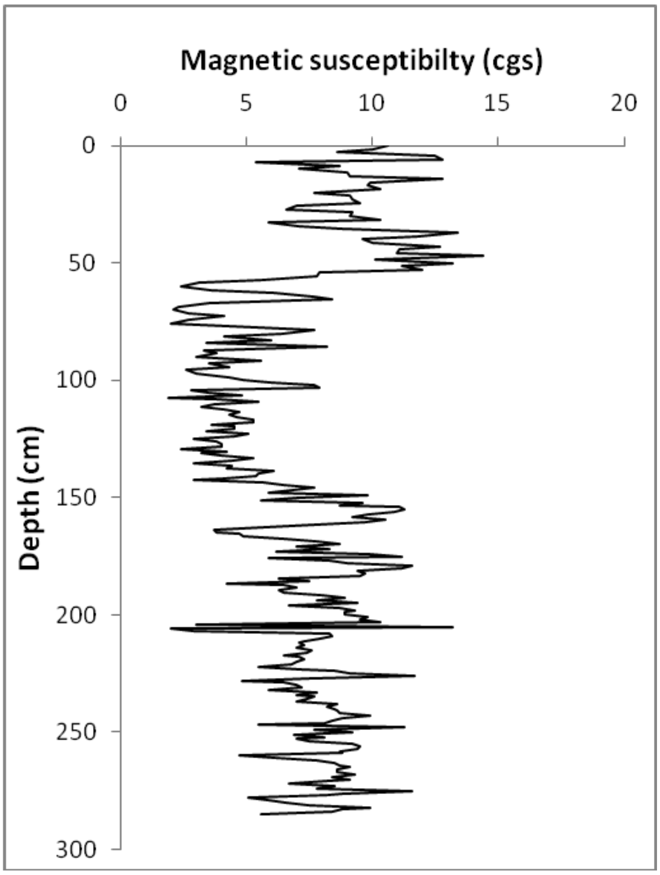

Figure 5: Magnetic susceptibility of the EVT Inkerman core

History of the EVT Inkerman site

Sediment ages for this record indicate that EVT Inkerman is notably older (>7,500 years old) than the other two study sites. The antiquity of the sediments, which would otherwise indicate a very slow rate of accumulation, suggests that there is a discontinuity in the sediment. The rapid shift in the nature of the sediments, most noticeable at around a depth of 50 cm and reflected in the magnetic susceptibility (Figure 5), suggests that this may be the location of the discontinuity. Prior to this, the sediments consist of either grey sandy clays, or coarse sand. Both indicate an environment quite different from the present.

Diatoms are not preserved below 70 cm in the EVT sediments (Figure 6). There are several factors that may hinder the preservation of diatom valves in sediments, including low silica levels, highly alkaline conditions and the physical destruction of the valves through wave action and abrasion in coarse-grained sediments. It is unclear what impeded the preservation of diatoms at this site.

Diatoms first appear in the record at 70 cm, and again at 40 cm, though valves are sparse and count sizes are low (see Figure 6). As such, little reliable information can be derived from these samples. Diatoms are more abundant in the uppermost 20 cm of the record and, though this consists of only three samples, there is a clear shift in the data between the samples from 20 cm and 4 cm and the sample from the surface of the sediments. Diadesmis confervacea dominates the species assemblages at 20 cm and 4 cm. This corresponds with the known history of the wetland, which would dry out in the dry season, providing habitat for this species. However, Diadesmis confervacea is not present at all in the uppermost sample, with Cyclotella meneghiniana and Nitzschia lacuum dominating the record. The demise of the aerial taxon D. confervacea, and its replacement with the planktonic C. meneghiniana, suggests a shift away from an ephemeral environment towards one with perennial standing water that is nutrient enriched. This aligns with the known history of the site, with a recent increase in the height of the bund wall downstream from the core site.

Conclusions

Although our results are only derived from one site in the EVT Inkerman wetland, the extent of the changes that are evident suggest broad scale changes that occurred over much of the wetland.

The lower sediments, dated at around 7,500 years old, are indicative of a low energy, estuarine environment. Woodroffe (2009) provides evidence of elevated relative sea-levels in Cleveland Bay, approximately 40 km to the north of the Burdekin River delta, during the last 10,000 years. She finds that relative sea-levels rose above current levels sometime between 8,000 and 6,200 years ago, though has insufficient data to precisely determine the timing of this. These data from the EVT Inkerman site suggest that the timing was likely to be before 7,500 years ago. Woodroffe (2009) found that after this initial rise, relative sea levels remained elevated above current levels, attaining a maximum of approximately 2.8 m above present levels around 5,000 years ago, and only receded to present levels in the last 1,000 years. Given the proximity of the study site to mangrove marshlands and the historically documented tidal influence on this site, the findings of Woodroffe (2009) suggest that this site may have been permanently inundated with sea water for much of its history and support the hypothesis that the finely grained grey clays are of a low energy, estuarine origin.

Little information can be gained from the diatom record at this site due to the low abundance of valves in the majority of the record. Only the uppermost 20cm of sediment provide interpretable results. These indicate a change away from ephemerality towards a permanently inundated site that is nutrient enriched.

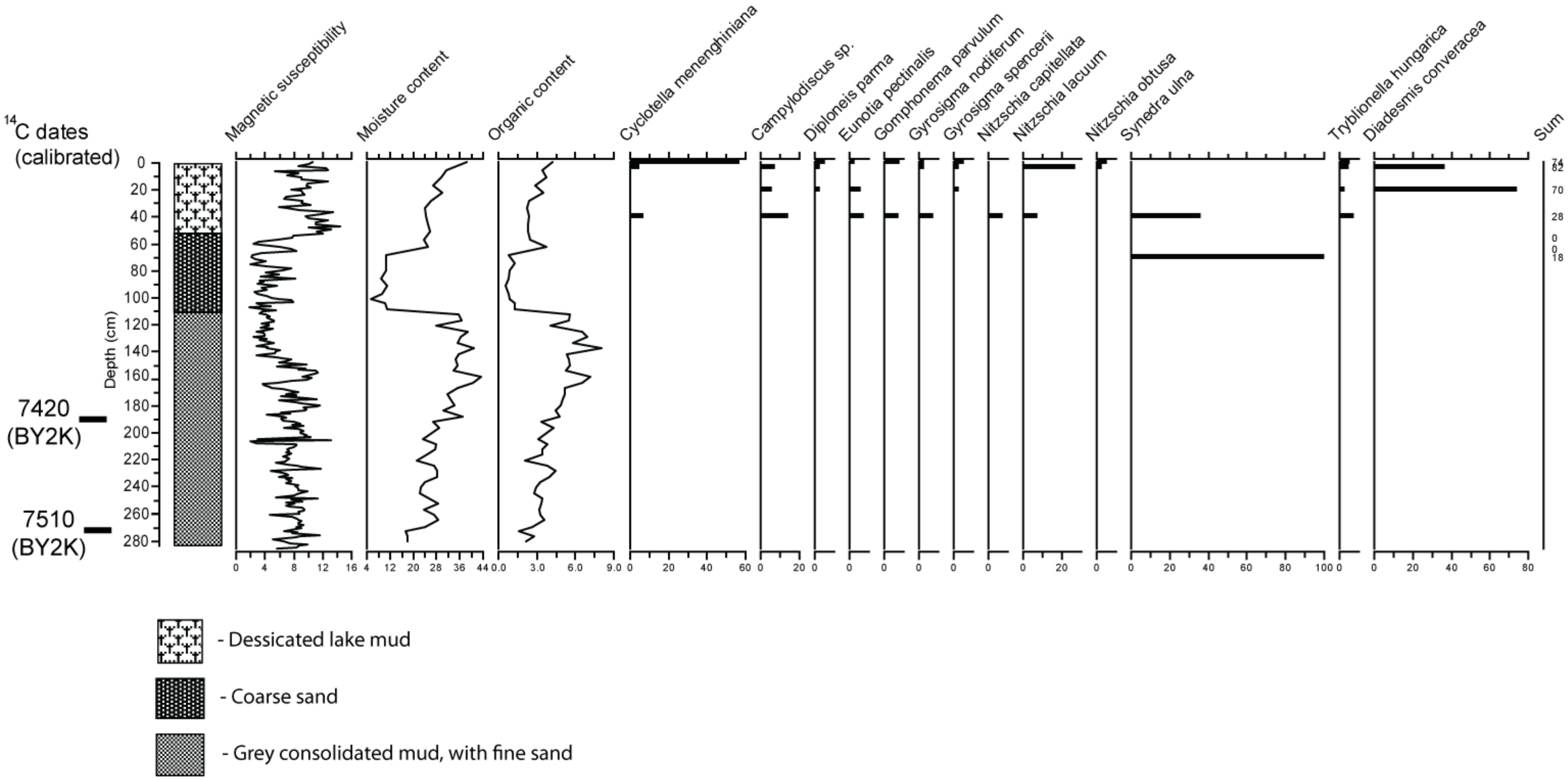

Figure 6: Summary diatom stratigraphy from the EVT Inkerman core. Note that the total number of diatom valves (illustrated as "sum") counted was low in all samples. Also shown is the sediment lithology, magnetic susceptibility and organic and moisture content of the sediment.

Summary: History of Burdekin River floodplain wetlands.

This study has identified clear changes in the history of Burdekin River wetlands. In particular, it is clear that Labatt Lagoon is in a condition that is very different from the vast majority of its more than 1200 year long history. Labatt Lagoon has changed from a site that was ephemeral, though was regularly flooded by the Burdekin River, to one which is permanent and dominated by aquatic plants. It is likely that the ecosystem of EVT Inkerman wetland is also markedly different from that which existed naturally. However, because this information is mainly drawn from the nature of sediments that provide less precise information than diatoms, this conclusion is somewhat less certain. By contrast with the marked changes observed in the other sites, and in many other studies (see above), the record from Swan’s Lagoon suggests that the site has undergone relatively little change.

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Appendix E Methods for stygofauna genetic analyses

Laboratory analyses

Specimens were identified to family (Parabathynellidae or Bathynellidae) prior to molecular analyses. Genomic DNA was extracted from 27 individuals (4 Bathynellidae and 23 Parabathynellidae) and stored in 100% ethanol, using the Gentra Systems PUREGENE DNA Purification Kit. Where possible, one to two body segments after the 9th segment and before the uropod were removed from each individual for DNA extractions, as these segments possess few or no morphologically useful characters. For bathynellids and smaller parabathynellids, approximately half of the animal had to be used in order to acquire sufficient DNA.

PCR amplifications were performed in 25 µl reactions containing PCR buffer, 0.1 units of AmpliTaq Gold® DNA Polymerase, (Applied Biosystems Inc.), 2-4µl $MgCl_2$, 2.5 mM of each dNTP, ~ 1 ng of DNA and 5.0 µM of each universal COI primer (Folmer et al., 1994). Thermal cycling occurred in an Eppendorf thermal cycler using the following conditions: enzyme activation at 94°C for 9 min, followed by 35 cycles of 94°C for 30 s, 47°C for 30 s and 72°C for 60 s with a final elongation step at 72°C for 6 min. PCR products were purified using the Ultraclean PCR Clean-up Kit (MOBIO Laboratories Inc.) and sequenced using the ABI Prism Big Dye Terminator Cycle Sequencing kit (Applied Biosystems). Raw sequences were compared with their corresponding chromatograms to clarify ambiguous bases, using Geneious Pro v5.4 (Drummond et al. 2011). Sequences were aligned using Clustal W (Thompson et al., 1994) and checked by eye. COI sequences were translated into amino acid sequences to determine if any gaps or stop codons were present.

Data analyses

Firstly, all COI sequences combined were analysed with the Neighbor-Joning (NJ) tree building method using the program MEGA version 5 (Tamura et al. 2011) and implementing the Kiumura 2 Parameter (K2P) model of nucleotide substitution (Kimura 1980) and 1,000 bootstrap replicates to assess statistical support for nodes in the tree. Anaspides tasmaniae (GenBank Accession Number DQ310660) was used as an outgroup because it is the sister lineage to the Bathynellacea and the only other extant order within the Syncarida. The mean percent genetic differences among the lineages in the resultant COI gene tree were calculated in MEGA and variation around these mean values was assessed using 1,000 bootstrap replicates. Seventeen of the 27 individuals that were genotyped belonged to Lineage P-A (see Results). To assess genealogical relationships within Linage P-A, we generated a parsimony haplotype network using the software TCS (Clement et al. 2000). As Lineage P-A was distributed across three bores (i.e. bores 11911170, 11910859 and 1191136), we assessed population genetic structure within this lineage. However, these analyses could only be performed across bores 11911170 and 11910859, as bore 1191136 was represented by only a single individual. We therefore calculated population genetic differentiation between the two bores using ΦST in ARLEQUIN (Schneider et al. 2000) and implemented 10 000 permutations of the empirical genotypes to assess statistical significance. ΦST is an index of genetic population differentiation that incorporates both the frequency and divergences of haplotypes in the sample.

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Appendix F Bacteria of the alluvial aquifer, lower Burdekin floodplain (consultant's report)

Bacteria of the alluvial aquifer, lower Burdekin floodplain

Report for:

Water Planning Ecology
Environment and Resource Sciences
Department of Environment and Resource Management
20 December 2011

Prepared by:
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Introduction

Although it is accepted that bacteria occur naturally in shallow groundwater (Cullimore 2008) research has focused on the impact on end-use water quality rather than the role of bacteria in natural ecosystems. Historically, characterization of the microbial component of groundwater ecosystems has focused on the fate and transport of pathogens, natural attenuation processes and bioremediation of contaminated aquifers (cf. Goldscheider et al. 2005). Very little is known of microbial communities in pristine aquifers. In general, uncontaminated aquifers are carbon-limited and devoid of light, and therefore reliant on chemical energy generation rather than photosynthesis (Goldscheider et al. 2005). In this context, heterotrophic bacteria are well adapted to the oligotrophic conditions in groundwater. Lithoautotrophic bacteria which fix carbon dioxide while oxidising inorganic compounds to obtain energy may also be autochthonous members of groundwater ecosystems.

Systematic studies in groundwater microbiology are required to develop an understanding of how microorganisms contribute to biogeochemical cycling and therefore, the food web and ecosystem maintenance (Griebler and Lueders 2009). It is essential to know more about the basic ecology and physiology of bacteria in these settings so that we can make better informed decisions in managing the biodiversity and integrity of these groundwater-dependent ecosystems (GDEs). The ecological water requirements of an ecosystem comprise those aspects of the natural water regime that are important for maintaining critical ecosystem ecological structure and function (NWC 2011). In this context microorganisms have the potential to impact on water quality and the hydraulics of an aquifer (cf. Goldscheider et al. 2005). A comprehensive understanding of groundwater biodiversity and related ecosystem function therefore requires a holistic framework including microbial ecology (Danielopol and Griebler 2008).

The following scoping study is one of the first of its kind, assessing the microbiological component of groundwater in Australia. The aim is to target a selection of physiologically relevant groups of bacteria which are a function of the groundwater hydrochemistry and hydrology on the Burdekin floodplain, a tropical coastal plain north-eastern Queensland, Australia. The region has a high value groundwater resource which is accessed by competing users for intensive irrigation of sugarcane, potable supplies to growing local communities and environmental flows. The Burdekin groundwater aquifers are also intimately connected to the river system, surface irrigation and drainages, and are subject to various water quality impacts that require a total system understanding.

Methods

A bulk water sample was taken from each of two groundwater bores of the upper, mid- and lower areas of the lower Burdekin floodplain (Fig. 11 in main report and Table 1) in September 2011. Samples were stored in sterile plastic bottles, wrapped in aluminium foil (see Appendix 1). Using serial dilutions, the water samples were then processed for standard heterotrophic spread-plate counts using R2A agar (HPC, 2 subsamples per bore, results expressed as CFU i.e. colony-forming units; Reasoner & Geldreich 1985; SM 9215C), and anaerobic Fe(III)-reducing bacteria using the dilution to extinction method (FeRB, 3 subsamples per bore; Sorensen 1982; Gould, Stitchbury et al. 2003). Water samples were also 0.45 µm filtered (2 x 100 ml and 2 x 500 ml from each bore) and filters were incubated on mFC agar (Millipore) to detect faecal coliforms (SM 9222 D). Contemporaneous with the water sampling, glass slides were also inserted into the bores, affixed to a nylon fishing line and weighted with lead sinkers. These slides were retrieved 4-6 weeks later and examined by light microscopy at 400x and 1000x magnification to assess the attached and deposited microorganisms.

Results

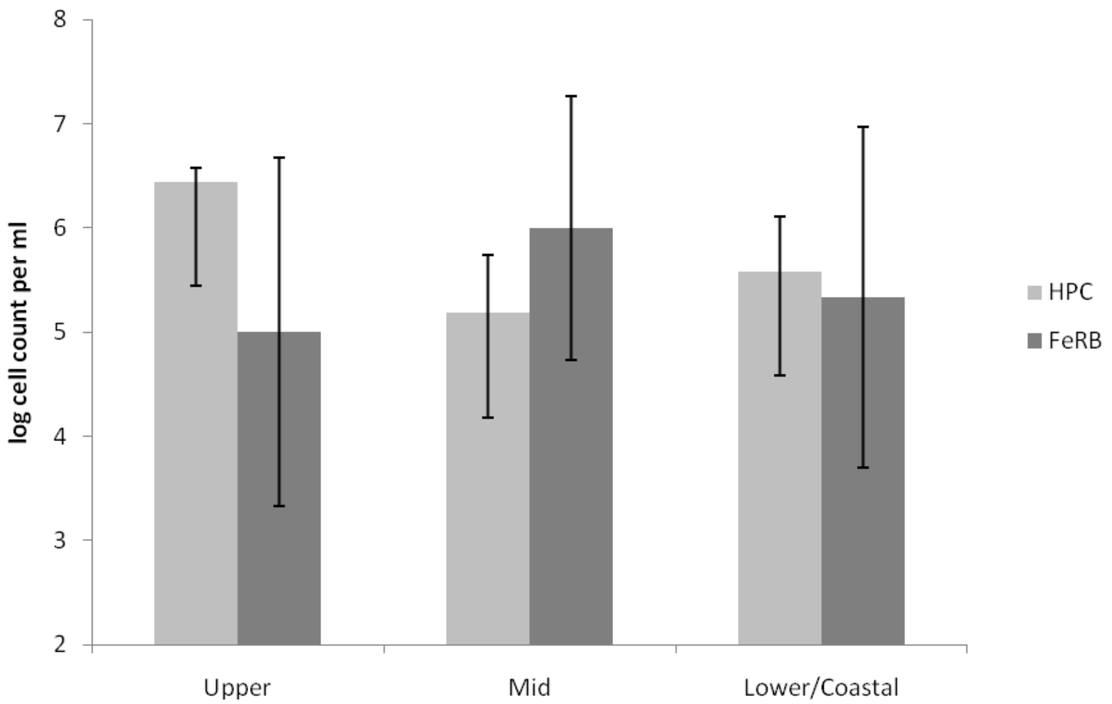

Figure 1. Burdekin groundwater cultivable bacterial enumeration (HPC = heterotrophic plate count, n=2, log CFU mL^{-1}; FeRB = Fe(III)-reducing bacteria, n=3, log cells mL^{-1})

Although there is a high level of variability in the Fe(III)-reducing bacterial counts, ranging from ca. 10^4 to 10^8 cells mL^{-1}, high numbers were present in all groundwater sample sites (Fig 1).

Heterotrophic bacterial counts were less variable, ranging from ca. 4.4 x 10^4 to 3.5 x 10^6 cells mL^{-1} over all sample sites, generally in the range of one log order of magnitude per site. The upper catchment groundwater demonstrated the highest HPC of all the sample sites, 1.8 – 3.5 x 10^6 cells mL^{-1}.

Flocs of organic and inorganic matter encrusted with Fe precipitate were deposited on slides from all groundwater sample sites (Table 1) as were filamentous cyanobacteria. Notably, only the lower/coastal sample sites displayed Fe(II)-oxidising bacterial morphologies, similar to *Leptothrix/Sphaerotilus*.

Table 1. Microscopy of glass slides incubated in situ (also see Appendix 2 for relevant micrographs)

Location	Bore No.	Microscopy – distinguishing features
Upper	11910848, 11910854	Flocs encrusted with Fe precipitate Filamentous cyanobacteria common cf. *Planktolyngbya* Brown, circular protozoan shells common cf. *Arcella* sp.
Mid	11910947, 11910989	Flocs encrusted with Fe precipitate Particulate matter Few filamentous cyanobacteria present cf. *Planktolyngbya*
Lower/coastal	11900201, 11910919	Flocs encrusted with Fe precipitate Large, branching filaments cf. actinomycetes Fe-encrusted tubular structures & precipitation on empty sheaths cf. *Leptothrix*/*Sphaerotilus* Few filamentous cyanobacteria cf. *Pseudanabaena*

Discussion

Anaerobic Fe(II)-reducing bacteria were ubiquitous in lower Burdekin groundwater (Fig 1). This is not surprising considering the high Fe content of Burdekin groundwater (Thayalakumaran, Bristow et al. 2008). Cell counts (10^4 – 10^8 cells mL^{-1}) were similar to the wide range (10^2 – 10^6 cells mL^{-1}) demonstrated by Lin et al. (2011) in groundwater of a managed forest catchment in SE Qld, the only other study of Fe bacteria in Australia. In the aforementioned study, levels of neutrophilic Fe(II)-oxidizing and Fe(III)-reducing bacteria were both correlated with dissolved inorganic carbon. Cell counts in Burdekin groundwater were also similar to a groundwater seep (10^2–10^6 cells mL^{-1}) where rapid Fe redox cycling has been demonstrated (Blothe and Roden 2009). Anaerobic reduction of Fe(III) has been reported as the primary mechanism for organic matter mineralization in coastal groundwater redox transition zones (Chapelle and Lovley 1992; Snyder, Taillefert et al. 2004).

Heterotrophic Plate Counts (HPC, ca. 10^4 – 10^6 cells mL^{-1}) were comparable to those of aerobic, heterotrophic bacteria in contaminated or fouled organic-rich groundwaters (ca. 10^3 – 10^5 cells mL^{-1}), and up to 5 orders-of-magnitude higher than those in pristine shallow unconsolidated aquifers (ca. <10 – 10^3 cells mL^{-1}) (Marxsen 1988; Taylor, Lange et al. 1997; Stuetz and McLaughlan 2004; Ultee, Souvatzi et al. 2004). In the bacterial survey of a groundwater system in SE Qld, Lin et al. (2011) demonstrated a strong correlation between HPC and DOC levels.

The co-existence and ubiquity of high levels of both anaerobic dissimilatory Fe(III)-reducing bacteria as well as aerobic heterotrophic bacteria demonstrate the resilience of these different physiologies and potentially the existence of micro-niches or physicochemical gradients/redox zonation in the aquifers. Together these findings also suggest aerobic heterotrophic- and anaerobic Fe(III)-reducing respiration is of significance to organic matter mineralization in the Burdekin aquifer system.

It is interesting to note that although all groundwater samples contained Fe precipitate, no typical Fe(II)-oxidising bacterial morphotypes were observed deposited on microscope slides from the upper- or mid-floodplain. With groundwater ferrous concentrations of up to ca. 361 mg L^{-1} in some locations within the Burdekin, in particular water drawn from bores of a depth less than 20m, (Thayalakumaran, BristoWab et al. 2008) there is certainly excess substrate for chemolithoautotrophic Fe(II)-oxidising bacteria. Fe precipitate-encrusted bacterial sheaths similar to *Leptothrix/Sphaerotilus* were only observed on the slides retrieved from the coastal groundwater bores. No *Gallionella*-like bacterial stalks were observed, although these are commonly associated with Fe-containing groundwater (Cullimore 2008). It was not within the scope of this proposal to investigate the diversity of these typical but fastidious, Fe(II)-oxidising bacteria through laboratory culture or phylogenetic methods. To speculate, considering the widespread distribution of unicellular chemolithoautotrophic Fe(II)-oxidising bacteria in groundwater samples from another coastal site in Queensland, Australia (Larsen 2010, Lin et al 2011, 2012), it is likely that these bacteria are present in Burdekin groundwater, possibly associated with the Fe precipitate flocs which were seen in all groundwater samples.

Although the examination of glass slides held *in situ* was a qualitative rather than quantitative measure, it gives an indication of organisms common throughout the catchment, and groundwater/surface water interaction (Table 1). The presence of allochthonous cyanobacteria in all groundwater samples suggests there is widespread, dynamic surface and groundwater interaction. These chemoautotrophs require light for energy production and CO2 as a carbon source. As an indicator of water quality, *Planktolyngbya* spp. are commonly associated with mesotrophic reservoirs and are not linked to toxin production (Komarek 1992).

Testate amoebae such as *Arcella* sp., observed from upper floodplain sites (Table 1), are commonly found in association with Fe bacteria. The reason for this is that *Arcella* sp. incorporates Fe/Mn into the cellular structure. The presence of this freshwater protozoan in surface water can be an indicator of surface water/groundwater interaction, although *Arcella* is not directly of concern or associated with disease incidence (Ingram 2004).

The absence of faecal coliforms (results not shown) indicates that the groundwater sites sampled were not impacted by anthropogenic factors related to faecal contamination such as residential wastewater. The impact of faecal contamination due to human activities or wildlife may be localised to point sources and may not be detected unless targeted sampling is carried out. To further assess the possibility of faecal contamination if of concern for potable water, a more persistent bioindicator such as *Giardia* or *Cryptosporidium* could be investigated.

Only the coastal groundwater demonstrated the presence of actinomycete-like bacteria (Table 1). Actinomycetes are known for producing geosmins in potable water supplies, creating taste and odour problems (Zaitlin and Watson 2006). They are a diverse group of filamentous organisms which require organic carbon. Their role in aquatic systems is under-studied and their presence is often attributed to the transport of spores from soils or sediment. These spores may germinate into vegetative cells when in contact with a suitable substrate and contribute to organic carbon degradation.

The preliminary results of this study could be confirmed in a quantitative manner by incorporating a Microscopic Particulate Analysis (MPA; USEPA 910/992-029, 1992) into the Burdekin floodplain hydrogeological and hydrochemical analyses. MPA is used to determine the extent of influence of surface water on groundwater, and comprises the identification, sizing, and population estimates of microorganisms and organic/inorganic matter from a large volume of filtered groundwater (up to 4 500 L).

Conclusion

The Burdekin region represents a highly managed catchment with competing use of groundwater for agriculture and residential consumption, as well as environmental flows. The high bacterial counts from groundwater selectively sampled throughout the catchment and the presence of microorganisms representative of surface water reflects that the groundwater aquifers are not pristine. However, the ubiquity of high numbers of anaerobic Fe(III)-reducing bacteria and aerobic heterotrophic bacteria, which both rely on organic carbon, is indicative of the integral nature of microorganisms in the biogeochemistry of the groundwater and hence the groundwater-dependent ecosystem food web.

Future research should include expanded characterization of the microbial diversity of both aquifer liquid- and solid phase (i.e. bacteria attached to sediment particles), as well as assessment of biogeochemical activity to determine the importance of microorganisms in GDE services. Although not currently included, ideally these techniques and results will be built into the future framework for GDE assessment (NWC 2011).

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Appendix 1. QUT/DERM Groundwater-dependent Ecosystems Proposal

Bacteria of Groundwater-dependent Ecosystems. Dr Eloise Larsen

Background. DERM is addressing knowledge gaps in Groundwater-dependent Ecosystems (GDEs) as part of the National Groundwater Assessment Initiative. Although it is generally accepted that bacteria occur naturally in shallow groundwater (Cullimore 2008) research has focused on the impact on end-use water quality rather than the role of bacteria in natural ecosystems. Systematic studies relating groundwater microbiology to physicochemistry are required to develop an understanding of how bacteria contribute to biogeochemical cycling and therefore, the food web and ecosystem maintenance. We need to know more about the basic ecology and physiology of bacteria in these settings so that we can make better informed decisions in managing the biodiversity and integrity of GDEs. At QUT, in recent work we enumerated bacteria of diverse physiologies in the groundwater system of Poona catchment, SE QLD, (Lin et al. 2011, Larsen 2010). The aim the following project is to target a selection of physiologically relevant groups of bacteria which are a function of the alterations in groundwater hydrochemistry and hydrology in the Burdekin, QLD.

Experimental plan and justification. Given the limited time and budget constraints, this research is restricted to a scoping project. Physiologically relevant groups of bacteria will be targeted through laboratory culture using standard plate counts and most probable number (MPN) techniques in conjunction with microscopy. Considering the variations in site hydrogeology (riparian, delta and coastal) and the wide range of human activities impacting the area (e.g. clearing of native vegetation and input of agricultural fertiliser) three reference (non-impacted) and 3 impacted sites will be identified for analyses. Quantitative heterotrophic plate counts (HPC, SM 9215 B) will be carried out as an indication of the aerobic oligotrophic bacterial community, correlated to the soluble organic carbon content of the groundwater. Considering the high iron (Fe) content of Burdekin groundwater (Thayalakumaran et al. 2008), the ubiquity of Fe(III)-reducing bacteria in other Australian coastal groundwater (Larsen et al. 2010) and their resilience in contaminated aquifers (Lin et al. 2005) this anaerobic group will be enumerated through use of selective culture medium. Their abundance will be affected by groundwater organic carbon content and the redox conditions (Eh and DO), which in turn affect the availability of Fe(III), used as a terminal electron acceptor during bacterial respiration. Determination of chemoautotrophic Fe(II)-oxidising bacterial communities (involved in primary production) is beyond the scope of this study, however, light microscopy will be used to examine biofilm growth on glass slides which have been incubated in groundwater sites. We expect to detect morphologically distinct forms of stalked *Gallionella* and sheathed, filamentous *Leptothrix*. The presence of these organisms will give insight into water flow and O_2/CO_2 gradients. Finally, the detection of thermotolerant coliforms (SM 9222 D), will be used as an indicator of faecal contamination due to human and agricultural activities and potential surface/groundwater interaction.

Table 1. Summary of bacteriological analyses and sampling requirements

Bacterial physiological group	Volume required per sample site	Notes for sampling, sterile bottles required	
HPC	2 x 100ml		Wrap all bottles in foil to protect from light. Keep cool but not refrigerated.
Fe(III)-reducing bacteria	2 x 100ml	Fill bottle to lip	
Faecal coliforms	2 x 100ml, 2 x 500ml		
Biofilm-forming bacteria	2 glass microscope slides	Drill hole in end of glass slide to attach retrieval string	Retrieve after 2-3 weeks, store in sterile container filled with groundwater from sample site

Budget and Milestones. The total budget as advised by DERM is \$5000+GST. At non-commercial-consultancy rates, we costed this project at \$6282. As an investment in producing publishable research outcomes QUT is offering the balance of $1282 as in-kind. This budget includes salary for a Research Associate, use of QUT laboratory, specialised bacteriological analyses, media and consumables, and delivery of a brief final report of 4-5 pages.

Start date	01/10/2011	Payment of $5000 from DERM
Completion date	20/12/011	Project report delivery including spreadsheet of data per bore

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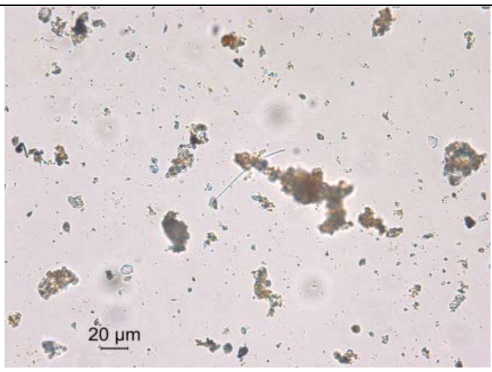
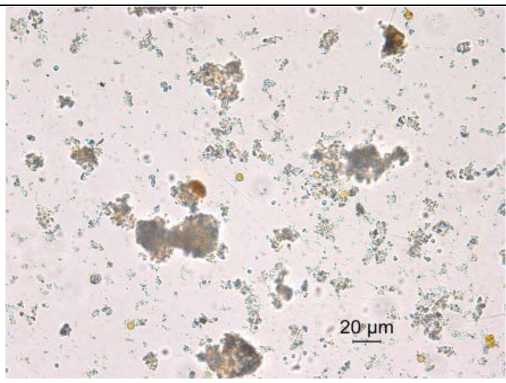
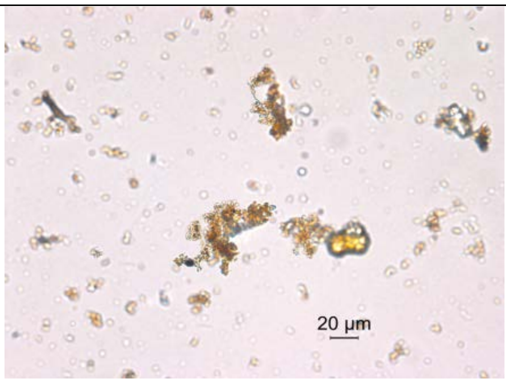
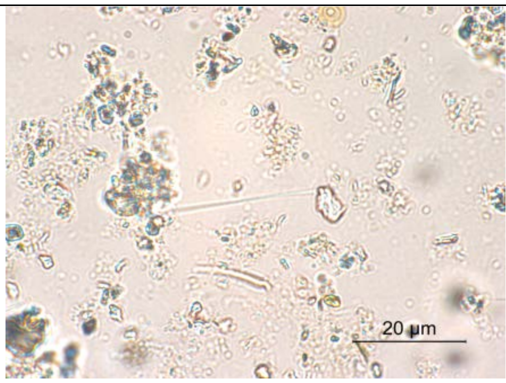
Appendix 2. Micrographs

Table 2. Representative micrographs (light microscopy, 100x and 400x magnification) of microorganisms deposited on slides held in groundwater bores (see Table 1 for description)

Catchment groundwater sample location		
Upper		
Mid		
Lower/coastal