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Final Report

A treatment wetland near Babinda, north Queensland: a case study of potential water quality benefits in an agricultural tropical catchment

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Cover photographs: (front) Babinda wetland on the floodplain; (back) Jaragun staff closing the gate to trap floodwater in wetland. Images: Nathan Waltham, James Cook University.

This report is available for download from the NESP Tropical Water Quality Hub website:
http://www.nesptropical.edu.au

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ACRONYMS

DAWE........... Department of Agriculture, Water and the Environment
DIN................ Dissolved Inorganic Nitrogen
ERT............... Ecologically Relevant Targets
GBR.............. Great Barrier Reef
GBRMPA...... Great Barrier Reef Marine Park Authority
MIP................ Major Integrated Program
N................... Nitrogen
NESP............ National Environmental Science Program
NRM.............. Natural Resource Management
PN................. Particulate Nitrogen
SILO.............. Scientific Information for Land Owners
TWQ.............. Tropical Water Quality
WTWHA........ Wet Tropics World Heritage Area
WTWQIP.......Wet Tropics Water Quality Improvement Plan

ABBREVIATIONS

cm......................centimetre
ha.......................hectare
kg N ha^{-1} year^{-1}..kilograms nitrogen per hectare per year
m........................metre
m^3......................cubic metre
mb.....................standard atmospheric pressure
mg N m^2 day^{-1}...milligrams nitrogen per metre square per day
mm.....................millimetre
^{15}N-N$_2$.................gas flux method
PSII....................photosystem II
µg L^{-1}.................micrograms per litre

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EXECUTIVE SUMMARY

There is a strong scientific consensus that sediments and dissolved inorganic nitrogen in runoff waters from agricultural land in north Queensland present a high risk to the Great Barrier Reef (GBR) ecosystems as well as coastal freshwater wetlands and estuaries. In response, the Reef 2050 Plan has identified a range of measures that are aimed at reducing end-of-catchment loads of sediment and nutrient and recognizes the role that freshwater wetlands may have in achieving this. However, there is currently a limited understanding of the potential for tropical wetlands to process agricultural runoff, and this National Environmental Science Program (NESP) Tropical Water Quality (TWQ) Hub project was set up to obtain the field data necessary to evaluate water quality improvements in a constructed wetland in north Queensland. The wetland covers an area of 10 ha and is located on the Babinda floodplain, in the wet tropics, which drains to the GBR lagoon. Water can enter the wetland either through a large inlet pipe and/or as runoff from the adjacent catchment. There are also 12 manually operated outlet pipes that can be used to control the release of water from the wetland. This wetland is representative of the type of wetlands that can be constructed by farmers, Natural Resource Management (NRM) groups, or councils under the government's water quality initiative.

Water depth, temperature and electrical conductivity were monitored by loggers in two locations in the wetland, from September 2017 to June 2018. These data were used to construct a water balance model for the wetland. Additional field measurements of water and soil denitrification rates were made in June 2018 using an isotopic tracing technique. By combining the water balance model with these local and literature water and soil denitrification data, we have made an initial estimate of the nutrient processing capacity of this wetland.

With the high rainfall (annual average 4287 mm) in the Babinda area, the constructed wetland is frequently filled with water, especially during the wet season. At these times severe flooding in this catchment raises the water level to over 2 m, well above the wetland berm walls, and water flows over the wetland. When the water level drops below the wall height, drainage is slower and this, combined with evaporation, slowly decreases the amount of water in the wetland; taking around 60 days to empty when the outlet pipes are closed. As a consequence, most of the wetland remains inundated for much of the wet season and this enhances the potential for denitrification of the water within the wetland.

Our first estimate of denitrification was made for a simulated single flood pulse event that filled the wetland with water containing 4000 µg L^{-1} of nitrogen (dissolved and particulate). Using a literature derived mean tropical wetland denitrification rate of 427 µg N L^{-1} day^{-1} (86 mg N m^2 day^{-1}), we estimate that all of the nitrogen in the wetland water is removed in 26 days (42% as gaseous denitrification, 21% as sedimentation of particulate nitrogen and 37% as drainage). So, if no further water enters the wetland, the best management option for maximising nitrogen removal is to drain the wetland after this time, (note this may not be possible when the surrounding area is inundated with flood water). If all of the outlet pipes are opened most of the water can drain from the wetland in only one day. However, unless the wetland has to be drained this rapidly, it is not necessary to have 12 outlet gates and reducing this number could make considerable cost savings. Some of this saving could be used to install remotely operated gates, as this would overcome some of the serious safety (crocodiles) and access (roads submerged) problems with getting to the site during periods of deep flooding. Clearly, the outlet gates are also needed to drain the wetland for maintenance exercises, such as desilting, removal of weeds and repairs to the berm walls and gates.

A second estimate of wetland denitrification was made for the 12 month period from October 2017 to September 2018 using depths measured in the wetland. In this year we estimate that 2024 kg of nitrogen entered the wetland, in a series of 8 flood pulses that completely filled the wetland on each occasion. Using the mean tropical wetland denitrification rate of 427 μg N L^{-1} day^{-1}, we calculate that most (44%) of this nitrogen left the wetland as drainage, with 36% lost to the air as denitrification and 20% settling on the soil as sedimentation of particulate nitrogen. Drainage is likely to be mostly as seepage through the berm wall, with a relatively small contribution from deeper groundwater movement. The fate of the nitrogen in the drainage water will depend on the denitrification conditions in the surrounding ditches, rivers and aquifer, hence further research is needed to determine these along with what the ecological impacts are in these environments. However, these processes happen outside the constructed wetland, and are not part of its processing effect. The large loss of nitrogen as drainage emphasises the importance of taking into account the actual wetland hydrology, which affects the dynamics of the water depth (and hence area and volume) and nitrogen concentration. This is clearly vital to obtaining the correct nitrogen processing capability of wetlands.

The above annual loss of nitrogen equates to a total denitrification rate of 116 kg N ha^{-1} $year^{-1}$. There is the potential to increase this figure by 18 kg N ha^{-1} $year^{-1}$ by repeatedly pumping drainage ditch water into the wetland during the dry season, but this management option comes with the additional cost of the pumps and fuel to run them.

The above estimates of nitrogen loss in the wetland are uncertain due to the wide range of published water denitrification rates and the lack of nitrogen concentration data for the water entering the wetland. Other influential parameters such as dissolved oxygen levels and sediment concentration have also been taken from the literature and ideally should be measured *in situ*. Nevertheless, a single point comparison of the modelled and measured nitrogen concentration in the wetland indicated that a water denitrification rate of ~ 93 μg N L^{-1} day^{-1} may be closest to the effective long-term (annual) value in the Babinda wetland. Further measurements of nitrogen and dissolved oxygen concentrations in the wetland at a range of times after flood pulses would be needed to validate and/or improve the current model.

The Reef 2050 Water Quality Improvement Plan has set target reductions for dissolved inorganic nitrogen (DIN) and particulate nitrogen (PN) loads for each catchment that drains to the Great Barrier Reef. For the Mulgrave-Russell a reduction in DIN of 300 tonnes and PN of 53 tonnes by 2025 is recommended. 10% of the DIN reduction target could be achieved from 398 ha of wetland with the same mean denitrification properties as the Babinda wetland; which amounts to 1.5% of the total sugarcane area in the Mulgrave-Russell catchment. This area of wetland would also remove 30% of the 2025 PN reduction target. If the wetland denitrification rate was as high as the spot measurements made in the Babinda wetland (768 μg N L^{-1} day^{-1}, the area of wetland required to make a 10% contribution to the DIN reduction target would reduce to 327 ha, or 1.3% of the sugarcane area.

This study gives some preliminary estimates of the sediment and nitrogen processing capacity of a constructed wetland in a high rainfall region of north Queensland. They are currently highly variable, so we have identified the key parameters that need to be measured in order to improve long-term wetland processing capacity estimation. These include further monitoring of the key components of the wetland water balance (especially seepage and groundwater) and additional *in situ* measurements of soil and water denitrification rates and how they vary seasonally.

1.0 INTRODUCTION

1.1 Water quality challenges in Great Barrier Reef catchments

The loss and modification of catchment ecosystems is contributing to increased loads of nutrients and sediments entering the Great Barrier Reef (GBR) lagoon (Brodie et al. 2010, Brodie and Pearson 2016). Broad scale vegetation clearing and catchment urban, agricultural and industrial development (Waltham and Sheaves 2015) has led to the widespread loss and degradation of freshwater wetlands, forested floodplains, woodlands, rainforests and other terrestrial and aquatic ecosystems in the catchment (GBRMPA 2009, Brodie et al. 2013, Adame et al. 2019b, Waltham et al. 2019). These ecosystems are essential for a healthy, resilient GBR because they can offer some assistance in trapping catchment sediments and nutrients (Adame et al. 2019a), slow surface water flow, improve coastal hydrological connectivity and provide habitat for a range of freshwater and marine species (Sheaves et al. 2012, Pearson et al. 2013, Great Barrier Reef Marine Park Authority 2014, Arthington et al. 2015).

Declining reef health will potentially have a significant impact on the environmental values and economic return generated by the GBR as an asset. Stoeckl et al. (2014) conclude that changes in the environment in the GBR would have a major impact on national and regional economies. Those authors found, through a major survey of visitors and residents, that degradation of environmental values would have real impacts in the tourism industry, including reductions in tourist satisfaction, reduced numbers of tourists visiting the region, reductions in the length of visits, and fewer repeat visits. Mustika et al. (2016) examined the potential implications of environmental deterioration for business and non-business visitor expenditures in the GBR. The authors concluded that nature-based tourism is an important source of income for the region. 90% of visitors came to the region for at least one nature-related reason, and that substantial environmental degradation could reduce visitor expenditure, and thus local tourism income, by at least 17%.

The impact of agricultural runoff on coral reefs is not limited to the GBR (Roebeling et al. 2011), with around a quarter of the total global reef area (Burke 2011), and a range of other aquatic ecosystems impacted by agricultural pollution (Verhoeven et al. 2006, Flanagan and Richardson 2010). Countries around the world have implemented programs in an attempt to address the impact of agricultural pollution on aquatic ecosystems. These have included the regulation of nitrogen fertiliser use on crops (Kronvang et al. 2008), soil and water conservation (Chu et al. 2009), reduced livestock stocking density (Kronvang et al. 2008), and conversion of agricultural land to alternative production systems or natural ecosystems (Frisvold 2004). The management of diffuse pollutants from agricultural land uses is therefore a key issue throughout the world and management approaches, including the construction of strategically positioned treatment wetlands, implemented in other countries may also be applicable in the GBR catchments (Waltham et al. 2017).

There is a need for new, innovative approaches to reduce nitrogen loads that are cost effective, and provide longer term viability (and maybe profitability) for land owners (Waltham et al. 2017). The Wet Tropics Water Quality Improvement Plan (WTWQIP) identifies that complete adoption of sugarcane best management practices would be insufficient to achieve the nitrogen load reductions needed to meet the GBR water quality guidelines and targets. This project, with project partners and end users, examines the potential benefits that a constructed treatment wetland can provide in a sugar cane dominated catchment in the wet tropics of north Queensland. In this project, we follow the program logic outlined in the Queensland Government's Whole-of-Catchment approach, where the Babinda treatment wetland has been positioned in the drainage network of the catchment, designed to treat water in the main Babinda drain (DEHP 2016).

1.2 Wet Tropics Region

The nine major river basins comprising the Wet Tropics NRM Region have been highly modified since European settlement, with extensive vegetation clearing and hydrological modifications to facilitate agricultural production (Terrain NRM 2015). Intensive agriculture dominates the coastal floodplain and grazing is the major land use in the drier, western parts of the region. Sugarcane production is the dominant intensive agricultural land use on the coastal floodplain and accounts for approximately 80% of the anthropogenic loads of dissolved inorganic nitrogen (DIN), and over 95% of the photosystem II (PSII) herbicide load to the GBR lagoon in the Wet Tropics region (Terrain NRM 2015) (Figure 1). In terms of DIN load from sugarcane, the highest loads per unit area are from the Russell Mulgrave, Johnstone, Tully and Murray catchments (Hateley et al. 2014, Waterhouse et al. 2016).

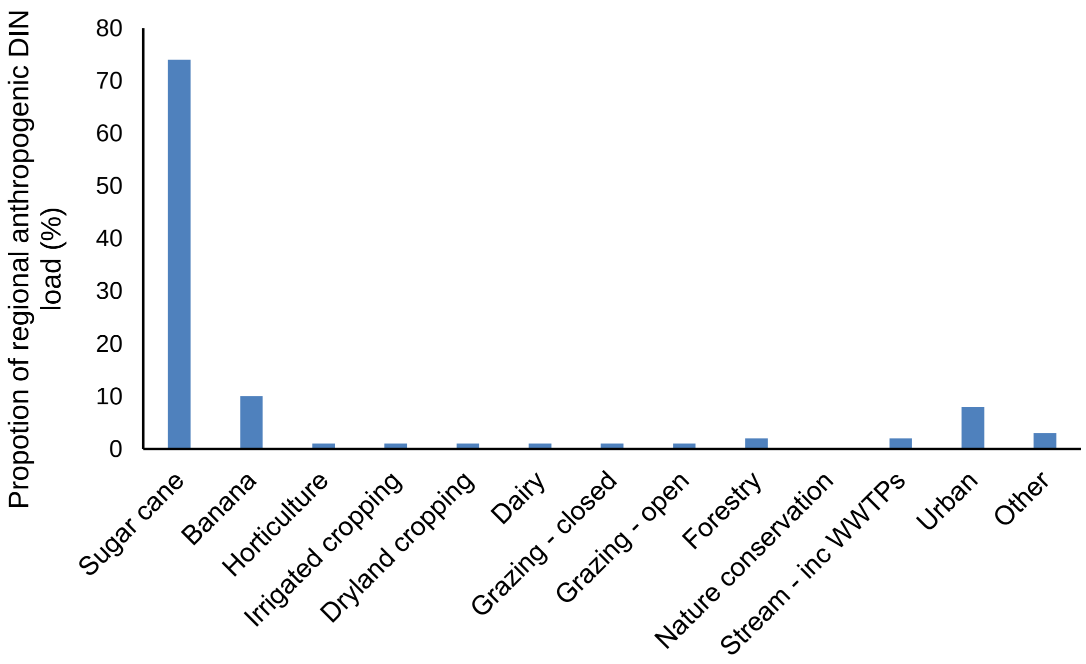

Figure 1: Land use contribution to anthropogenic DIN loads for the Wet Tropics region (2008 baseline data) (Hateley *et al.,* 2014)

The Wet Tropics Water Quality Improvement Plan 2015-2020 (WTWQIP) has been 'designed to identify the main issues impacting catchment waterways and the coastal and marine environment from land-based activities, and to identify and prioritise management actions that will halt or reverse the trend of declining water quality within the Wet Tropics region' (Terrain NRM 2015). The WTWQIP defined Ecologically Relevant Targets (ERT) for the pollutant load reductions needed to meet GBR water quality guidelines. These are defined for each river basin and vary from 50% to 80% reduction in DIN (from the modelled 2008 baseline estimate) by 2035 (Brodie et al. 2016).

The WTWQIP also aims to restore the ecological function of the landscape through system repair, outlining values, threats and actions for managing freshwater and coastal ecosystems vital for the health of the GBR. Almost half of the palustrine wetlands in the Wet Tropics NRM region have been drained, filled and developed (Department of Environment and Heritage Protection 2016). Drains and levees have been constructed in many agricultural areas to drain or divert surface water and these have also impacted natural hydrological processes and connectivity (Bruinsma 2001). These modifications have altered landscape processes and functions, reduced water detention time, and the trapping and processing capacity of the floodplain and increasing water velocity, erosion and pollutant transport (Brodie et al. 2004, Great Barrier Reef Marine Park Authority 2014).

The WTWQIP identifies that the adoption of current best management practices in sugarcane will only go part of the way towards meeting the DIN water quality targets - it predicts 19% DIN load reduction for all B management practices (best practice) and 30% DIN load reduction for A management practices (innovative/aspirational practices) (Hateley *et al.*, 2014). It explains that the restoration of ecological functions in coastal ecosystems and floodplains is needed, although the benefits of such actions are yet to be quantified (Terrain NRM 2015). Similarly, the Alluvium (2016) report examining the costs of meeting the Reef 2050 Plan water quality targets found that under current policy options the target for DIN reduction in the Wet Tropics Region could not be met and alternative policy responses would be needed (p.34). The WTWQIP did not assess alternative policy options such as land use change or land buy-back options in the cost-benefit analysis due to 'insufficient availability of information to make appropriate cost estimates, and the potentially high socio-political risk that is likely to be associated with these options at this time' (Terrain NRM 2015). Alluvium (2016) investigated changing 50% of sugarcane land under D class management (i.e., poor management with high risk of pollutant loss (2,210ha) to either biodiversity conservation or grazing - no fertiliser application). The modelling showed minimal DIN reduction from this land use change, due to the small area of D class management in the Wet Tropics Region. This suggests that limiting land use change to D class management land constrained the opportunity for this policy option to contribute more considerably to reducing DIN loads. For this reason, and the lack of commensurate information available on other agricultural industries in the Wet Tropics, DIN load reduction from sugarcane production is a key focus of the WTWQIP.

1.3 Transitioning low lying, high DIN risk, sugarcane land to treatment wetlands

Transitioning high DIN risk sugarcane land to an alternative land use that has a lower nitrogen input requirement or increased ability to process excess nitrogen is an alternative policy response outlined in the Reef 2050 Plan targets. Kroon *et al.* (2016) argue that long term reduction in nitrogen export to the GBR lagoon could be assisted by replacing current high-input crops with other crops suited to the climate and soils of the area which require lower nitrogen fertiliser input. These authors also identify that land-based pollution reductions can be achieved through hydrological restoration of landscapes (Kroon *et al.,* 2016). More recently, a companion NESP TWQ Hub project (Project 2.1.2) examined whether low-lying, high DIN risk, sugarcane locations could transition to a lower DIN risk land use and achieve water quality improvements for the reef. However, the opportunity to test and refine this idea is needed (Waltham et al. 2017). The Wet Tropics Major Integrated Program (MIPs) is also trialling a series of treatment wetland designs, with the data available in the coming years (https://terrain.org.au/projects/wet-tropics-major-integrated-project/).

There are several potential reasons why land use conversion may be an economically and environmentally rational response, particularly in low-lying, high DIN risk, sugarcane production areas. These areas are noted (but not in every case) for being unreliable from a production perspective and provide consistently difficult drainage and management challenges for landholders. Low-lying sugarcane lands generally have lower productivity and require greater inputs of fertiliser, and pesticide, to meet desired yield targets (Roebeling *et al.,* 2007). It's also difficult to establish and harvest crops in these areas, high rainfall and inundation could potentially result in total crop loss, and they are likely to be significant sources of high nitrogen loss (Roebeling *et al.,* 2007). Such sugarcane lands are commonly referred to as 'marginal' (Waltham et al. 2017).

The report of the Great Barrier Reef Water Science Taskforce (The Great Barrier Reef Water Science Taskforce and the Office of the Great Barrier Reef Department of Environment and Heritage Protection 2016) makes reference that the sugarcane industry could be facing greater regulatory burden in order to bring nutrient pollution levels in line with established targets. Transitioning low-lying, high DIN risk, lands that are comparatively difficult to farm, low yielding and leaky from a nitrogen perspective, with an alternative land use, could therefore be an option that is of benefit to the farmers, and to receiving waters in the GBR Lagoon (Waltham et al. 2017).

1.4 Russell Catchment - Babinda

The Russell catchment is located south of Cairns and is approximately 669km^2 in area (amounting to 3% of the Wet Tropics region) (Figure 2a and b). Prior to development the vegetation was mainly dominated by rainforest and scrubs, with small areas of melaleuca and eucalypt woodlands and forests. This vegetation community and extent is thought to play an important role in slowing the catchment water flow, retaining it longer in the landscape and recharging the aquifers, which may also provide some water quality benefits. Current land use on the floodplain, however, is dominated with sugarcane (~13%), grazing (~8%), urban (~3%) and minor areas of other cropping and services (Wetlandland Info).

Soils in the catchment are highly variable (Killin, 2011) and across the floodplain mainly fertile alluvial soils. Adjacent to Babinda Creek they are quite poorly drained Tully, Coom and Timara soil types. Some further details of the soil at the constructed wetland site are given in the results (Section 3.2).

Babinda creek is a major tributary in the Russell catchment. The creek rises in the Bellenden Ker Range between two of Queensland's highest mountains, Mount Bellenden Ker (1593 m) and Mount Bartle Frere (1622 m). The upper reaches of the catchment are protected as part of the Wooroonooran National Park in the Wet Tropics World Heritage Area (WTWHA) (Russell Landcare & Catchment Group 2001). Once water flows from the upper reaches, it reaches the coastal floodplain, which has been extensively modified and drained for agricultural expansion. Land use on the floodplain is dominated by sugarcane, where the creek continues to flow, picking up water from a number of constructed drainage lines, before meeting the Russell River and the Coral Sea.

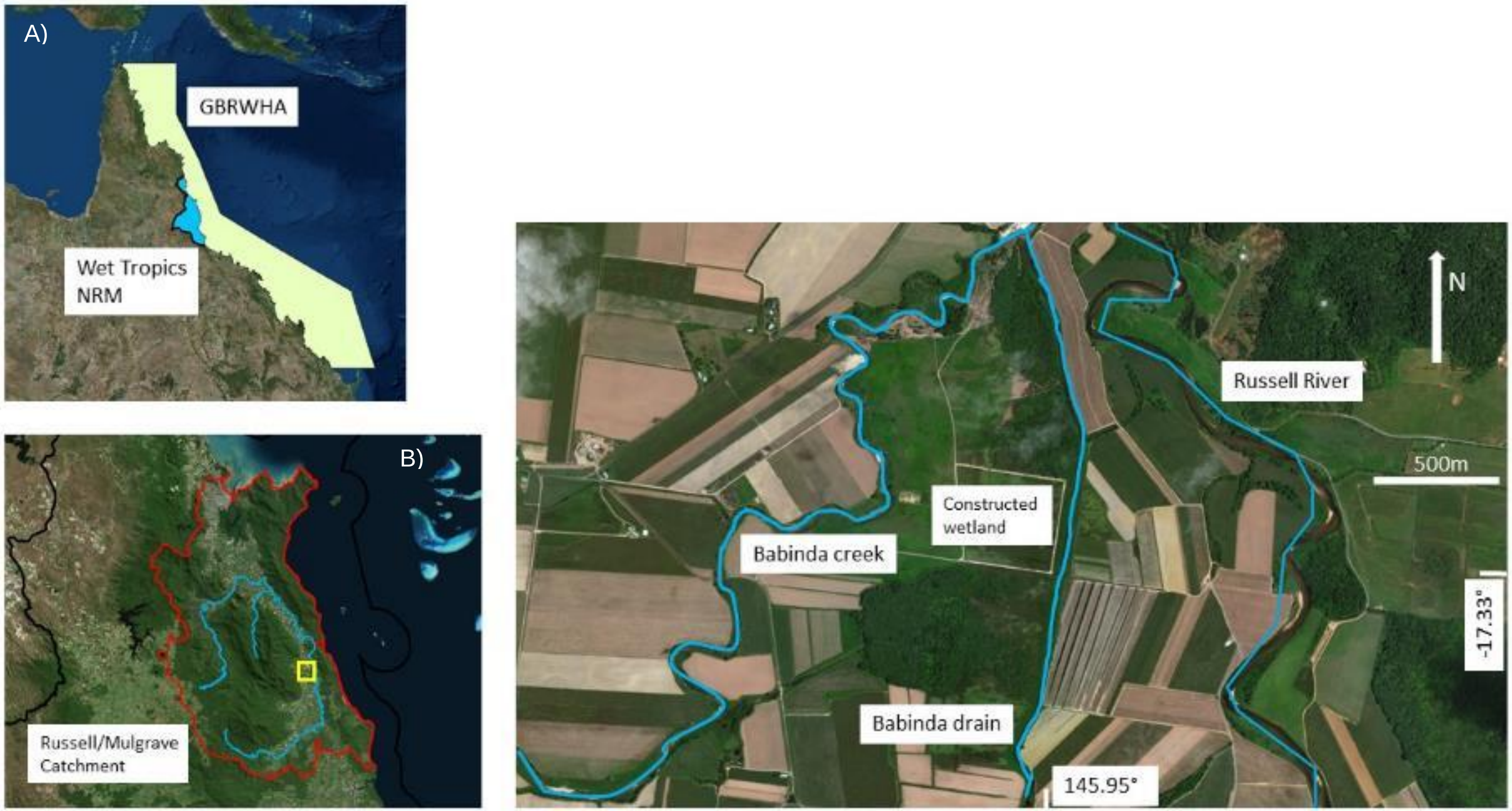

Figure 2: A) location of the Great Barrier Reef World Heritage Area (GBRWHA), and the Wet Tropics NRM; B) Russell/Mulgrave catchment showing the location of the constructed wetland (yellow box); and C) location of Babinda treatment wetland adjacent to Babinda drain

Rainfall has been recorded daily at the Babinda Post office (Station 31004) since 1911. Rainfall is strongly seasonal, with an average 70% of total annual rainfall falling between October and March each year. Analysis of the data (Figure 3) shows that the highest accumulative wet season (November to March) rainfall occurred 2010/2011 (5351mm) while the lowest was 1995/1996 (775 mm). Annual summer rainfall totals recorded in the two years (2015/16 and 2016/17) prior to the study were below long term average while rainfall total experienced during this study is within the 70% percentile (Table 1).

Table 1: Summary wet season (Nov – March) statistics of rainfall recorded at Babinda Post Office

Statistic	Wet season rainfall (mm)
Mean	2731.6
Median	2684.0
95th percentile	4518.7
5th percentile	1232.1
2015/16 wet season total	2285.0
2016/17 wet season total	1970.2
2017/18 wet season total	3664.8

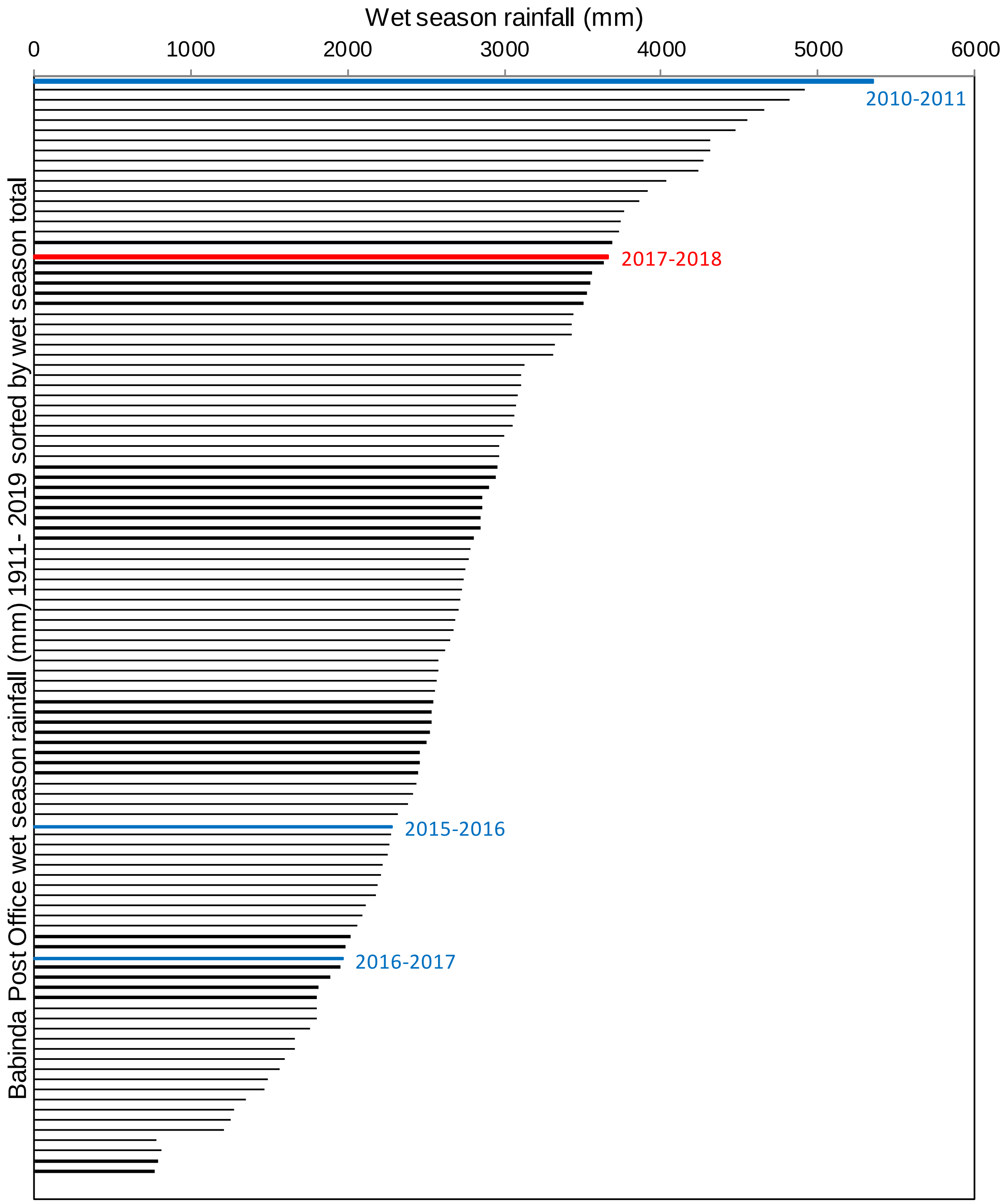

Figure 3: BOM wet-season (Nov - March) rainfall data recorded at Babinda Post Office (station number 31004) ranked in order of decreasing total rainfall (mm). Blue bars show total rainfall over the past few years, red bar cover wet season during this study

1.5 Improving water quality with treatment wetlands

Catchment land use modification and disturbance has contributed to increased delivery of sediment and nutrients reaching estuaries and near shore coastal waters in many places (Balls 1992, Davies and Eyre 1998, Bainbridge et al. 2014, Davis et al. 2017). In such instances, excessive supply can have major negative consequences on downstream services, such has critical nursery areas for fish production (Loneragan and Bunn 1999, Grange et al. 2000, Gillanders and Kingsford 2002), hypoxia and algal blooms (Breitburg et al. 2018), and impacts on benthic habitat areas such as seagrass or coral ecosystems (Abal and Dennison 1996, York et al. 2015). The need for whole of catchment management protection and restoration planning is a major interest to coastal managers challenged with achieving (and protecting) ecosystem conservation while also allow more development to occur. A push for restoration of coastal wetland and estuaries is likely to increase within the next few years, particular in response to the United Nations Declaration of a decade on ecosystem restoration (2021 to 2030) (Waltham et al. 2020).

Improvement in coastal water quality conditions can be linked to improvement in broader ecosystem values and services in the coastal seascape (e.g. improvements in turbidity increase light availability which is important for aquatic plant photosynthesis). One way of improving water quality in modern day catchment areas is by constructing treatment wetland systems that are designed and positioned in the landscape to intercept sediments, nutrients and other pollutants (e.g. pesticides, heavy metals) before reaching adjacent sensitive receptor habitats (usually freshwater creeks, estuaries or coastal areas) (Valiela and Costa 1988, Batson et al. 2012, Mitsch et al. 2013) (Figure 4). There are many factors known to affect the processing capacity of wetlands. One foremost factor is hydrology, which unlike in wastewater treatment situations that benefit from consistent flow and nutrient concentrations (Jordan et al. 2003), wetlands in the landscape are known to have complex spatial and temporal hydrological patterns. In addition to hydrology, nutrient loading (Blahnik and Day 2000), available oxygen in soil and water column (Land et al. 2016), and the presence of vegetation to absorb nutrients (Greenway 1997) also contribute in complex ways to the processing efficacy of a wetland, and quantifying these assists with understanding the maximum possible services provided by wetlands (Mitsch and Gosselink 1993, Mitsch et al. 2005, Land et al. 2016). While there is, therefore, interest in the role that nutrient uptake in treatment wetlands might play in further contributing to delivering on water quality improvements in GBR catchments (Waltham et al. 2017), and conceptual models of processes are instructive (Figure 4), though access to quantitative data is limited – this is a major research opportunity requiring attention.

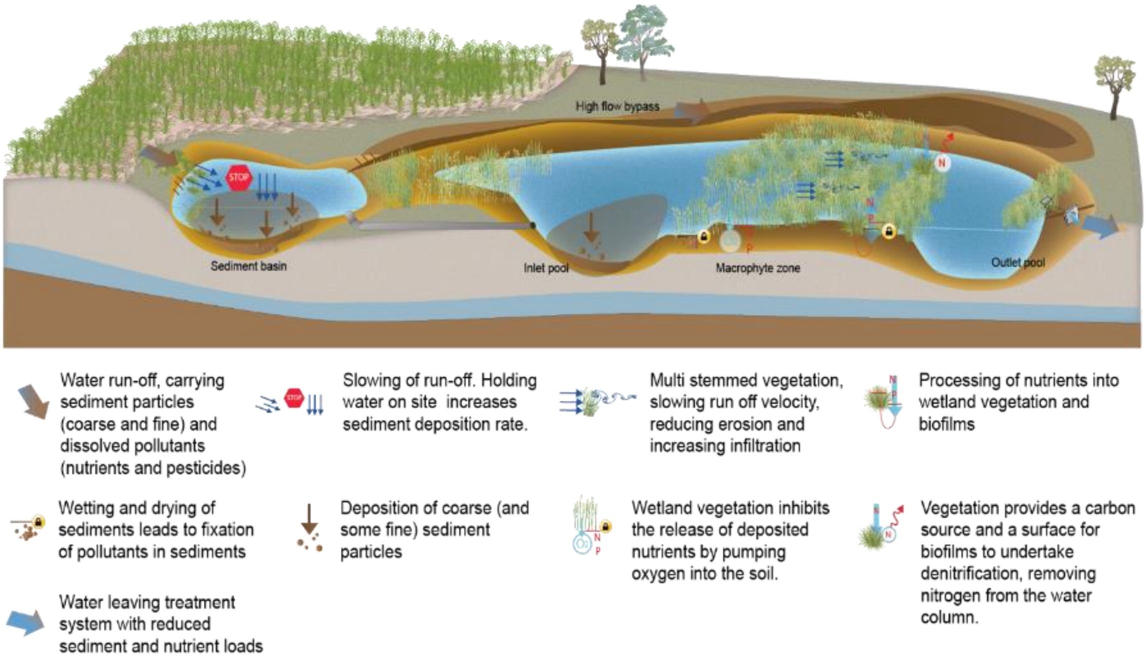

Figure 4: Example of treatment train wetland conceptual diagram (Wetland*Info*, Queensland Government, accessed November 2019).

1.6 Project outline

In response to the lack of understanding regarding the potential for tropical water treatment wetlands to process agricultural runoff, this National Environment Science Program (NESP) Tropical Water Quality Hub project has partnered with Jaragun NRM to collect field data to evaluate a constructed wetland on the Babinda floodplain in north Queensland. The selected wetland is located on the Babinda floodplain, in the wet tropics, which drains to the GBR lagoon. This report presents data collected in the field to establish a water balance model for this wetland, in addition to field data to calculate water and soil denitrification rates. By combining the water balance model and denitrification data, this project then quantifies the processing capacity potential of this wetland using spreadsheet models. One outcome of this project is to document the learnings from this project, including determining management and maintenance requirements for this type of constructed wetland. This project is representative of the type of wetlands to be constructed by landholders under the government's desire to improve water quality via a private partner incentive scheme that is on the horizon in Australia.

2.0 METHODOLOGY

2.1 Babinda treatment wetland

The land use prior to the constructed wetlands being in place was used for cultivation of sugarcane, but was abandoned several decades ago due to low productivity due to flooding and high water table. The site is largely dominated by the introduced paragrass (*Urochloa mutica*) with large areas of singapore daisy (*Sphagneticola trilobata*) and guinea grass (*Megathyrsus maximus var maximus*). Common sensitive plant (*Mimos pudica*) also covers lard areas, in addition to plant pond apple (*Annona glabra*). Some native species are present on the site including the native fern (*Diplazium dietrichianum*), native sedges nutrush (*Scleria sumatrensis*) and bogrush (*Schoenoplectus mucronatus*), and some remnant palm forests nearby. As part of this project site, but not measured here, including the establishment of a nursery to propagate sedges for the wetland and replanting of the site (Jaragun 2014).

The design of the Babinda water treatment wetland is different to others that are currently planned in the region. This wetland is an off channel facility, where water flowing down the Babinda drain (Figure 5) can be diverted into the wetland once the drain level is above the inlet pipe height. A one-way manual value can be opened allowing water from the drain to fill the wetland (Figure 6a and b). This is not the only way the wetland could receive water as catchment runoff from the west can run freely into the wetland, overtopping the drain banks, or crossing paddocks from Babinda creek; in addition to direct rainfall inputs (AWC 2015).

There are 12 outlet pipes located along the northern wall of the wetland (Figure 5) in addition to an outlet weir. The 12 outlet gates are also manually operated (Figure 6 c and d) to control the release of water and draining of the wetland.

Figure 5: Babinda constructed wetland showing location of the inlet control structure between the wetland and the Babinda drain, position of the diver loggers (Logger 1, Logger 2) recording water depth, water temperature and conductivity, the outlet weir, and the twelve manually control outlet pipes

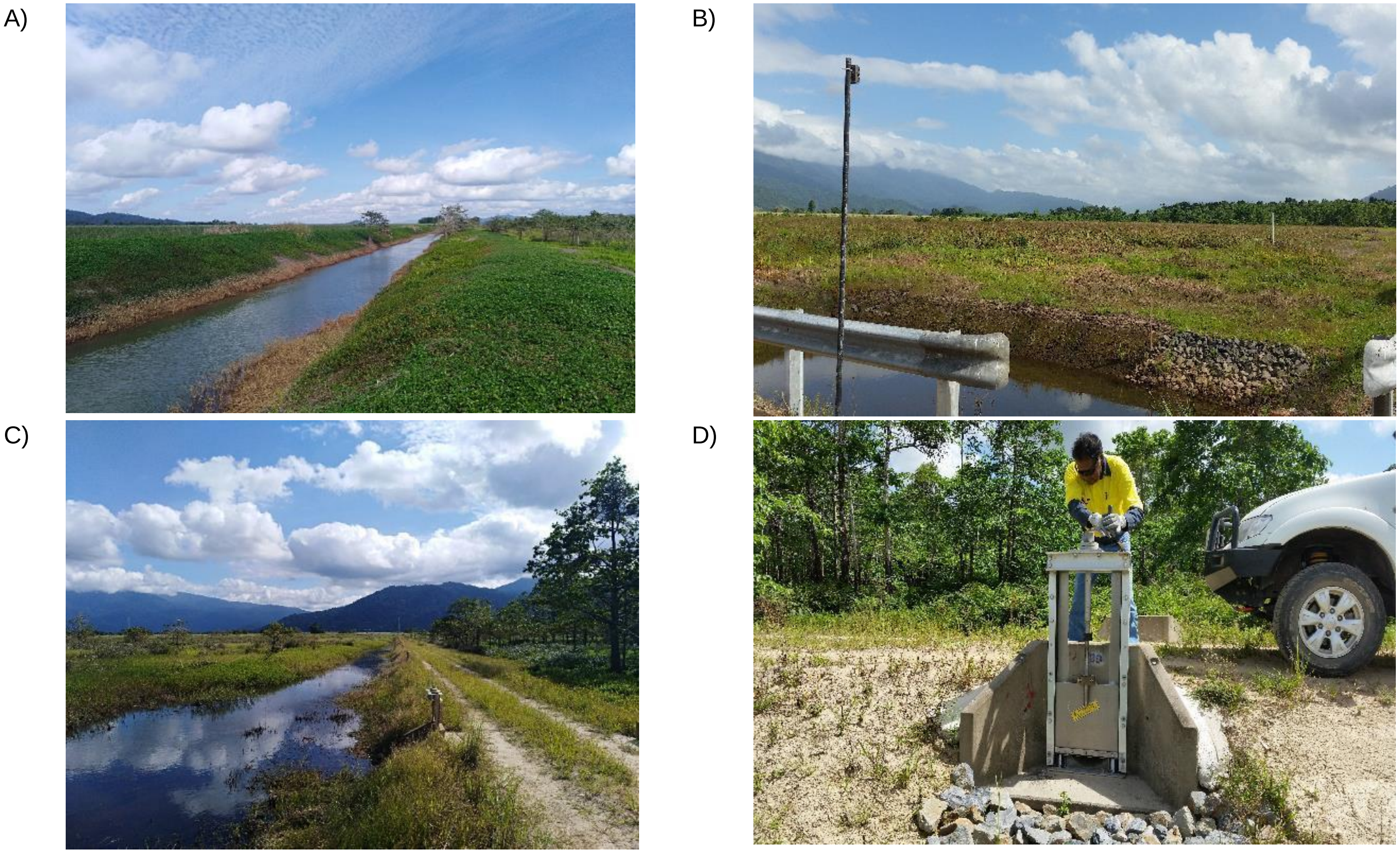

Figure 6: Babinda constructed wetland showing location of the inlet control structure between the wetland and the Babinda drain, position of the diver loggers (Logger 1, Logger 2) recording water depth, water temperature and conductivity, the outlet weir, and the twelve manually control outlet pipes

2.1.1 Wetland bathymetry

Prior to the construction of the Babinda wetland, the site topography was surveyed (AWC 2015). The data (redrawn) is presented in Figure 7 and shows that the land within the wetland slopes from east to west, rising by 0.85 m over a distance of ~ 320 m. The different transects across the wetland show that the land surface depth below the berm wall also varies in the south – north direction (Figure 7b), and that the maximum depth of water that the wetland can retain is 0.85 m. The presence of acid forming soils (Morrison 2014) were surveyed, in addition to vegetation communities (Mitchell 2014) at this wetland site prior to commencement of earth works.

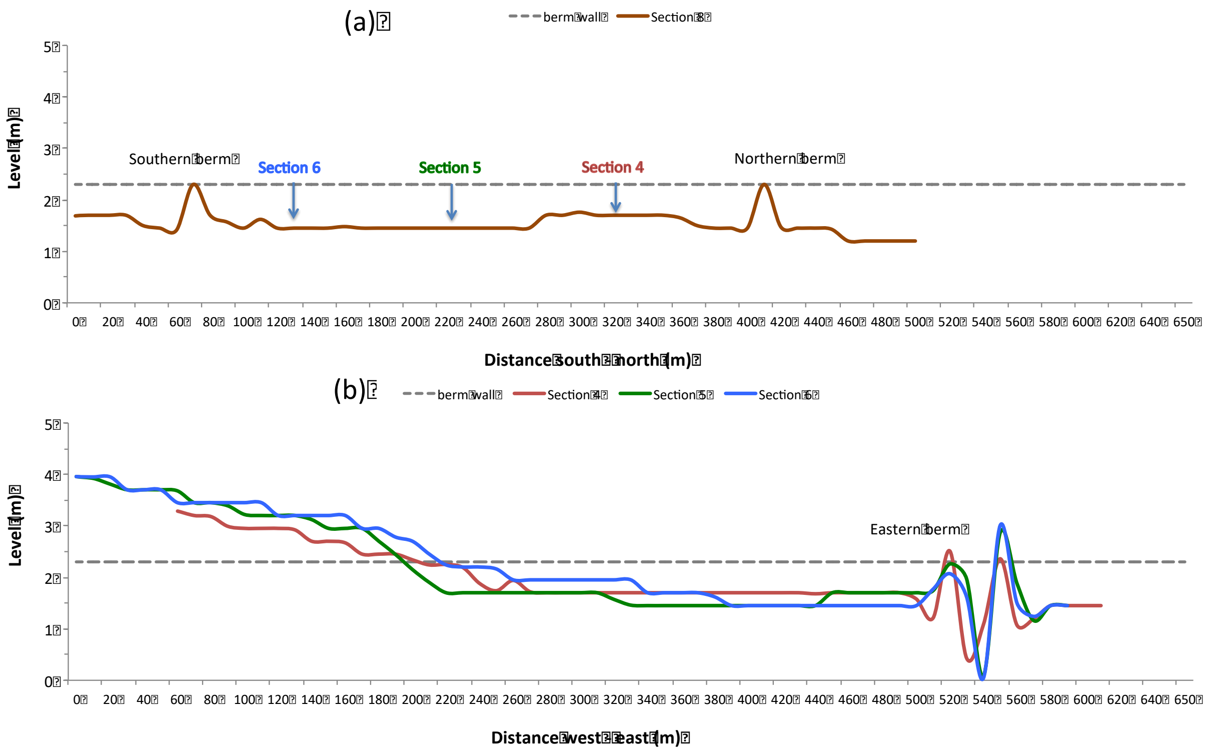

Figure 7: Ground level surveys across the Babinda wetland (a) south – north and (b) three transects west – east. The height of the berm wall is also shown

These transect data can be used to calculate how the area of the wetland, and the volume of water within it change with wetland depth (Figure 8). Wetland area increases with wetland depth, reaching 105,400 m^2 (10.5 ha) when the wetland is filled to the top of the berm wall. The irregular shape of the area / depth relationship is due to variations in ground level in the wetland, most notably the construction of deep drainage channels to allow for rapid escape of floodwater while the property was under cane. Wetland volume increases more smoothly with depth with the maximum volume being 63,251 m^3.

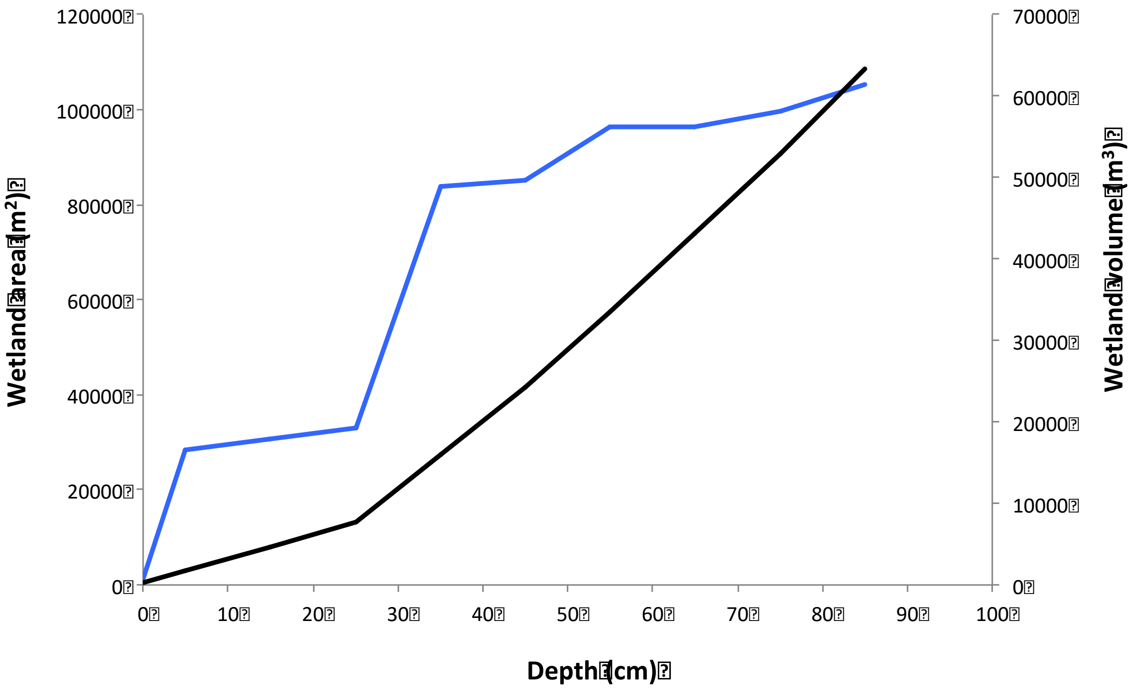

Figure 8: The change in wetland area (blue) and volume (black) with water depth

The above relationships between wetland area, volume and depth are used in the calculation of wetland denitrification as the soil and water nitrogen loss rates are expressed per unit area and volume of wetland respectively (see Section 3.4) (Figure 8).

2.1.2 Water quality - high frequency loggers

Water depth, temperature and electrical conductivity were monitored by loggers (CTD-Diver, Eijkelkamp Soil & Water, Netherlands) located in two permanent positions in the wetland, beginning in September 2017. The locations were adjacent to the inlet structure (Logger 1), and near to the outlet structure (Logger 2) in the north eastern corner of the wetland (see Figure 5). The loggers captured data from the bottom of the water column (~ 10 cm above the soil surface) every 20 minutes and were downloaded as part of routine maintenance visits. Each logger was installed inside a PVC pipe (3m height, 90mm diameter; Figure 9) that was attached to a steel star picket. Loggers were attached to a stainless steel wire cord that was attached to the top of the PVC pipe for easy retrieval (downloading the data and maintenance). The loggers were downloaded every few months (between October 2017 and June 2018). Water quality samples are collected in the Babinda drain, adjacent to the wetland site, as part of NESP TWQ Hub Project 25. Some data around the time of the logging period were access for the modelling here. These data are not reproduced here but can accessed by contacting the authors herein.

Figure 9: CTV Diver logger configuration in Babinda wetland

2.1.3 Time lapse cameras

Two trail cameras were installed in the wetland to record local conditions occurring in the wetland. Both cameras were attached to a steel pole that extended approximately 3m above the surface of the berm wall. The first camera was located on the inlet structure facing north, while the second camera was located to a pole installed near to release gate 12. Cameras were programed to take a photograph between 6-8am, and 4-6pm each day. Cameras were maintained approximately every 3mths, to ensure batteries and the SD card were operational. The cameras captured major rainfall events and filling of the wetland that occurred early 2018 (Figure 10).

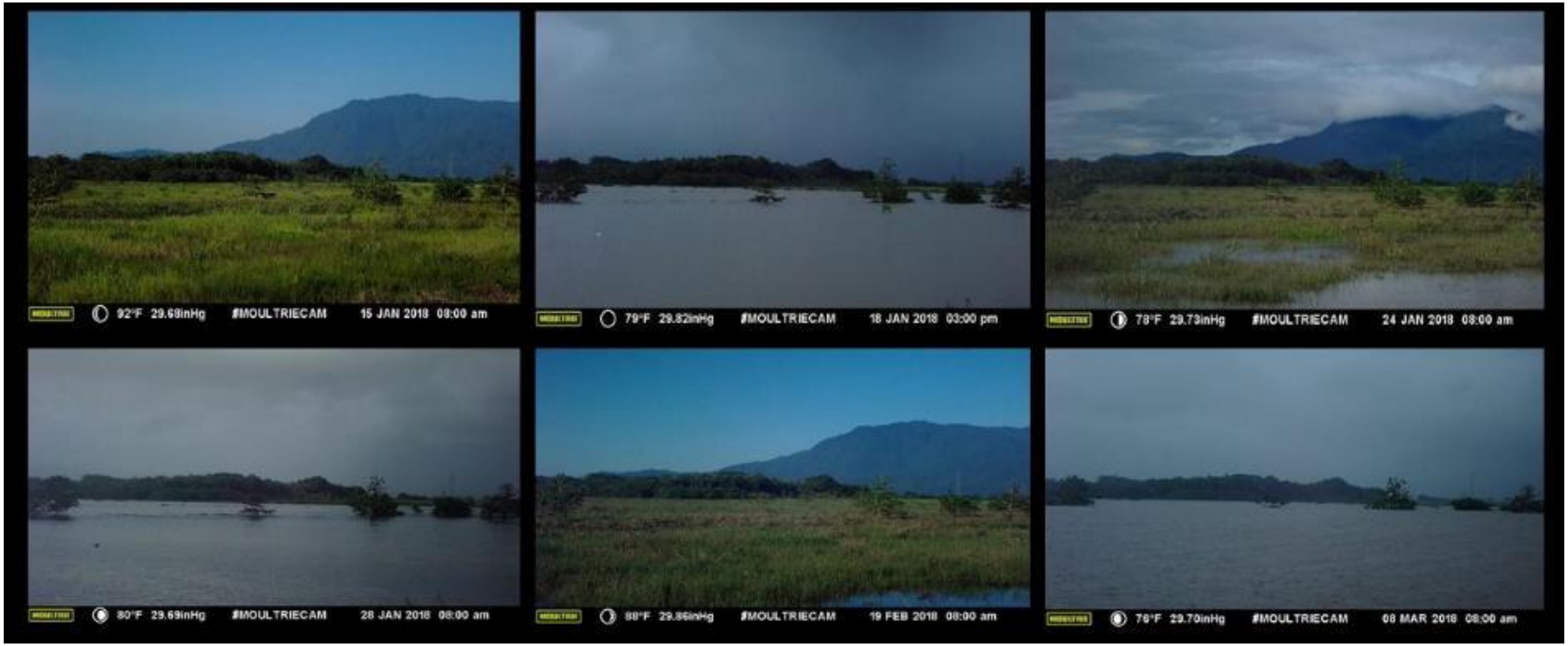

Figure 10: Camera installed near to release gate 12, along the bund wall, facing south. The photos shown here illustrate that the wetland can fill multiple times over a 2mth period

2.2 Wetland denitrification field experiments

Denitrification was measured with the isotope pairing technique (Nielsen 1992, Steingruber et al. 2001), which consists of adding enriched ^{15}N-nitrate to water overlying sediments and estimating ^{15}N-N_2 gas production. The isotope pairing technique is one of the most precise techniques for measuring denitrification and includes coupled sediment nitrification-denitrification and denitrification from the nitrogen in the overlying water (Steingruber et al. 2001), i.e. nitrogen removed from the water column.

On 14 June 2018, we sampled two locations within Babinda wetlands, a dry and a wet portion of the wetland (Figure 11a). At each site, we collected 6 sediment cores of 8 cm length inside Perspex tubes (4.8 cm internal diameter x 0.3m long) capped with a rubber bung (Figure 11b). The sediment cores included small roots and overlying litter. The sediment cores were filled with water collected from the site and left to equilibrate overnight. Note that the sediment cores were 8cm deep, it is therefore possible the denitrification occurs in deep soil horizon, but this was not measured here.

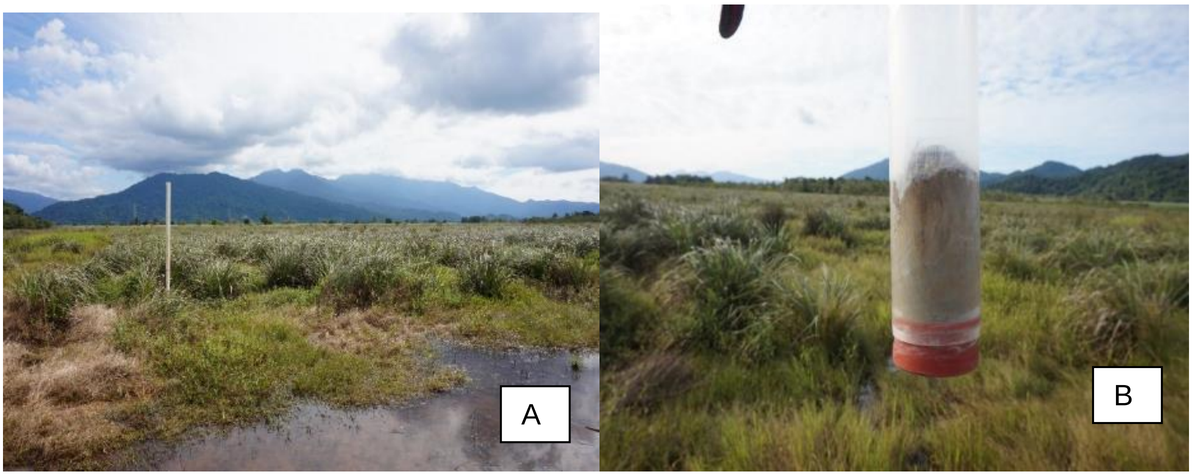

Figure 11: (A) Samples were collected from two sites within Babinda wetlands, a dry, and a wet site; (B) sediment cores were collected from each site and filled with water to run denitrification experiments

In the field we collected surface water samples (~20 cm deep) to measure nutrient concentrations. The samples were collected in triplicate, filtered through a 0.45 µm membrane filter, and stored frozen before being analysed for nutrients within the next week (colourimetric analyses based on APHA/AWWA/WPCF, 2012; Chemistry Centre, Department of Science Information Technology and Innovation, Brisbane, Australia). Detection limits (mg L^{-1}) were: 0.002 for NH_4^+-N, and 0.001 for NO_x^--N and PO_4^--P. Soil samples were also collected at each with a stainless-steel corer of 4 cm diameter. The samples were oven-dried at 60˚C for 48 h and analysed for %N, and organic %C with an elemental analyser (EA-IRMS, Serco System, Griffith University). Bulk density was estimated from the dry weight of the sample and its volume.

The day after collection, the experiments were run at Lucinda, QLD, with similar ambient light and air temperature as in the field. The sediment cores were set in large plastic containers (1030x 510 x 495 mm) filled with water to keep a constant temperature throughout the experiment (ProPlus, YSI meter, OH, USA). During the experiment we ran a blank sample of only distilled water (Figure 12).

At the beginning of the experiment, ^{15}N-NO_3 was added to each core. Cores were capped to minimise headspace. Natural water movement was simulated by a stirrer bar suspended ~ 3 cm above the sediment at each core driven by a rotating magnet that moved at ~60-70 rpm. After approximately 20 min, one core from each batch was sacrificed by adding 1 ml of 50% w/v zinc chloride ($ZnCl_2$) and mixing it throughout the sediment and overlying water to stop bacterial activity. Triplicate 10 mL-water samples from each core were collected with a syringe and placed in a 12.5-ml Exetainer vial (Labco, High Wycombe, UK) which had 250 µL of 50% w/v $ZnCl_2$. For each experiment, we sampled two cores at a time 0, 3, and 5 hours, sacrificing a set of three cores at each time interval (Figure 12).

The headspace gas was analysed by continuous-flow mass spectrometry for $^{28}N_2$, $^{29}N_2$ and $^{30}N_2$-gas (EA-IRMS, Serco System at Griffith University). The detection limit for denitrification rate was 0.01 mg N m^{-2} h^{-1}. Denitrification rates are reported for ambient light conditions in mg N m^{-2} h^{-1} as mean ± standard error.

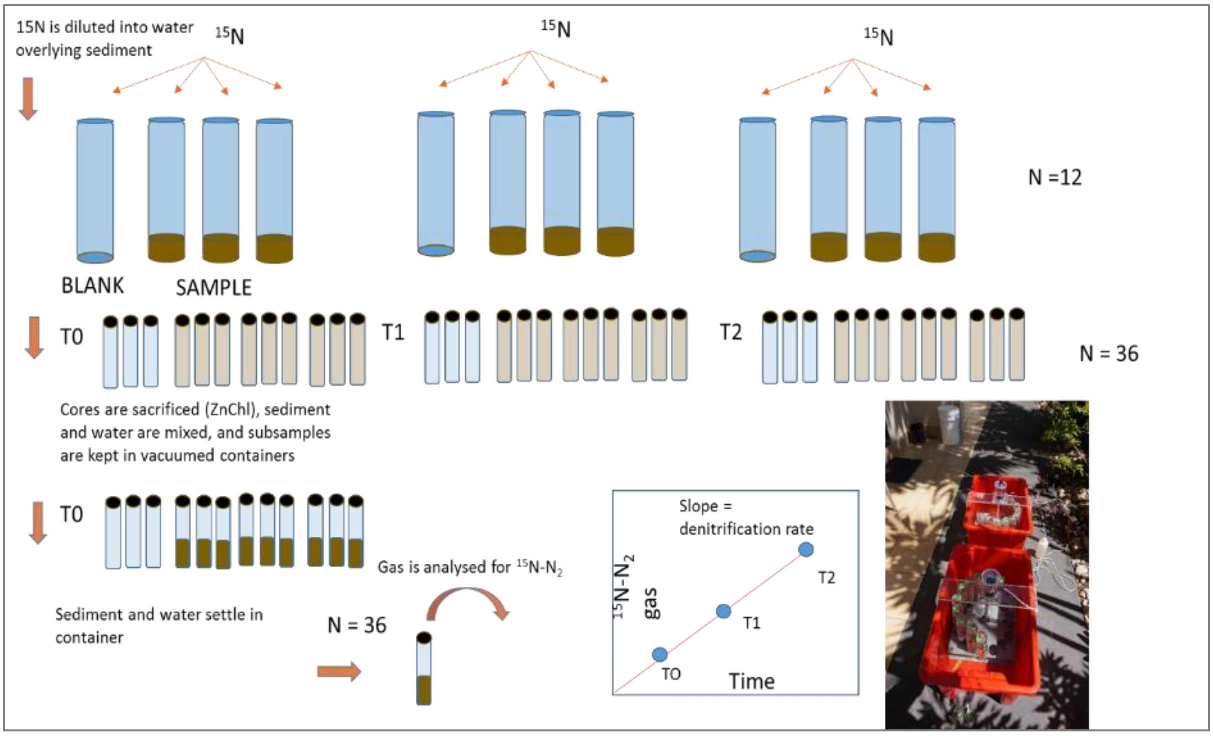

Figure 12: Sampling strategy for denitrification experiment used during this project

The calculations for estimating the different rates of denitrification with this method are shown below:

1. D_{15}: denitrification from labelled $^{15}NO_3$- measured from the production rate of the $^{29}N_2$ and $^{30}N_2$

Equation 2.

$$D_{15} = r_{29} + 2r_{30}$$

Where r_{29} and r_{30} are the production rates of $^{29}N_2$ and $^{30}N_2$, respectively

2. D_{14}: denitrification from unlabelled $^{14}NO_3$

Equation 3.

$$D_{14} = D_{15} \cdot \frac{r_{29}}{2r_{30}}$$

3. D_{tot} = total denitrification or potential denitrification

Equation 4.

$$D_{tot} = D_{15} + D_{14}$$

4. D_w^{tot}: total denitrification of NO_3^- from the water column

Equation 5.

$$D_w^{tot} = \frac{D_{15}}{\varepsilon}$$

where ε is NO_3^-enrichment during incubation as a result of $^{15}NO_3^-$ additions

$$\varepsilon = \frac{[NO_3^-]_a - [NO_3^-]_b}{[NO_3^-]_a}$$

where a and b refer to concentrations after and before $^{15}NO_3^-$ addition. The ε factor during the experiment was 0.79

5. D_w: denitrification from the water column corrected for tracer addition

Equation 6.

$$D_w = D_w^{tot}(1- e)$$

6. D_n: Coupled nitrification-denitrification

Equation 7.

$$D_n = D^{tot} - D_w^{tot}$$

2.3 Wetland water balance

A schematic diagram of the key components of the Babinda wetland water balance is shown in Figure 13.

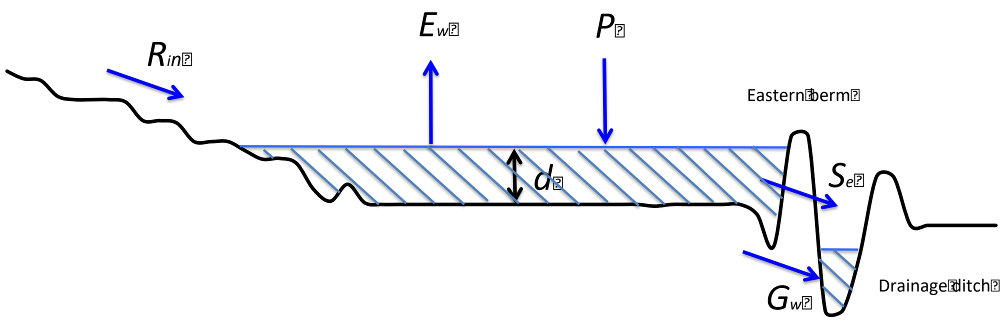

Figure 13: Schematic representation of the Babinda wetland water balance. When the inlet and outlet pipes are closed, water runs in from the higher land to the west and is contained by the constructed eastern berm wall. There is also a deep drainage ditch next to the berm. The symbols are explained in the text.

The change in depth (δd) of the wetland on any day is given by the difference between water entering and leaving it. This can be expressed as:

$$\delta d = (P + Rin) - (DRw + Ew), \qquad \text{(Equation 1)}$$

where P is the rainfall directly entering the wetland, R_{in} is the water which flows into the wetland from its surrounding catchment, and DR_w is the total drainage from the wetland. This has two components, seepage of water through the berm walls, S_e , and deeper groundwater movement under the berm walls, G_w. It is not possible to separate S_e and G_w with the data we have, but we will see later that seepage is likely to be the dominant drainage term (section 3.2.2). E_w is rate of evaporation from the inundated part of the wetland, which is dominated by open water (See Figure 10). Hence we use a method to estimate evaporation from open water (see section 2.3.1 below) and can ignore transpiration. The inlet gate was closed at the start of the study (23 October 2017) and the outlet gates only opened twice when the water level was low (see Figure 13), so they had little/no effect on the wetland water balance in this study.

2.3.1 Evaporation

The rate of evaporation from the wetland (E_w) was estimated using the energy balance model described by McJannet et al. 2008 and Wallace et al. 2015. The main input of energy to the model is solar radiation and the main losses are via heat conduction to the atmosphere and evaporation. The model requires daily weather data, which were obtained for nearby Babinda Post Office from the Scientific Information for Land Owners (SILO) database (http://www.nrw.qld.gov.au/silo/). The SILO database consists of interpolated meteorological variables on a 0.05° (5 km) grid for the whole of Australia (Jeffrey et al. 2001). The SILO variables used in the evaporation model are air temperature, vapour pressure, solar radiation and rainfall and the way these variables are used to calculate daily evaporation are described by McJannet et al. (2008) and Wallace et al. (2015).

2.3.2 Wetland drainage

Drainage (DR_w) from the wetland was calculated from the daily decrease in wetland depth recorded in the rain free periods following rain events. As water also evaporates from the wetland during this time, daily drainage estimates are given by the change in water level (δd) minus the evaporation (E_w) on each day (assuming there is also no run-in R_{in} to the wetland; see equation (1)). Drainage from the wetland was separated into two phases; 1) 'rapid' drainage which occurred when the water level was above the height of the berm wall; and 2) 'slow' drainage when the water level was below the height of the berm wall.

2.3.3 Water run-in from the surrounding catchment

Estimates of the amount of water that flowed into the wetland from its surrounding catchment (R_{in}) during rainfall were made using a simple runoff coefficient model that assumes that R_{in} is a fixed fraction of rainfall, C (e.g. see Pilgrim and Cordery 1993).

$$Rin = C * P \tag{2}$$

As small amounts of rainfall do not generally produce runoff due to losses from interception and depression storage in the land surface (Critchley and Siegert, 1991), only events > 5 mm were used to calculate R_{in}. Above this threshold the value of R_{in} is also affected by the wetness of the surrounding catchment, with less runoff occurring when it is dry. To account for this in a simple way we calculated daily values of the soil moisture deficit (SMD_c) in the surrounding catchment as the difference between rainfall (P) and catchment evaporation (E_c). Unless the catchment is saturated, E_c will be less than the wetland evaporation, E_w, and mainly controlled by the soil moisture deficit and following Shuttleworth (1993) is given by:

$$Ec = Kc\ Ew \tag{3}$$

where K_c is a 'crop coefficient' with a value between 0 and 1 depending on the soil moisture deficit (SMD_c). K_c = 1 when the soil is reasonably wet (SMD_c < 100 mm) after which K_c decreases linearly until it reaches zero at the maximum soil moisture deficit SMD_{max} of 200 mm; typical of a 2 m deep sandy loam soil (Burk and Dalgleish, 2013). Run-in to the wetland is then calculated as:

$$Rin = C * P\ (1 - \frac{SMDc}{SMDmax}) \tag{4}$$

The value of the catchment runoff coefficient, C was then estimated by optimization using the increases in wetland depth observed during large rainfall events.

2.4 Wetland sediment and nitrogen losses

The concept of using constructed wetlands to improve water quality is that water leaves the treatment system with reduced sediment and nutrient loads (Figure 4). Here we estimate this processing capacity of the Babinda wetland by calculating the difference between the amounts of sediment and nutrient entering and leaving the wetland. To do this we use the water balance model described above (section 2.3) in combination estimates of particulate settlement and water and soil denitrification. Each of these is described in more detail below.

2.4.1 Sedimentation and particulate nitrogen

When water enters the wetland the suspended sediment and particulate nitrogen (PN) adhered to it will settle to the bottom of the water column over time. The rate at which this occurs depends on the particle size distribution of the sediment. The larger sand particles will settle first, followed by the silt particles then the smallest fine clay particles.

For the estimation of sediment and PN losses in the Babinda wetland we have used the rate at which total suspended sediment (TSS) declined in the overbank floodwater on the nearby Tully-Murray floodplain (see Appendix A1). This catchment is very similar to the Babinda catchment in that it has high rainfall, steep rainforest covered headwaters and extensive sugar cane agriculture on the floodplain. The equation derived in Appendix 1 (Equation A1) was used to calculate the fraction of the TSS and PN that was lost from the water column over time after the wetland was filled with floodwater. The decline in PN slows over time and it takes 12 days for all the TSS, and hence PN, to settle out of the water column. Ultimately, measurements of TSS and its decline over time in the Babinda wetland should be used to replace the Tully-Murray TSS data.

To calculate the loss of nitrogen due to sedimentation it is also necessary to know the dissolved nitrogen and PN concentrations of the water entering the wetland. Ideally, this should be measured directly in the wetland, however, an initial estimate has been made here using the typical total (dissolved plus PN) nitrogen concentration measured in storm water entering a riverine wetland in the nearby Tully-Murray catchment (McJannet et al. 2012), i.e. ~ 4000 μg L^{-1}. Note that this concentration is higher than the total nitrogen concentration measured several days after the first flush of storm water (i.e. 1000 to 2000 μg L^{-1}), reflecting the loss of nitrogen due to sedimentation and denitrification. This study also found that the nitrogen in wet season storm water was mostly nitrate, with very little (~ 1%) ammonium (NH_4^+) and that the PN concentration was ~ 30% of the total nitrogen. This PN value has been used in our calculations of nitrogen loss due to sedimentation.

2.4.2 Water column denitrification

Another potential pathway for nitrogen loss from a wetland is through gaseous nitrogen (N_2) loss from the water through the process of denitrification (DeVries et al. 2012, Mao et al. 2017). Although many factors can influence the rate of denitrification, the main ones are nitrate supply, readily available carbon and dissolved oxygen (DO) concentrations (Groffman 1994). Other water variables such as temperature and pH appear to only have a secondary influence on denitrification rates.

We have made our estimates of denitrification from the water column using rates estimated here (Section 3.2) along with values that have been reported in three other studies of tropical wetland denitrification in Australia (Table 2). The denitrification measurements made in the Babinda wetland (this study; see section 3.2) for periods of up to 6 hours gave potential water column values (D_w^{tot}) that ranged from 130 to 178 (mean 154) mg of N m^{-2} d^{-1}. A wider range of values was found using the same technique in five tropical wetlands along the north Queensland coast, i.e. 16 to 154 (mean 80) mg of N m^{-2} d^{-1} (Adame et al., 2019a). In a similar study of eight wetlands in the Tully catchment north Queensland, (Adame et al., 2019b) reported an even wider range from 36 to 211 (mean 108) mg of N m^{-2} d^{-1}. Adame et al. (2019c) refer to these generally high rates as denitrification 'hot spots'. In sharp contrast, (McJannet et al. 2012) made estimates of water column denitrification rates in a tropical floodplain lagoon in the Tully catchment north Queensland, finding monthly mean values in a 3 year study between 0 and 3.1 (mean 0.52) mg of N m^{-2} d^{-1} only 0.3 to 0.7 % of the values found above. This type of wetland could be referred to as a denitrification 'cold spot', as the residence time of water in the wetland is generally too short to allow much denitrification to take place.

Many studies report total denitrification, the sum of nitrogen losses in the water and from within the soil (see Table 2). However, the one by Brezonik and Lee (1968) reported rates for the water column (over a 38 day period) ranging from 8 to 26 (mean 17) mg N m^{-2} d^{-1} from the bottom of the nutrient rich Lake Mendota in the USA. A similar range (and mean) of water denitrification rates was reported by Seitzinger (1988) for a number of lakes from around the world. More recently, Pina-Ochoa and Alvarez-Cobelas (2006) made a comprehensive review of denitrification rates worldwide and found the widest range from 5.8 to 112 (mean 76) mg N m^{-2} d^{-1}.

Some of the differences in the above studies can be ascribed to differences in their duration, methodology and climate, but it is clear that denitrification rates in water can be highly variable even amongst nearby tropical wetlands. To make our model estimates of denitrification in the Babinda wetland we have therefore made simulations using the range of mean values recorded in the four tropical wetland studies in north Queensland (i.e. studies 1 to 4 in Table 2). To allow for water column nitrogen losses to vary with wetland depth, we converted the denitrification rates to a volumetric basis, using a sample depth of 0.2 m for studies 1, 2 and 3 and an average lagoon depth of 1.5 m for study 4. This gave values ranging from 0.34 (McJannet et al., 2012) to 768 µg N L^{-1} d^{-1} (this study), with an average value of 427 µg N L^{-1} d^{-1}. These daily rates were converted to the total denitrification loss from the entire wetland using the wetland volume on each day.

Table 2: Literature values of denitrification within the soil and water found in tropical wetlands in Australia (studies 1 to 3). For comparison, a selection of values reported in various natural and created wetland studies from around the world are also shown (studies 4 to 11).

	Wetland type / Location	Climate	Water & Soil denitrification Range from	to	Mean	Water column denitrification Range from	to	Mean	Soil denitrification Range from	to	Mean	Source
			mg N m^{-2} day^{-1}			mg N m^{-2} day^{-1}			mg N m^{-2} day^{-1}			
1	Babinda wetland, North QLD, Australia	Tropical	163	226	**194**	130	178	**154**	34	48	**41**	This study
3	Tropical wetlands, North QLD, Australia	Tropical	40	410	**181**	36	211	**108**	4.3	199	**73**	Adame et al (2019a)
2	Forested wetlands, North QLD, Australia	Tropical	26	233	**125**	16	154	**80**	0.0	154	**45**	Adame et al (2019b)
4	Riverine wetland, North QLD, Australia	Tropical	11	166	**33**	0.0	3.1	**0.52**	10.9	163	**33**	McJannet et al (2012)
5	Mangroves, Malaysia	Tropical	11	308	**160**							Alongi et al., (2004)
6	Tropical streams, Puerto Rico	Tropical	0.0	192	**22**							Potter et al., (2010)
7	Mangroves, Mexico	Tropical							9.7	74	**42**	Rivera-Monroy and Twilley, (1996)
8	Sub-tropical lakes, China	Sub Tropical							0.8	40	**5.8**	Yao et al. (2016)
9	Lake Mendota, USA (summer average)	Temperate						**19**				Brezonik and Lee (1968)
10	Created wetlands (2 x 1 ha), Ohio, USA	Temperate	2.4	60	**31**							Bateson et al., (2012)
11	Natural and created wetlands, northern hemisphere	Various	1	408	**91**							cited in Bateson et al., (2012)
12	Freshwater wetlands, worldwide	Various							44	367	**164**	Johnson (1991)
13	Lakes and rivers, wordlwide	Various							0.7 0.0	57 116	**29** **58**	Lakes; Seitzinger (1988) Rivers

A critical parameter in estimating denitrification in water is its dissolved oxygen (DO) concentration, as it has been reported that denitrification can only occur when DO is low (e.g see Seitzinger 1988). More recently, Rivett et al. (2008) summarised a number of studies showing that the maximum DO level at which denitrification occurred in groundwater varied between 0.2 and 4 mg L^{-1}, with typical values around 2 mg L^{-1} (equivalent to ~ 25% saturation). Ideally we would require measurements of DO in the Babinda wetland at both the top and bottom of the water column. In the absence of these data our initial estimate of water column denitrification has been made assuming the average amount of time DO was below 2 mg L^{-1} in the nearby Tully wetland lagoon, i.e. ~ 30% (McJannet et al. 2012).

The absolute amounts of nitrogen lost from the wetland (by both denitrification and sedimentation) are dependent on the volume of water in the wetland. This was estimated from the topographic survey made by the Australian Wetlands Consulting prior to the wetland installation. When the wetland was full to the top of the berm wall, the maximum depth in the wetland was 0.85 m and the volume was 63241 m^3. This volume has been kept constant in the first denitrification calculation. In practice, in the absence of any rainfall, this volume reduces over time due to losses associated with evaporation and drainage. These have been calculated using a water balance model of the Babinda wetland and are used in a second version of the denitrification model, which also calculates the decrease in wetland volume (and area) with time after it was filled. Further details of the wetland water balance model are given in a Section 2.3 of this report.

2.4.3 Denitrification from within the soil

Denitrification can also occur in the sediment and organic matter in the soil at the bottom of the wetland if the conditions are suitable (i.e. anaerobic). The key factor is that dissolved oxygen levels must be low and these are most likely to coincide with times when there is standing water in the wetland. Wetland soil and/or sediment denitrification rates have been reported for a number of wetland types (see Table 2). The Seitzinger (1988) review reports soil denitrification rates in lakes under low oxygen and anoxic conditions of between 0.7 and 57 (mean 29) mg N m^{-2} d^{-1}. In rivers the equivalent figures are 0 to 116 (mean 58) mg N m^{-2} d^{-1}.

In tropical wetlands in north Queensland we have calculated soil denitrification rates as the difference between total denitrification rates (D_{tot}) and denitrification rates from the water column (D_w^{tot}). Using values of D_{tot} and D_w^{tot} reported by Adame et al., (2019a) the soil denitrification rates ranged from 0 to 154 (mean 45) mg N m^{-2} d^{-1}, similar to the soil rates used by McJannet et al., (2012) and those found in the current study in Babinda (Table 2). In their most recent tropical wetland study Adame et al., (2019b) also found values of D_{tot} and D_w^{tot} that give a similar range of soil denitrification rates, 4 to 199 (mean 73) mg N m^{-2} d^{-1}.

For comparison, studies of sediment cores from lakes in China found very low rates 0.8 to 40 (mean 5.8) mg N m^{-2} d^{-1}. The widest range of soil denitrification rates was reported in the extensive review by Johnston (1991), at 44 to 367 (mean 164) mg N m^{-2} d^{-1}. Again, soil denitrification rates are very variable, so in our estimates for Babinda wetland we have used the range found in the tropical wetland studies in north Queensland (studies 1 to 4 in Table 2) of 33 to 74, with an average value of 48 mg N m^{-2} d^{-1}. These daily rates were converted to the total nitrogen lost from the entire wetland using the wetland area on each day.

3.0 RESULTS AND DISCUSSION

3.1 Wetland denitrification field data

Nutrient concentrations in the standing water of Babinda at the time of sampling (14 June 2018) were 0.022 ± 0.001 mg L^{-1} of N-NOx-, 0.022 ± 0.001 mg L^{-1} of NH4+ and < 0.001 mg L^{-1} of P-PO4-. Soil %C and %N were similar in the wet compared to the dry site with values of 4.0 ± 0.73 % and 3.2 ± 1.2 %C for dry and wet sites, respectively, and 0.29 ± 0.046 % and 0.24 ± 0.091 % N (Table 3). Soil bulk density was 0.66 ± 0.069 and 0.64 ± 0.073 g cm^{-3} for the dry and wet site, respectively.

Mean total potential denitrification rates (D_{tot}) were 8.1 ± 2.8 mg m^{-2} h^{-1} and were similar between the wet and the dry site (Table 3), which means that once the wetland is flooded, the sediment starts denitrifying even if it has been dry for long periods of time. The mean total denitrification rate of the water with tracer addition (D_w^{tot} = 6.4 mg m^{-2} h^{-1}) was over four times that from within the soil (D_{tot} - D_w^{tot} = 1.5 mg m^{-2} h^{-1}). These values were multiplied by 24 to give the daily rate values given in Table 2.

Table 3: Denitrification rates (mg m^{-2} h^{-1}) of soil and water in the Babinda wetlands collected from a site that was dry and another one that was wet at the time of sampling. D_{tot} = total or potential denitrification during experiment; D_w^{tot} =total denitrification of labelled plus unlabelled NO3- from water column; D_w = natural denitrification rate of the water column without tracer addition; coupled nitrification-denitrification (Nielsen 1992)

	D_{tot}	D_w^{tot}	D_w	D_n
DRY site	9.4 ± 7.2	7.4 ± 5.6	2.0 ± 1.5	1.6 ± 1.2
WET site	6.8 ± 3.6	5.4 ± 2.8	1.5 ± 0.76	1.1 ± 0.60

3.2 Water balance

3.2.1. Wetland water depth

Water first entered the Babinda constructed wetland on 19 October 2017, following heavy rainfall (300 mm) on this and the previous day (Figure 14). The daily average water depth at site 2 rose rapidly, reaching a maximum of 140 cm (note that the daily average depths shown here may be lower than instantaneous depths reached during the day). When the depth at site 2 was ~ 60 cm, the water level was at the top of the berm wall; note that the water depth in the deepest part of the wetland was 85 cm at this time.

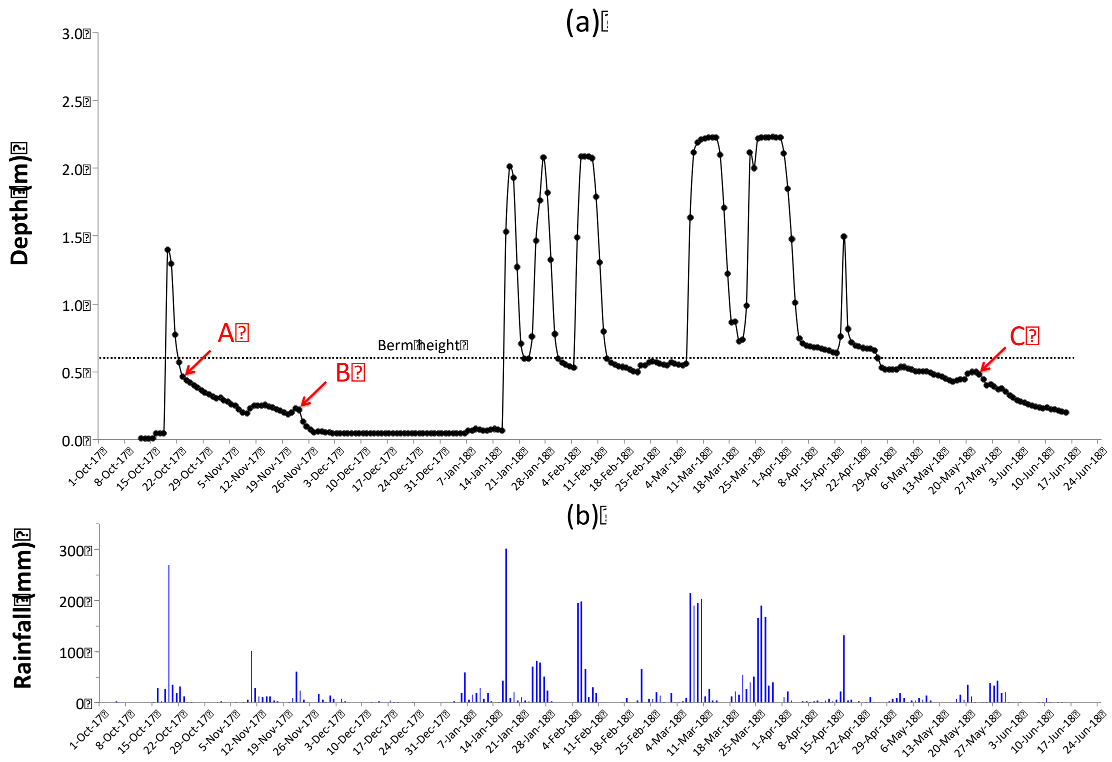

Figure 14: (a) Changes in daily average depth at wetland site 2 between October 2017 and June 2018 and (b) Daily rainfall during the same period. The height of the berm wall spillway is shown by the dotted line. Sluice gates were operated on three occasions; (A) 23 October 2017 – inlet gate closed, (B) 23 November 2017 – outlet gates opened, (C) 23 May 2018 – outlet gates opened.

Once the heavy rainfall on 19 October 2017 had stopped, the water level in the wetland dropped very rapidly, reaching 46 cm four days later (on 23 October 2017). After this, the water level dropped more slowly (coinciding with the closure of the main inlet gate), at a fairly constant rate of ~1.7 cm per day for the next 15 days, at which time the wetland depth was ~ 20 cm. Rain on 9 and 10 November 2017 (80 mm) caused a small rise in water level (~ 5 cm), after which the water level stayed around 25 cm due to a further series of small rainfall inputs in November 2017 (Figure 13b).

After draining the wetland on 23 November 2017 (point B Figure 12; see also *Wetland drainage* section below), water depth was below 5 cm (the lowest level the sensors can record) throughout December and early January 2018. The water level rose rapidly after 343 mm of rain on 16-17 January 2018 (Figure 13b), reaching just over 2 m before falling rapidly again. There were a further 4 occasions after this when heavy rainfall caused the water depth to rise to over 2 m. Interestingly, during the last three events water depth stayed fairly constant at ~2.2 m for up to 7 days. These events were characterized by prolonged very heavy rainfall, i.e. 4 – 9 February 2018, 517 mm; 6 – 13 March 2018, 847 mm and 20 – 28 March 2018, 768 mm. A possible explanation of the steadiness of the high water level during these events is discussed in the *Preliminary interpretations* section below.

One consequence of the high rainfall from mid-January to the end of May 2018 (3615 mm) was that the water level in the wetland never fell below 30 cm in this period (136 days). It was only 7 days after the outlet gates were opened on 23 May 2018 (point C in Figure 13) that the wetland depth decreased below this level.

3.2.2. Wetland drainage

When rainfall stops and the water depth (at site 2) is above the surrounding berm wall spillway (~ 60 cm), water drains very rapidly from the wetland. Estimates of this 'rapid' drainage were made from the rate at which the water level in the wetland dropped after the five major rainfall events in 2018. The drop in water level on each day after rainfall stopped was adjusted for losses due to evaporation (calculated from weather data – see section 2.3.1) and any relatively small amounts of rain (< 12.4 mm). Figure 14 shows that this 'rapid' drainage estimate increases with water depth and can reach over 60 cm day^{-1}. However, there is a fair amount of scatter in the data (r^2 = 0.42), which may be associated with processes determining run-in and water depth outside the wetland, both of which are controlled off site and may vary between different flooding events. For this reason, the linear regression in Figure 15 should only be use to give a first order estimate wetland drainage when depths are above the height of the berm wall.

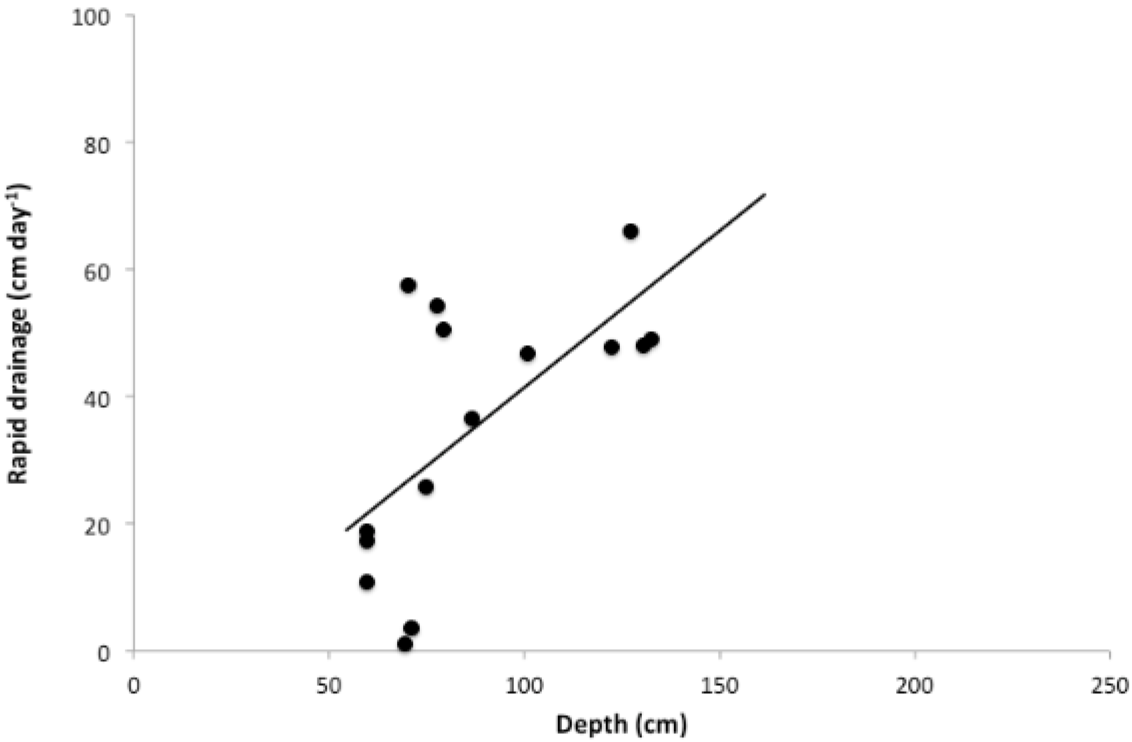

Figure 15: The relationship between 'rapid' drainage from the wetland and depth when the water is above the berm wall spillway. The linear regression has the form, DR_w = 0.49 d – 7.8 (r^2 = 0.42)

Once the water level in the wetland fell below the height of the berm wall spillway, the rate at which the level dropped slowed markedly, as it was controlled by losses due to internal wetland drainage and evaporation. Calculations of this 'slow' drainage (DR_w) on two relatively rain free periods in 2017 (16 days) and 3 similar occasions in 2018 (14 days) are summarised in Figure 16. On each day evaporation was subtracted from the drop in water level and any small amounts of rain (< 2.1 mm) added to give the net drainage on that day.

The black dots are from October and November 2017 and show the expected linear relationship between drainage and depth, with the line going through ~ zero, i.e. no drainage when there is no water. The linear regression through the origin (black line) has a reasonably high correlation coefficient (i.e. r^2 = 0.63). These data are for before the gates were opened in 2017 (24 November 2017), so represent an estimate of drainage from the wetland when all the outlet gates are shut. The blue dots are for February, May 2018 (before the gates were opened on 23 May 2018) and 7 – 15 June 2018 after the water level had dropped below the bottom of the outlet pipes. The 2018 drainage rates are substantially lower than those in 2017 (by a factor of ~ 2). The lower drainage rates in 2018 may have been due to higher water levels surrounding the wetland in this year, which also affected the rate of water loss through the outlet gates (see Section 3.3.3 below). The linear regressions in Figure 15 can be used to estimate the wetland drainage rate for any depth below the height of the berm wall.

As explained in the methods (Section 2.3), it is not possible to separate total drainage (DR_w) into its seepage (S_e) and groundwater (G_w) components. However, given the relatively high values of DR_w obtained, i.e. 5 to 20 mm day^{-1}, it is likely that drainage from this wetland is dominated by seepage, as groundwater recharge is normally much lower than these values. For example, Narayan, Scheelberger and Bristow (2007) reported values of the recharge rates from constructed recharge pits on the Burdekin delta of 1.4 to 3.4 mm day^{-1}.

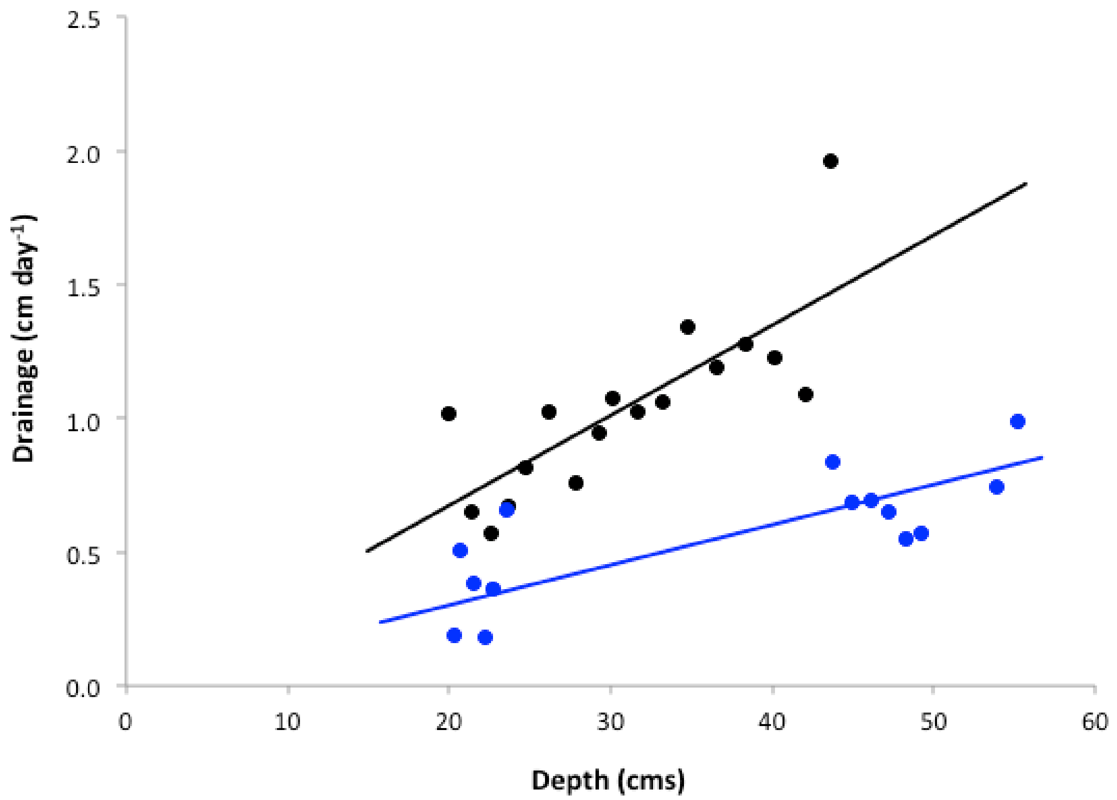

Figure 16: A plot of drainage rate versus wetland depth (< 60 cm) when no water flowed through the inlet and outlet gates. The black dots (and linear regression line) are for October/November 2017 and the blue dots are for February/May/June 2018. The linear regressions through the data in each year have the form; 2017 DR_w = 0.034 d (r^2 = 0.63) and 2018, DR_w = 0.015 d (r^2 = 0.58).

3.2.3 Water loss through the outlet gates

Around midday on 23 November 2017 the three sluice gates nearest site 2 were opened to help drain the wetland for repairs to leaks at these sluices. This caused an increase in the rate at which the water dropped over the next 4 days (see point B in Figure 14). This was used to give an estimate of the rate at which water is lost via the open sluice gates. On the four days following the opening of the gates there was little/no rain and the recorded level drop in sequential 12 hr intervals was adjusted to allow for losses due to evaporation and drainage and for any very small additions due to rainfall (Table 4). The resultant 'pipe flow' was greatest on the first day after opening and this decreased rapidly as the depth in the wetland fell.

Table 4: Estimates of flow through the three sluice gates 1, 2 and 3 made in November 2017 using the recorded level drop and rainfall along with estimates of evaporation and drainage (as in 2017 when DR_w = 0.034 *d*; see Figure 17)

12 hr period	Level drop (cm day^{-1})	Evaporation (cm)	Rain (cm)	Drainage (cm)	Outlet 1, 2 & 3 flow (cm day^{-1})	Depth (cm)
12:40 to 00:00 23/11/2017	-16.8	0.25	0.0	0.48	16.1	29.9
00:00 to 12:00 24/11/2017	-6.0	0.25	0.1	0.41	5.4	24.4
12:00 to 00:00 24/11/2017	-4.4	0.25	0.0	0.37	3.8	21.8
00:00 to 12:00 25/11/2017	-2.2	0.27	0.0	0.34	1.6	20.2
12:00 to 00:00 25/11/2017	-2.8	0.27	0.0	0.32	2.2	18.9
00:00 to 12:00 26/11/2017	-0.6	0.30	0.0	0.30	0.0	17.9
12:00 to 00:00 26/11/2017	-3.3	0.30	0.0	0.29	2.7	17.2
00:00 to 12:00 27/11/2017	-1.3	0.31	0.0	0.27	0.7	15.9
12:00 to 00:00 27/11/2017	-0.9	0.31	0.0	0.26	0.4	15.5

The rate at which water flows through a pipe is proportional to the depth of water above the pipe. This relationship is shown in Figure 17, which shows a plot of flow rate through the three pipes (F_3) versus depth (d) using the data in Table 4. The linear regression through the points has the form:

$$F_3 = 0.99\,d - 16.4$$

where F_3 is the flow loss through all 3 gates (cm day^{-1}) and d is the average depth of water (cm) over each 12 hr calculation period. Equation (Islam and Tanaka) can be used to make an estimate of the flow loss for any number of gates assuming they all lose water at the same rate for any given depth.

It should be noted that estimates of water loss through the outlet gates 1, 2 and 3 were also made when the outlet gates were opened on 23 May 2018 (point C in Figure 14). Calculations at this time were made more difficult as there was significant rainfall from 25 May 2018 onwards, which may have generated unknown amounts of run-in to the wetland. It was therefore only possible to make pipe flow estimates for the 23 and 24 May and these were much lower (~ 25%) than those estimated above in 2017. It is not clear why the wetland drained more slowly at this time, but it could have been due to higher water levels on the land and in the ditches surrounding the wetland. Alternatively, there may have been some blockage of the outlet pipes that impeded flow through them.

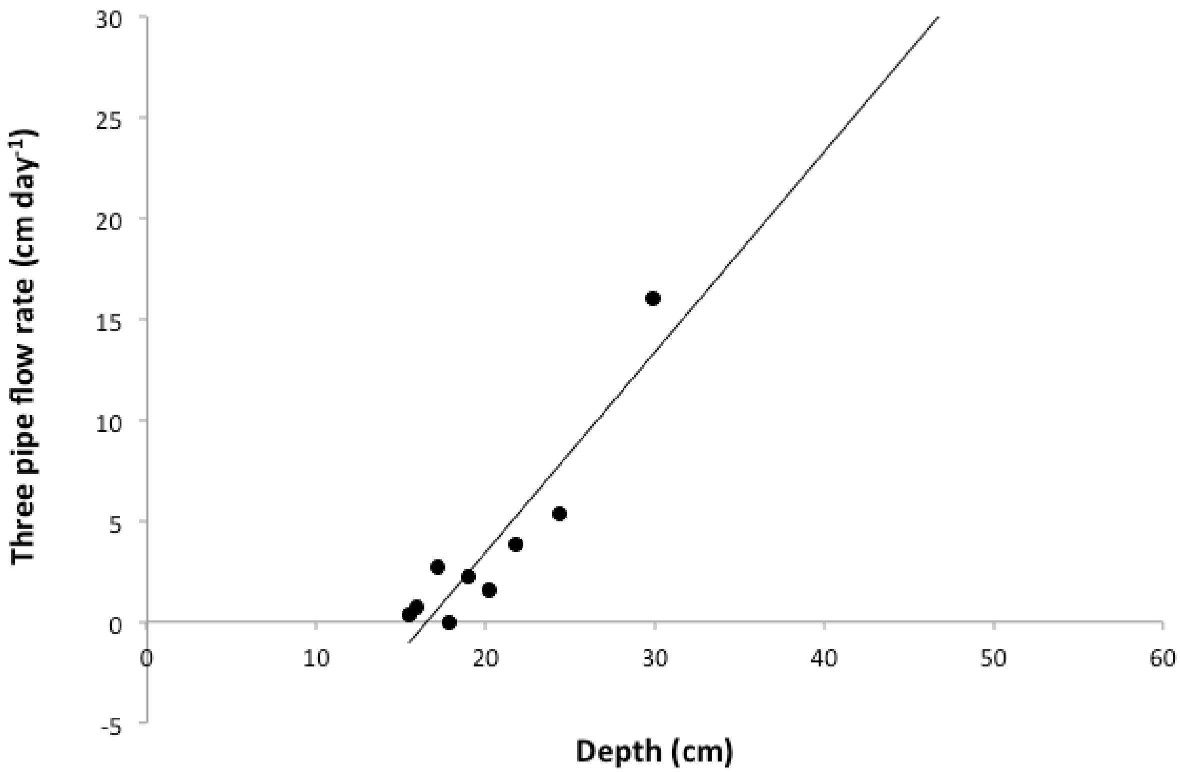

Figure 17: The relationship between flow loss through sluice gates 1, 2 and 3 and the average depth of water over sequential 12 hr periods between 23 and 27 November 2017. The linear regression has the form F_3 = 0.99 *d* – 16.4 (r^2 = 0.85)

Figure 17 shows the relationship between wetland depth and time (days) after the opening of either 1 or 3 sluice gates. In this simulation, the initial depth of the wetland is taken as 60 cm, i.e. the water level is to the top of the berm wall spillway. The pipe flow estimates were made using the linear regression shown in Figure 17 (i.e., 2017 data). The dashed lines in Figure 18 show the effect of losses via the sluice gates only, where the minimum depth that would be reached is 16.4 cm; effectively the height of the bottom of the sluice pipe above the soil surface of the wetland (at logging site 2). The figure shows that water would drain to this level after ~15 days, with one gate open and after ~5 days with 3 gates open. By extrapolation, it would take just over 1 day to drain the wetland from full to the bottom of the outlet pipes if all 12 sluice gates were opened.

In practice water would also be lost from the wetland via evaporation and drainage. In Figure 18, evaporation was taken as the average daily value (0.46 cm) calculated between 12 October 2017 and 15 June 2018. Drainage was calculated from wetland depth using the linear regression derived from 2017 data; i.e. the black line in Figure 16. These simulations are shown as the bold lines in Figure 18. For example, with 3 sluice gates open it would take ~ 32 days to completely empty the wetland, with flow through the pipes stopping after ~ 5 days. If all 12 sluice gates were opened when the wetland was full, the depth would drop to the bottom of the pipes (16.4 cm) in just over 1 day, but it would still take a further ~ 27 days to completely empty the wetland, since the loss of water once the depth is below the bottom of the outlet pipes is controlled by evaporation and drainage.

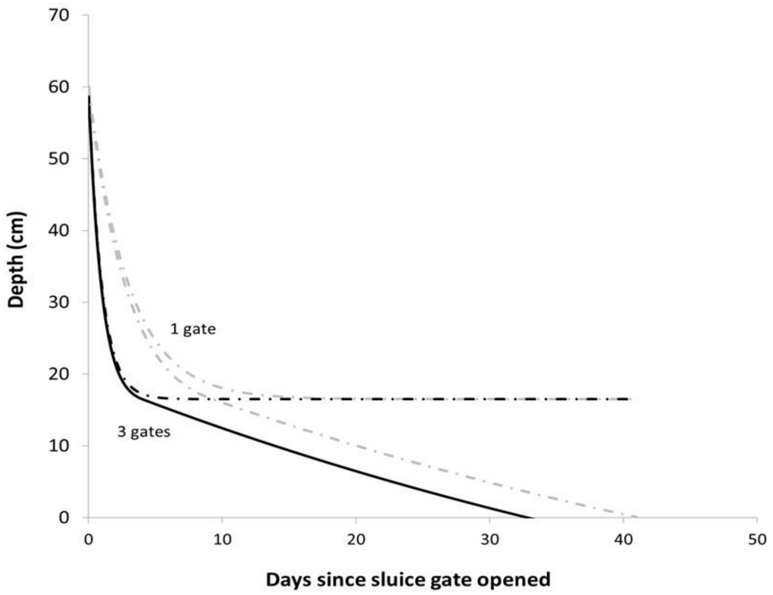

Figure 18: The fall in wetland depth with time after 1 (grey) or 3 (black) sluice gates are opened. Dashed lines show the effect of losses via the sluice gates only. Solid lines show the additional effect of evaporation and drainage.

3.2.4 Water balance interpretations

The data recorded so far show that large rainfall events are sufficient to raise the water depth in the constructed wetland to ~ 2.2 m. This is well above the level of the berm access roads (~ 60 cm) and so water initially drains very rapidly (~ 10 - 70 cm day^{-1}) from the wetland. The rapid drainage lasts for several days (~ 4 to 6) until the water level drops below the height of the berm spillway. Once the water level is below the berm wall height 'slow' drainage continues between 0 (when d = 0) and 2 cm day^{-1} (when d = 60 cm) and there are additional losses of 0.2 – 0.7 cm day^{-1} via evaporation.

High rainfall during November to March (3789 mm) was 42% above the long-term average for this period (2674 mm) and this kept the water level in the wetland above 50 cm between January and the end of May 2018. This period of 113 days of continuous inundation should have a marked effect on the ecology of the wetland and its potential for denitrification of the water within the wetland (see Section 3.4).

Very heavy and prolonged rainfall can raise the wetland water level to 2.2 m for several days, but the level does not appear to go above this. This is characteristic of the behaviour of a weir, so it may be that the earth bund next to the main parallel drain is acting as such under very heavy rainfall. If so, deep water in the wetland may be flowing over this bund and into the main drain. This continues for as long as the surrounding catchment runoff generated by the heavy rain is greater or equal to the loss of water over the bund – hence the sustained depth of 2.2 m for several days. This would imply that most of the water entering the wetland at this time comes in as runoff from the catchment area to the west of the wetland.

Drainage from the wetland after the inlet gate was closed in October 2017 varied between 2 and 1 cm day^{-1} as the depth dropped from 44 to 20 cm. This is several times greater than the evaporation loss at this time, ~ 0.53 cm day^{-1}. Drainage rates from the wetland in 2018 were around half of those recorded in 2017, possibly because of the sustained higher water levels surrounding the wetland during the very high rainfall in 2018. It is also possible that some of the higher drainage in 2017 may have been due to leakage from the base of sluice gates 1, 2 and 3, which was visually observed at this time.

The wetland can be emptied very rapidly by opening the outlet sluice gates; in a little as ~ 1 day with all 12 gates open. However, this only drops the water level to the bottom of the outlet pipes with ~ 16.4 cm of water remaining in the wetland. At this point the wetland inundated area is 29% of that when the wetland is full to the height of the berm wall. In the absence of rain, the wetland would take a further 27 days to become completely empty, as the losses by evaporation and drainage are much slower than via the outlet pipes.

3.3 Wetland denitrification

Preliminary estimates of nitrogen loss from the Babinda wetland are given below. The first estimate is for a bank full wetland where the depth stays constant with time. A second, more realistic, estimate of nitrogen losses are given for conditions in which the wetland depth decreases in time by allowing for losses due to evaporation and drainage. We have also made an estimate of denitrification from within the soil beneath the water in the wetland. Finally, the denitrification of the wetland over the 12 month period from October 2017 to September 2018 was estimated using the depth changes recorded for most of this period.

3.3.1 Denitrification from the water column

The first estimate of nitrogen loss from the Babinda wetland is shown in Figure 19. In this simulation the depth of the wetland is held constant. This demonstrates that the total denitrification over the 60 days of the simulation is very dependent on the water column denitrification rate. When the low rate is used (0.34 μg N L^{-1} day^{-1}), the maximum nitrogen loss is 27% (68 kg) of the initial nitrogen (253 kg) in the wetland water. Virtually all of this loss (98%) is due to sedimentation of particulate nitrogen, which occurs during the first 12 days of the simulation. All of the total nitrogen is lost (253 kg) when the average denitrification rate is used (427 μg N L^{-1} day^{-1}), mostly due to gaseous denitrification from the water column (81%). If the denitrification rate were as high as 768 μg N L^{-1} day^{-1}, then all of the nitrogen would be lost after 14 days. The denitrification rate is therefore very important in determining the true loss of nitrogen as nitrogen gas (released to the atmosphere), as the PN fraction falls onto the surface of the soil beneath the water column. The fate of the PN is considered below.

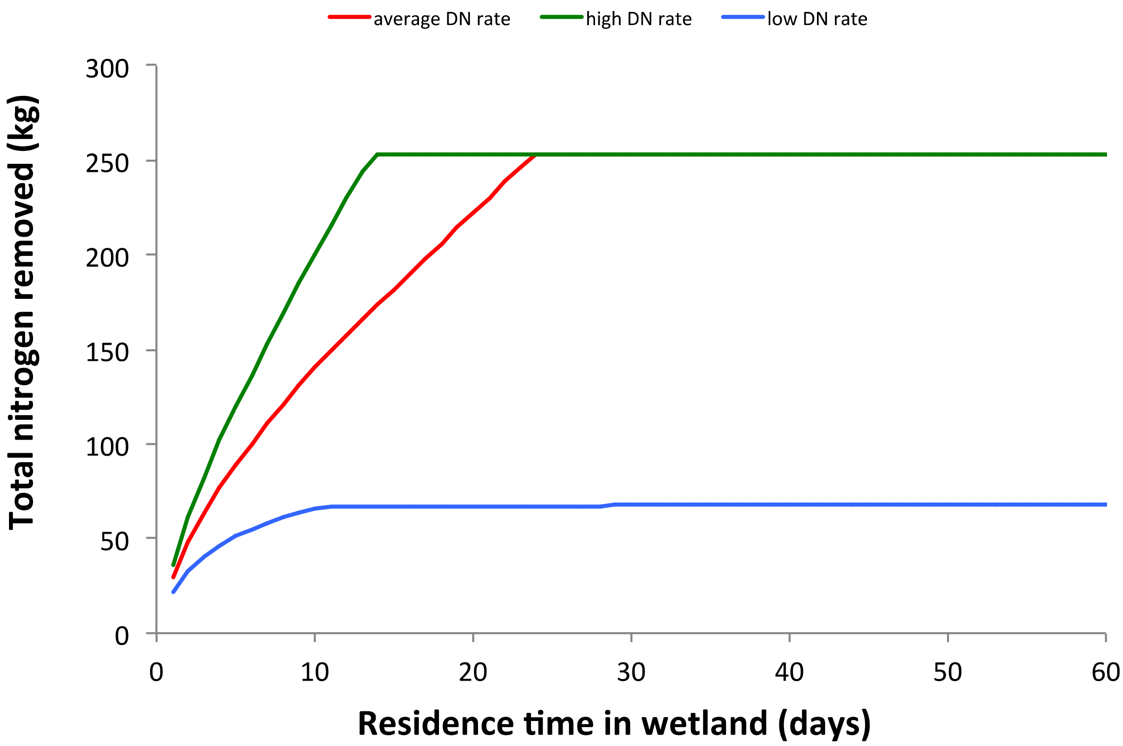

Figure 19: The amount of nitrogen removed from the water column (kg) as a function of the length of time the water is in the wetland (residence time). Three water column denitrification rates are shown (i) low (blue line) – 0.34 µg N L^{-1} day^{-1}, (ii) average (red line) - 427 µg N L^{-1} day^{-1} and high (green line) - 768 µg N L^{-1} day^{-1}

3.3.2 Water denitrification with evaporation and drainage

The second denitrification simulation is more realistic as it allows for depth to decrease due to losses from evaporation and drainage. In this scenario we assume that storm water has entered and filled the wetland and there is no further rainfall or run-in. Subsequent depth decreases are calculated using a simple water balance model that assumes a constant rate of evaporation (4.6 mm day^{-1}; the average for the period October 2017 to June 2018) and drainage is calculated from depth as $DR_w = 0.034\ d$ (see Figure 15). With these evaporation and drainage rates, it takes ~ 60 days for the wetland to empty. Over this period the total amount of nitrogen lost from the wetland is shown in Figure 20.

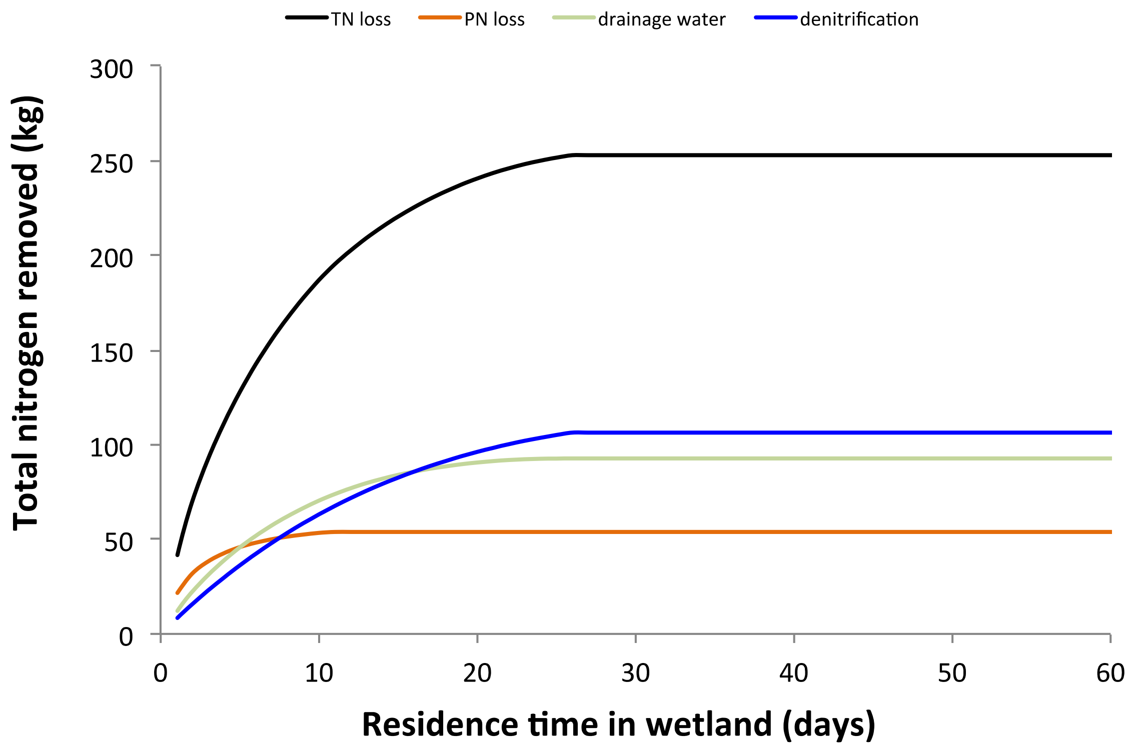

Figure 20: The total amount of nitrogen (kg) removed from the water column (TN -black) as a function of the length of time the water is in the wetland (residence time). Losses due to PN sedimentation (orange), water column denitrification (blue) and loss of drainage water (green) are also shown

In this simulation the average rate of denitrification was used (427 μg N L^{-1} d^{-1}) and this led to all of the water column nitrogen (253 kg) being removed after 26 days. Gaseous denitrification constitutes the largest loss of nitrogen, at 42% (106 kg) of the total N loss after 26 days. Nitrogen losses due to sedimentation are lower at 21% (54 kg) and all of this occurs in the first 12 days, as it is assumed that all of the PN settles to the bottom of the water column by this time (see Figure A2). In this simulation there is also a major loss of nitrogen from the water column due to drainage, which continues until the concentration of nitrogen in the wetland is zero. This mechanism makes up 37% (93 kg) of the total nitrogen loss from the wetland, nearly as much as is lost by denitrification. Whether this is a true loss of nitrogen (as N_2 gas to the atmosphere) from the wider area outside the wetland depends on where the drainage water goes; to surrounding ditches, streams and/or groundwater, and the denitrification conditions within those locations.

The time courses of the percentage of nitrogen lost due to sedimentation, denitrification and drainage are shown in Figure 21. This confirms the initial dominant influence of the loss of PN, which is overtaken by drainage losses after ~ day 5. Drainage losses then dominate up to day 15, after which denitrification losses account for most nitrogen loss.

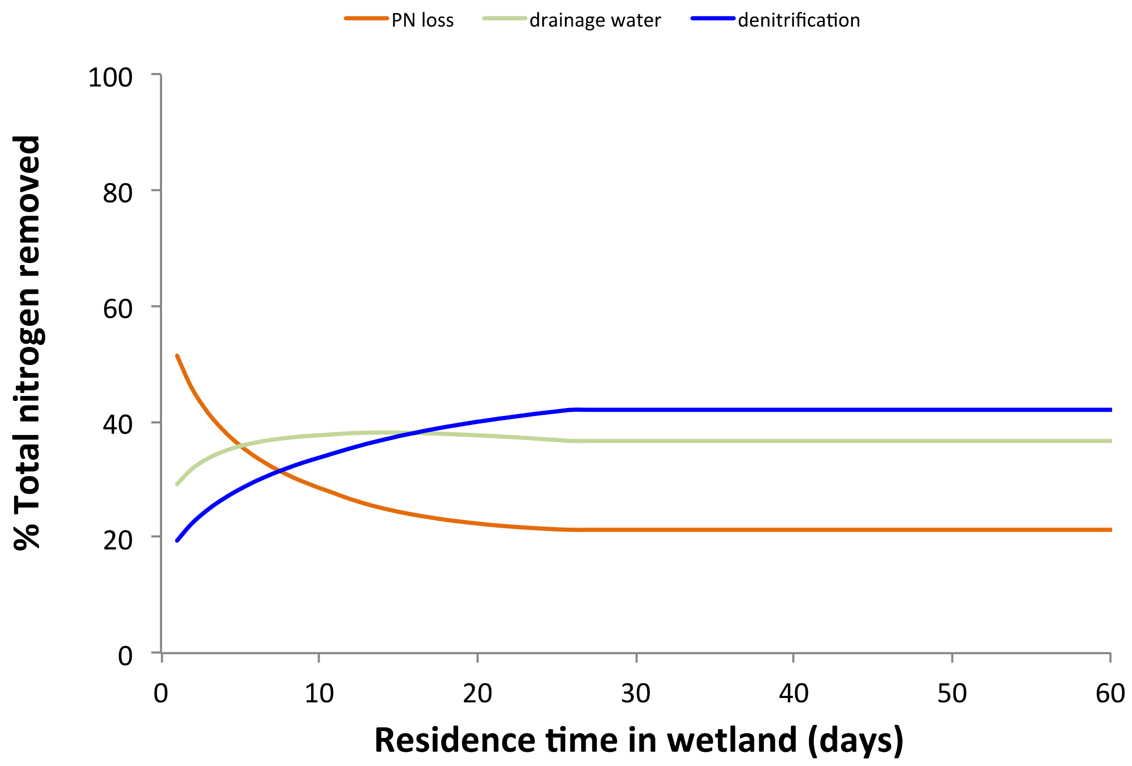

Figure 21: The fraction (%) of nitrogen removed from the water column as a function of the length of time the water is in the wetland (residence time). Percentage losses due to PN sedimentation (orange), water column denitrification (blue) and loss of drainage water (green) are shown.

3.3.3 Water nitrogen concentration

The way in which the concentration of nitrogen in the wetland water column changes with time is shown in Figure 22. Again, in this simulation the average rate of denitrification was used (427 µg N L^{-1} day^{-1}). The initial concentration of dissolved and particulate nitrogen is assumed to be 4000 µg L^{-1} and this decreases quite rapidly initially as the PN fraction precipitates from the water column in the first 12 days. After this, the nitrogen concentration falls more slowly and at a fairly constant rate (due solely to denitrification) and reaches zero on day 26.

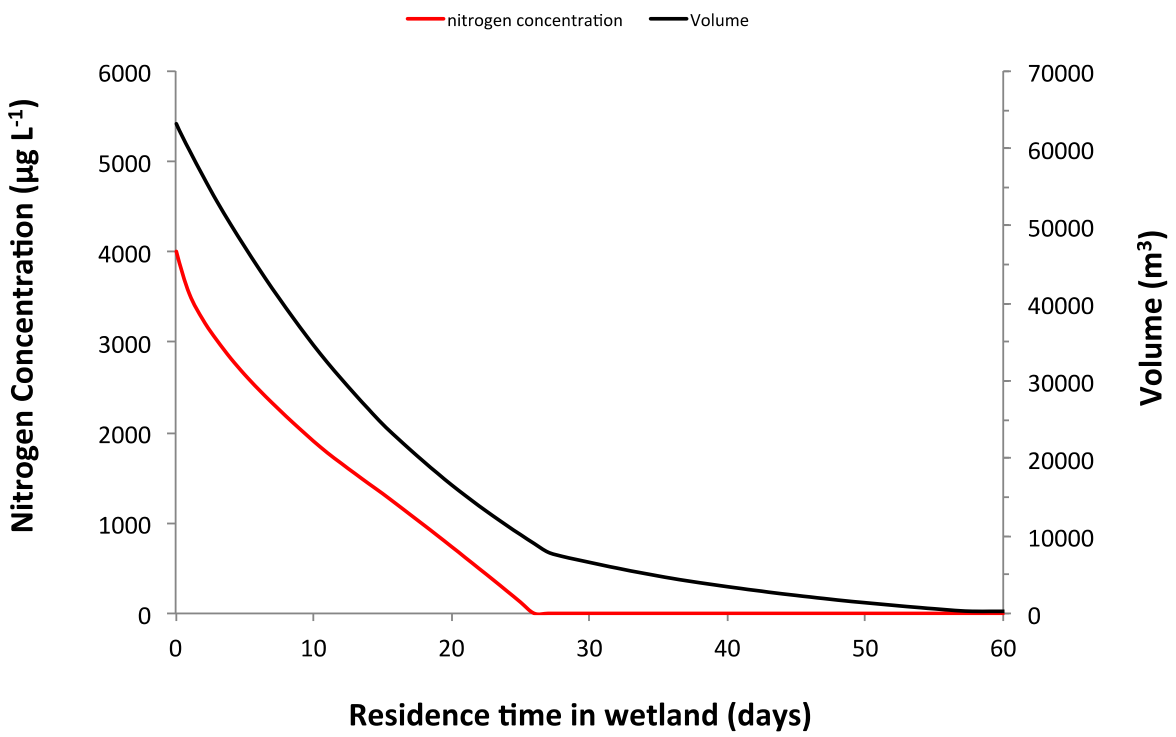

Figure 22: The change in nitrogen concentration in the wetland (red) as the wetland volume (black) decreases due to evaporation and drainage. In this simulation the average rate of water column denitrification was used (427 µg N L^{-1} d^{-1})

The way in which the concentration of nitrogen varies with residence time is very dependent on the water column denitrification rate. This is illustrated in Figure 23, which shows the predicted time course of nitrogen concentration at three denitrification rates, 0.34, 427 and 768 µg N L^{-1} day^{-1}. As expected, nitrogen concentration is lowest at the highest denitrification rate and approaches zero after 14 days. At the average denitrification rate nitrogen concentration decreases more slowly, but still reaches zero by day 26. Simulations at the lowest denitrification rate show an initial drop in nitrogen concentration, followed by steady rise. This occurs because volume decreases that concentrate the nitrogen are greater than the slower loss of nitrogen due to denitrification only. The irregular shape of this curve after ~ day 27 is due to irregular topography of the wetland (see Section 2.1.1) which affects the rate at which the wetland volume decreases (Figure 23). This simulation demonstrates that nitrogen concentrations could become very high in water where the denitrification rate is very low.

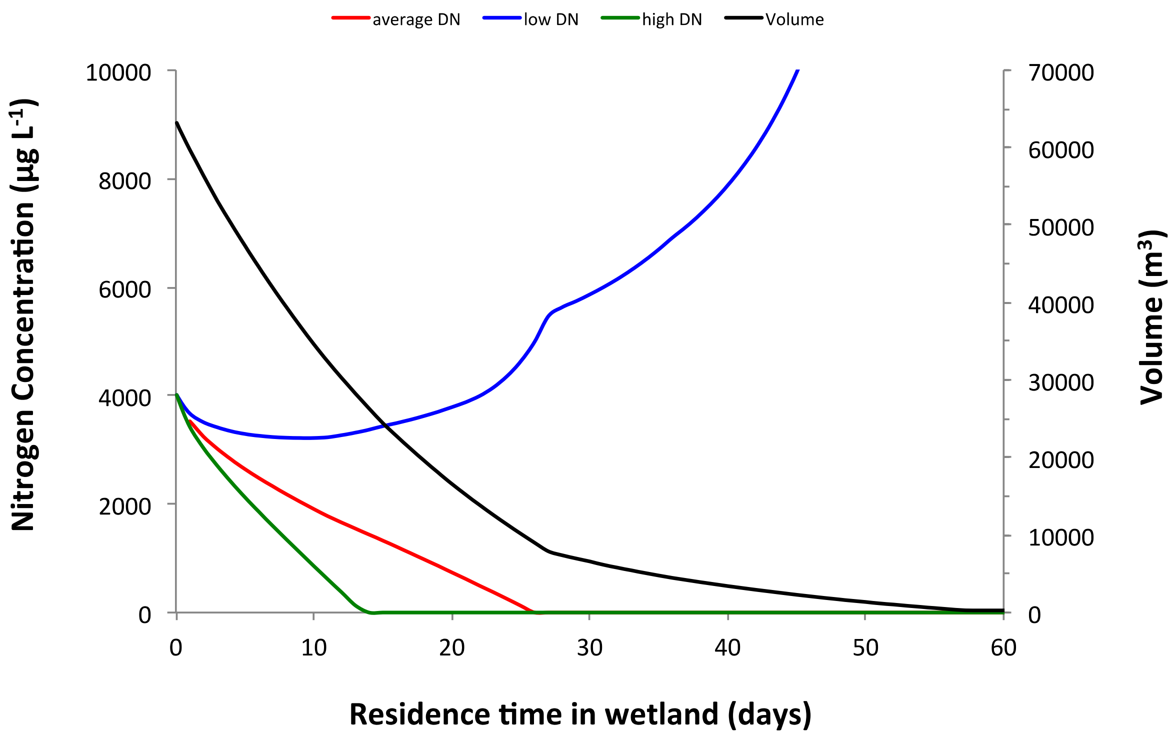

Figure 23: The change in nitrogen concentration in the wetland with time at the low denitrification rate (blue); average denitrification rate (red) and high denitrification rate (green). The decrease in wetland volume due to evaporation and drainage is also shown (black)

3.3.4 Denitrification from within the soil

Estimates of the cumulative denitrification from within the soil over time are shown in Figure 24. At the lowest soil denitrification rate (33 mg N m^{-2} day^{-1}), 99 kg of nitrogen is lost from the anaerobic wetland soil after 60 days when the wetland becomes empty. After this, no further soil denitrification was assumed to have taken place, as the soil would become oxygenated due to exposure to the air. The anaerobic soil area is assumed to be that which is covered by water. As the wetland soil surface slopes (downwards) irregularly from west to east (see Section 2.1.1), the area of the wetland decreases non-linearly as the wetland depth decreases over time, Figure 23. Total denitrification from within the soil increased to 144 kg when the average soil denitrification rate was used (48 mg N m^{-2} day^{-1}) and reached 220 kg at the high soil denitrification rate (73 mg N m^{-2} day^{-1}).

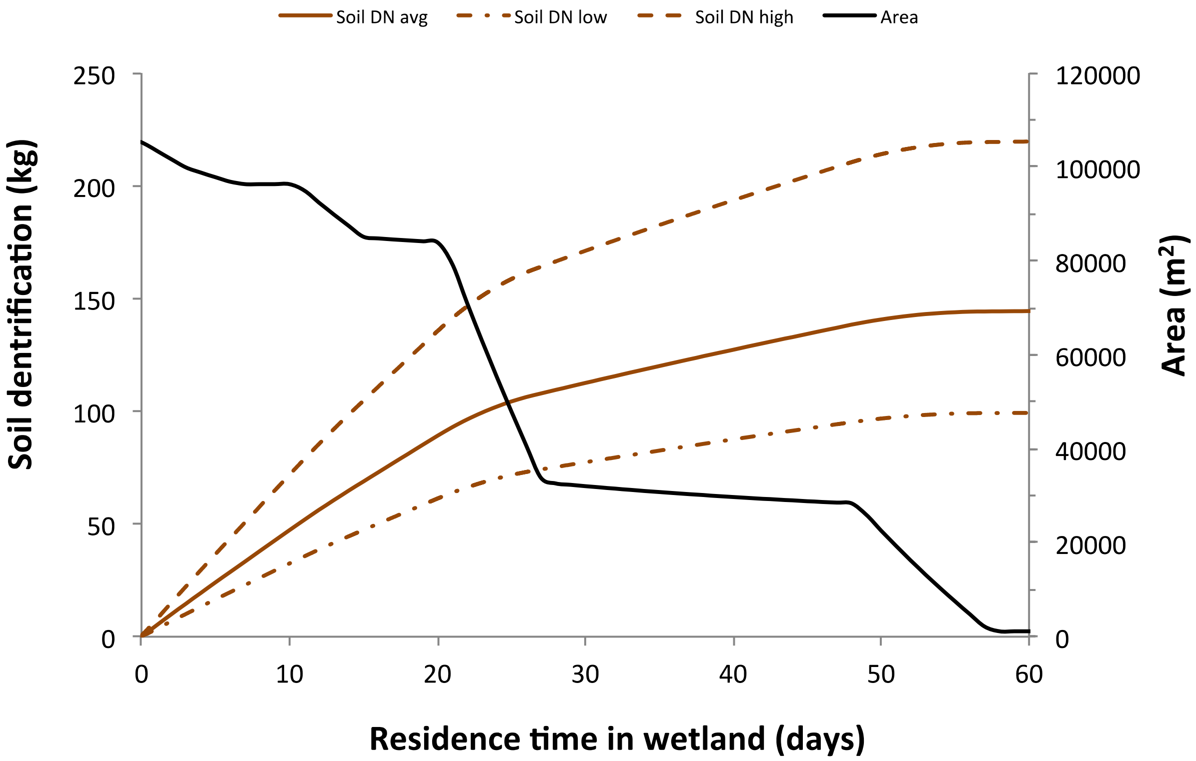

Figure 24: The increase in denitrification from within the soil (brown) with the residence time of water in the wetland. Calculations are shown for three soil denitrification rates; low, 33 mg N m^{-2} day^{-1}; average, 48 mg N m^{-2} day^{-1} and high, 73 mg N m^{-2} day^{-1}. Also shown is the decline in wetland area over time (black)

3.3.5 Wetland denitrification summary

A summary of the soil and water denitrification losses in the wetland is shown in Table 5. The total water column denitrification is the sum of gaseous denitrification and nitrogen loss due to particulate sedimentation (PN). Nitrogen dissolved in drainage water is not included, as it will remain in solution outside the wetland unless conditions become suitable for further gaseous denitrification off site. As PN sedimentation adds to the nitrogen stored in the existing sediment and organic materials in the soil, the net soil denitrification (in Table 5) is given by its gaseous denitrification less the nitrogen addition due to particulate sedimentation.

At all of the soil denitrification rates used there is a net loss of soil nitrogen of between 40 and 170 kg over the full 60 days of the simulation. However, loss of nitrogen from the wetland soil in excess of the addition of PN will reduce the storage of nitrogen in the soil, but it is only the nitrogen that is removed from the water that is effectively 'processed' by the wetland. Table 5 shows that total water denitrification for a single flood pulse ranges from 59 kg at the lowest water denitrification rate, to 190 kg at the highest denitrification rate: equivalent to between 5.9 and 19 kg per hectare. These losses are dominated by denitrification (DN) when the denitrification rate is average (66% DN) or high (74% DN). However, when the denitrification rate is below ~ 200 μg N L^{-1} day^{-1}, DN is no longer the largest nitrogen loss mechanism and only constitutes 0.2% when the denitrification rate is as low as 0.34 μg N L^{-1} day^{-1}. As the total initial amount of nitrogen in the wetland water is 253 kg, then the effective wetland processing ranges from 23 to 75%. On average, 63% of the nitrogen entering the wetland as a single storm water pulse would be lost via PN sedimentation and water column denitrification in the following 26 days.

Table 5: The net change in nitrogen content (kg) of the soil and water in the Babinda wetland after filling with storm water with a total nitrogen concentration of 4000 μg L^{-1}

		Water column denitrification rate (μg N L^{-1} day^{-1})		
		0.34	427	768
Soil denitrification rate (mg N m^{-2} day^{-1})				
33	soil	-40	-45	-49
	water	-59	-160	-190
	total	**-99**	**-206**	**-239**
48	soil	-85	-91	-95
	water	-59	-160	-190
	total	**-144**	**-251**	**-284**
73	soil	-161	-166	-170
	water	-59	-160	-190
	total	**-220**	**-326**	**-359**

From the above denitrification modelling it can be concluded that the effective processing of nitrogen by the Babinda wetland is given by the sum of losses due to sedimentation of particulate nitrogen (PN) and gaseous denitrification in the wetland water. Any soil denitrification in excess of PN will alter the storage of nitrogen in the wetland soil, but this does not necessarily remove any further nitrogen from the water column. Similarly, losses of nitrogen from the wetland in drainage water do not constitute wetland 'processing' as the nitrogen remains in the drainage water as it moves out of the wetland and into the surrounding ditches, streams and aquifer.

3.3.6 Annual nitrogen processing of the wetland

Our final estimate of wetland nitrogen losses has been made over the entire monitoring period between October 2017 and June 2018 (250 days) during which there was a series of 7 flood pulses that entered the wetland (see Figure 14). In this simulation we have used the measured daily depths to calculate the wetland volume, from which the daily denitrification was calculated using the average tropical wetland value of 427 μg N L^{-1} day^{-1} Drainage from the wetland was also calculated from the measured changes in depth, allowing for losses due to evaporation. In line with our previous denitrification calculations we have also made the following assumptions:

1. The initial nitrogen concentration (dissolved and particulate) in the water entering the wetland during flood pulses is 4000 μg L^{-1}.

2. The particulate fraction of the total nitrogen in the water entering the wetland is 30%.
3. The amount of time that dissolved oxygen concentration is below 2 mg L^{-1} is 30%.

Figure 25 b shows that each time a flood pulse raises the water level in the wetland above the berm wall the main initial loss of nitrogen is via sedimentation of PN, which lasts for 12 days after each flood event (Figure 25 b). There is no further loss of nitrogen by this mechanism until the next flood pulse. Gaseous denitrification losses from the water column are proportional to the volume of water in the wetland, which was initially low, but increased markedly during the January to April 2018 wet season. Note that we have only included nitrogen losses from the water within the wetland and not from the water above the berm wall height. This is because water above this level simply flows across the wetland and any losses (or gains) of nitrogen from this water is not part of the wetland processing.

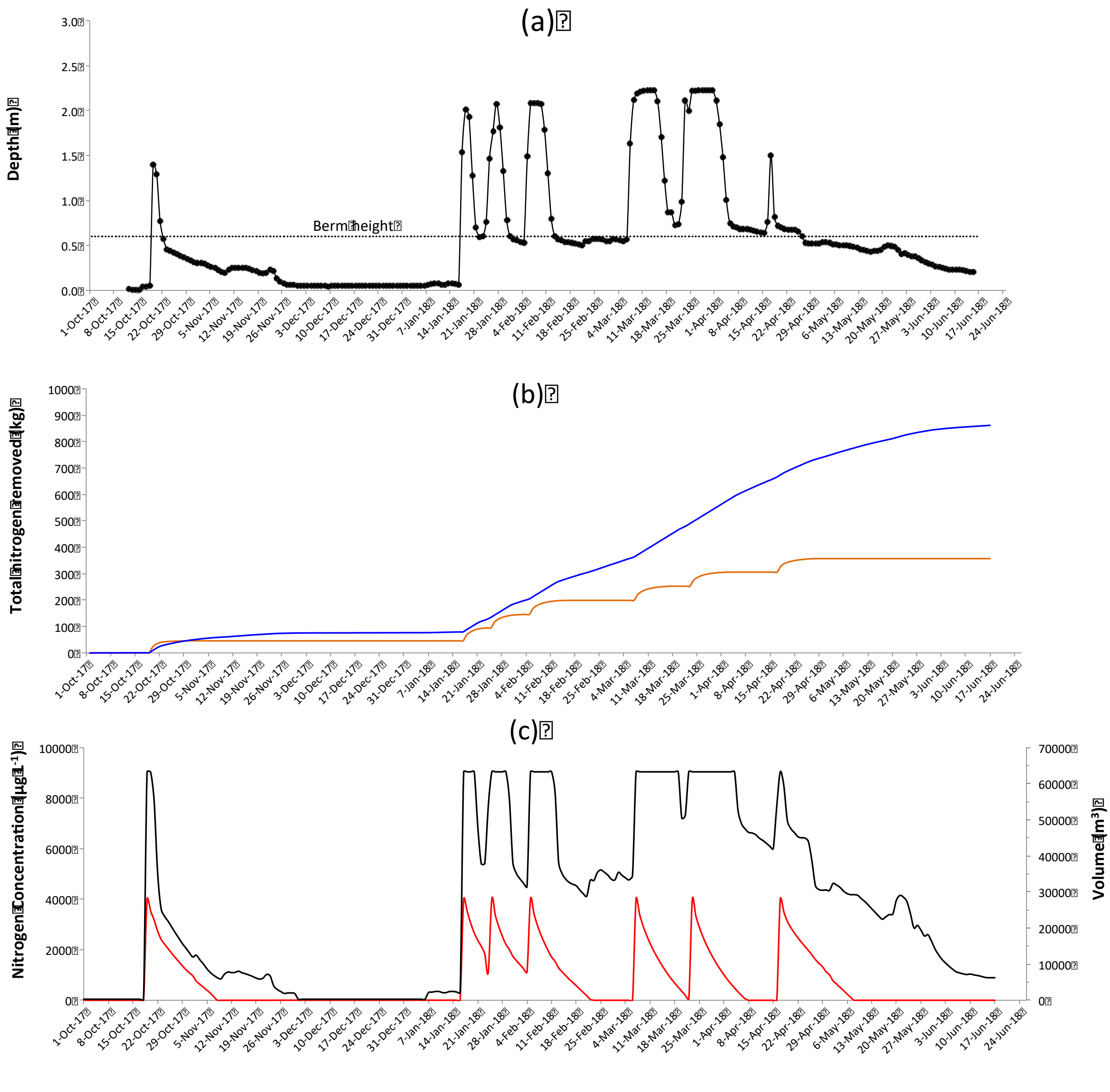

Figure 25: Time series of (a) depth, (b) nitrogen PN loss (orange) and denitrification loss (blue) and (c) wetland water nitrogen concentration (red) and volume (black).

Over the entire 250 day monitoring period (October 2017 and June 2018) the seven flood pulses brought a total of 1771 kg of nitrogen into the wetland. The total loss of nitrogen was 999 kg (65% as gaseous denitrification and 35% as PN), so the wetland was able to process 56% of the nitrogen that entered it. The remaining 44% of the nitrogen left the wetland in its drainage water (or was swept out by the flood pulses), but as stated above, this is not part of the processing effect of the wetland. After each flood pulse the concentration of nitrogen in the wetland decreased fairly rapidly, reaching zero when there was more than 3 weeks between flood pulses.

The estimation of the denitrification in the wetland was extrapolated to the full year October 2017 to September 2018 using the combined water balance and denitrification model to calculate denitrification for the period 18 June to 30 September 2018. . This added a further flood pulse (in July 2018) and gave an annual total denitrification of 1159 kg, equivalent to a rate of 116 kg N ha^{-1} $year^{-1}$. If the higher denitrification rate measured in the Babinda wetland (768 µg N L^{-1} day^{-1}) is used, then this figure increases to 141 kg N ha^{-1} $year^{-1}$.

The estimation of annual nitrogen removal by wetlands by extrapolating measured hourly rates can be misleading. For example, the higher hourly rate in the Babinda wetland is 768 µg N L^{-1} day^{-1}. This can be scaled up using a sample depth of 0.2 m to an annual rate of 561 kg N ha^{-1} $year^{-1}$. This is four times the rate derived above using the same hourly rate in the water balance and denitrification model. The difference is mainly due to ignoring the seasonal hydrology of the wetland and the periods when denitrification does not take place because the dissolved oxygen level is too high.

The concentration of nitrogen was measured during the denitrification field trials carried out on 14 June 2018 (see section 2.3). These were 22 µg L^{-1} of N-NOx and 22 µg L^{-1} of N-NH_4, giving a total nitrogen concentration of 44 µg L^{-1}. We iteratively adjusted the water column denitrification rate used in the above model to match this value on this day and got a value of 93 µg N L^{-1} day^{-1}, significantly lower than the mean value of 427 µg N L^{-1} day^{-1} used in the above simulations. Using the low denitrification value of 0.34 µg N L^{-1} day^{-1} yielded a nitrogen concentration of 1287 µg L^{-1} on 14 June 2018, much higher than the value we recorded in the field. Using the high denitrification value of 768 µg N L^{-1} day^{-1} led to the nitrogen concentration reaching 0 µg L^{-1} over six weeks before 14 June 2018. On the basis of this single direct measurement of nitrogen concentration, it would appear that the mean denitrification value of 427 µg N L^{-1} day^{-1} based on short term (6 hours) may be higher than the effective annual mean value in the Babinda wetland (93 µg N L^{-1} day^{-1}). Clearly, more measurements of nitrogen concentration in the wetland at a range of times after flood pulses would be needed to corroborate this.

The Reef 2050 Water Quality Improvement Plan has set target reductions in dissolved inorganic nitrogen (DIN) and particulate nitrogen (PN) loads for each catchment that drains to the Great Barrier Reef. For the Mulgrave-Russell catchment a reduction in DIN of 300 tonnes and PN of 53 tonnes by 2025 is recommended. The potential contribution of wetlands with the same average denitrification properties as the Babinda constructed wetland (i.e. 116 kg N ha^{-1} $year^{-1}$) was estimated as ~ 40 ha per 1 percent of the 2025 DIN reduction target. For example, 10% of the DIN reduction target could be achieved from 398 ha of wetland, which amounts to 1.5% of the total sugarcane area in the Mulgrave-Russell catchment. This area of wetland would also remove 30% of the 2025 PN reduction target. To remove all of the PN, the area of wetland would have to increase to ~ 1305 ha, or 5% of the total sugarcane area. This area would also remove a third of the DIN reduction target.

3.3.7 Potential for denitrification in the dry season

If water can be pumped into the wetland from the adjacent drainage ditch during the dry season then there may be potential for enhancing its annual denitrification. For example, if sugar cane irrigation pumps were available then a 60 mm pump could fill the wetland in ~ 12 days (using a pump rate of 60 L s^{-1}). This duration could be shortened using multiple pumps. After the wet season has been over for several weeks, the water in the adjacent drainage ditch will have lost most of its particulate nitrogen (by sedimentation) and the dissolved nitrogen concentration will have declined below its flood peak value (by denitrification). Water quality measurements made in Dickson Creek, just upstream of the drainage ditch, show that suspended sediment concentration is ~ 10% of its peak value and nitrogen concentration is ~ 500 µg L^{-1} when the effect of the flood pulse has declined. Water pumped into the wetland in the dry season may therefore have a nitrogen concentration around 500 µg L^{-1} with a PN fraction of ~ 3% (1/10th of the 30% assumed for flood water). Using these values in our water balance and denitrification model with the average denitrification rate (427 µg N L^{-1} day^{-1}), in the 60 days for the wetland to empty from full, ~ 30 kg of nitrogen would be lost from the wetland water, mostly (99%) as gaseous denitrification. However, all of the nitrogen is lost in the first 4 days (since suitable DO conditions are assumed to occur 30% of the time), so after this time there is no further need to retain the water in the wetland.

If it takes 12 days to fill the wetland, 4 days to remove the nitrogen and a further 4 days to empty the wetland by opening 3 sluice gates (see Figure 17), the total time to remove 30 kg is around 20 days. If the dry season were to last ~ 4 months, this means that the wetland could be filled and emptied up to six times, giving a total nitrogen loss of ~ 180 kg, adding a further 16% to the annual mean loss above (1159 kg). There would of course be a cost to be taken into account as the hire of the pump/s and their fuel costs.

4.0 CONCLUSIONS AND RECOMMENDATIONS

4.1 Learnings from this wetland project

Conversion of low-lying, low productivity, sugarcane land to treatment wetlands represents a new and additional management strategy for government in advancing towards achieving the water quality targets for protection of the reef. There is increasing support for this land conversion for landholders, and particularly if there are market mechanisms available to incentivise projects. This is the first project to evaluate a treatment wetland in northern Queensland and therefore provides some important learnings to consider when designing future treatment wetlands. These are outlined below.

4.1.1 Outlet gates

The number of outlet gates used in this design (12 in total) are an overdesign. The rate of water discharge when opening three gates would see the wetland drain to the bottom of the outlet pipes in a relatively short period of time (~ 4 days), sufficient for the management of this wetland. Reducing the number of gates would also reduce the overall project costs, with savings achieved by using fewer gates and less on ground construction works. The desired number of gates for a project site should be modelled during the design phase of the project, based on the hydrological process presented in this project.

4.1.2 Remote control gate management

The manual gates installed at this wetland does require a staff member to access the site to open and close gates (both inlet and outlets). This presents several problems, including safety accessing the site after flood events where crocodiles are known in the region, which could be in the wetland. Furthermore, because of the low lying nature of the region the site is not accessible for several weeks after high flow events (i.e. access roads are impassable). By not attending the site following rainfall events there is a chance that water in the wetland could escape either through the outlet pipes, or back out through the inlet pipe. There is a possibility in this instance that inlet and outlet gates could be controlled via remote access automation, where from the safety from an office site the gates could be opened and closed remotely. This technology is available, however, would increase the upfront capital costs during the construction.

4.1.3 Maintenance after construction

Maintenance of this wetland will be necessary, either in the form of desilting accumulated sediments that enter and settle in the wetland, repairs to gate structures that are damaged during major flooding or debris, damage to access roads during major flooding, or accumulation of invasive weeds that may choke the wetland. The cost for this maintenance will vary depending on the frequency and scale of works required. The exact costs for this maintenance is unknown at this stage, but should become available as more similar projects come on line in the region.

4.1.4 Validation data

The need to test the efficacy of the wetland in reducing nutrients, sediment (and pesticides) requires an appropriate level of sampling and scientific rigor. Water sampling campaigns can be completed via several different approaches, and each has risks and advantages.

The most appropriate water sampling methodology for monitoring wetland nutrient and sediment concentrations is the use of refrigerated auto-samplers (Hawdon et al., 2007, see Figure 26). These auto-samplers ensure water samples are collected across the entire hydrograph (Figure 27) so that the true peak concentrations and loads to the wetland can be calculated. Note that on many occasions rainfall events and the peak of the hydrograph occurs at night, which means field staff accessing the sites during the day to take grab water samples will not capture peak flow stages, where concentrations are highest, and will therefore only sample part of the hydrograph. In addition, field staff safety when accessing sites could become problematic to collect water samples. For these reasons, at least during the initial first few years of monitoring, it is recommended that auto-samplers are installed, which are programmed to trigger sampling into an internal refrigerated well of container chambers (the Avalanche refrigerated portable sampler shown in Figure 26 has a maximum of 14 bottles) that progressively collect samples across the hydrograph (Figure 27). Importantly, until the hydrology of each wetland is known the programming of the auto-samplers may require some adjustment. As a starting point, at least 2-3 flow hydrograph events should be monitored, following the optimised sampling procedure described by Hawdon et al (2007).

Figure 26: Example of an auto-sampler system installed on the Murray River floodplain (reproduced from Hawdon et al., 2007)

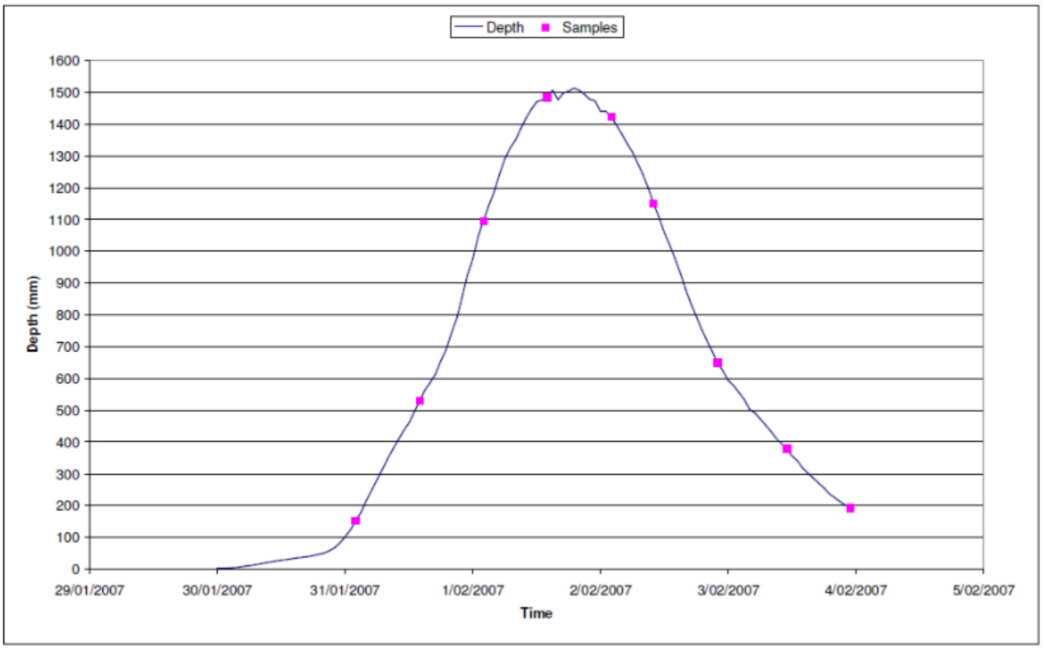

Figure 27: Example of flood depth and sampling times for auto-sampler during a four day flood (reproduced from Hawdon et al., 2007)

The use of a refrigerated auto-sampler also assists with complying with water sample protocols, where unchilled water samples left for several days would be compromised. This could result in data that is not useful for interpretation, particularly in tropical environments.

Auto-samplers can also be programmed to take a composite sample of water that represents the entire hydrograph during a single flow period. While this might be a useful step, the time interval set to collect the composite sample is not yet known, but could be determined after a few flow events pass through the wetlands.

Another advantage with installing auto-samplers is that once water samples are collected, processed after collection, and stored appropriately in freezers (possibly by the local land holder as in-kind contribution to the project), samples can remain on hold until a later date, for example, until several hydrograph events are sampled, with then a suitable subset of samples proceeding through to laboratory analysis. This has the advantage of saving analytical costs, but also allows for examination of specific flow events. Use of nitrate probe technology is not recommended here, on the basis of on-going maintenance and calibration checks to account for probe drift over time, the potential for fouling and loss of loggers during flood flow, and importantly, the mere fact that probes only measure a single nutrient parameter (compared to the full suite of nutrients, turbidity and total suspended sediments which can be measured when using auto-samplers).

4.2 Wetland inundation duration

With the high rainfall (annual average 4287 mm) in the Babinda area the constructed wetland is frequently filled with water, particularly during the wet season. In the 2017/18 wet season, severe flooding in this catchment raised the water level in the wetland to over 2 m for nearly

10% of the time monitored (12 November 2017 to 15 June 2018). Almost 80% of the wetland area remained inundated by over 30 cm of water for 61% of the time. These long inundation periods enhance the potential for denitrification of the water within the wetland.

When water levels are above the berm wall, drainage is very rapid, with water flowing freely into the surrounding land. Once the water level drops below the berm wall, drainage is much slower, less than 1% of the over berm flow. However, this is still a significant component of the wetland water balance and it is a key factor (along with evaporation) affecting the duration of inundation when rainfall stops. It appears that high water levels outside the berm may reduce the internal drainage rate, thereby prolonging the wetland inundation duration.

The wetland can be emptied in a little as ~ 1 day with all 12 outlet gates open. However, this only drops the water level to the bottom of the outlet pipes, when 29% of the wetland is still inundated. Drainage and evaporation continue to lower the water level, but it would take nearly 4 weeks to become completely empty.

4.3 Wetland denitrification

4.3.1 Preliminary conclusions

Estimates of soil and water denitrification in the Babinda wetland vary widely as they are based on the range of denitrification rates found in tropical wetlands in Australia (McJannet et al., 2012, Adame et al., 2019a and Adame et al., 2019b and the current study – see Table 2). The latter three studies of tropical wetlands along the north Queensland coast found water denitrification values in the range 16 to 211 (mean 114) mg of N m^{-2} d^{-1}; the higher values are representative of denitrification 'hot spots' where wetland conditions at certain times of the year support rapid conversion of dissolved nitrogen into nitrogen gas. In contrast, (McJannet et al. 2012) made long term estimates of water column denitrification rates in a tropical floodplain lagoon in the Tully catchment north Queensland, finding monthly mean values in a 3 year study between 0 and 3.1 (mean 0.52) mg of N m^{-2} d^{-1}, less than 1% of the values found here and by Adame et al., (2019a,b). The McJannet et al. (2012) study therefore represents a wetland denitrification 'cold spot', where the hydrological conditions for most of the year do not allow much denitrification. In any tropical catchment floodplain there is likley to be a range of wetland types, some hot spots and some cold spots.

Some of the difference in the above studies are due to the very different methods used; Adame et al., 2019a,b derived potential denitrification rates over periods ~ 6 hours using isotopic tracing, whereas McJannet et al. (2012) calculated actual monthly and annual rates from a water and nutrient balance taking into account periods of low dissolved oxygen. The much lower denitrification rates in the latter study also reflect the short residence times of water in their riverine wetland lagoon. In contrast, soil denitrification rates in the three studies above are much more similar, Table 2.

All of the suspended sediments entering the wetland are retained provided there is more than 12 days between flood pulses. The sedimentation of particulate nitrogen is a significant mechanism for loss of nitrogen from the wetland, which reduces the concentration of nitrogen in the water leaving the wetland as drainage; hence it is an important part of the processing effect of the wetland. Sedimentation adds to the nitrogen load in the wetland soil, where denitrification can occur at rates comparable with those in the wetland water. This soil denitrification can reduce the amount of nitrogen stored in the soil, but does not necessarily affect the concentration of nitrogen in the drainage water as this mainly occurs as comparatively rapid seepage through the berm walls. Further studies of the relative amounts of seepage and groundwater (along with their respective nitrogen concentrations) in the wetland drainage are required to verify this conclusion.

Large amounts of nitrogen can be lost from the wetland as drainage, which contains dissolved inorganic and/or organic nitrogen. In the Babinda wetland this mechanism accounted for 44% of the annual loss of nitrogen from the wetland, meaning that the processing capacity of the wetland was 56%. Other studies of constructed wetlands have also found drainage (as groundwater losses) to be the main nitrogen loss mechanism (Batson et al. 2012). Once outside the wetland further denitrification may or may not take place, depending on the conditions in the nearby, ditches, streams or aquifer. If the drainage mainly appears in nearby streams and/or ditches, gaseous denitrification may continue at similar rates to those found in the wetland. Using the average wetland denitrification rate in this study it would take 26 days to loose the nitrogen in the drainage water, by which time it may well have travelled to the main river and even into the nearby ocean.

The estimates of how the nitrogen concentration in the wetland changed with residence time varied widely according to the value of water column denitrification rate used. This provides an opportunity as measurements of the initial nitrogen concentration of water entering the wetland and further periodic (~ every 7 days) concentration measurements in the wetland would greatly help identify which of the wide range of water column denitrification rates is closest to the actual rate in the Babinda wetland. Ideally, these nitrogen concentration measurements should include particulate nitrogen and dissolved inorganic and organic nitrogen. The single measurement of dissolved inorganic nitrogen made during the denitrification trials on 14 June 2018 indicate that the mean water column denitrification rate of ~ 93 μg N L^{-1} day^{-1} may be close to the effective annual value for the Babinda wetland.

Until more *in situ* measurements of nitrogen concentration become available our current estimate of wetland nitrogen loss is based on average denitrification rates from the four tropical wetland studies in north Queensland. These show that over an annual cycle ~ 56% of the total nitrogen entering the wetland is lost at a rate of 116 kg N ha^{-1} $year^{-1}$. Most of this loss is due to gaseous denitrification in the water column (65%), with the rest due to sedimentation of particulate nitrogen (35%).

4.3.2 Recommendations

As denitrification in the wetland water appears to be the dominant nitrogen loss mechanism it is vital that periodic measurement of dissolved nitrogen (as nitrate, DON and ammonium) over time are required for testing and improving the wetland water column denitrification model. The most reliable way to do this is to use refrigerated auto-samplers, as described in Section 4.1.4 above. PN is also a major component of the wetland denitrification, so PN concentrations should also be measured along with the rate at which the suspended sediment declines after the wetland is filled with floodwater. This could be done using the above refrigerated auto-samplers or with less expensive passive rising stage samplers that capture water as it enters the wetland (e.g. see Hawdon et al. (2007). The suspended sediment in these samples can be measured back in the laboratory and some of the sample water should also be used to determine the settling rate of the different soil fractions.

The duration of low dissolved oxygen (< 2 mg L^{-1} or 25% saturation) needs to be monitored in the wetland, as this is a critical parameter in determining gaseous denitrification. It would also be useful to determine the relationship between dissolved oxygen concentration and denitrification rate in the tropical waters present in the north Queensland wetlands.

Table 6 below summarises the parameters used in the wetland denitrification model, along with the current values used, their origin and any further updating that may be required. Also shown is the sensitivity of the calculated water column denitrification over 60 days to each of the input parameters (using the average rate of water column denitrification, 427 µg N L^{-1} day^{-1}). The calculated denitrification is most sensitive to the value of the initial nitrogen concentration and least sensitive to the value of evaporation rate. Most of the other controlling variables change denitrification by ~ 2 - 5% as their value changes by 10%.

This study has identified the key parameters that need to be measured in order to improve the estimation of nitrogen processing of wetlands. It is highly recommended that future constructed wetland design should include these measurements. Long-term estimation of wetland processing requires the wetland hydrology to be taken into account and further monitoring of the key components of the wetland water balance (especially seepage and groundwater) are needed to achieve this. To reduce the current uncertainty in denitrification estimation, additional *in situ* measurements of the soil and water denitrification rates and how they vary seasonally are required.

Table 6: List of parameters examined, with the average calculated and used in the wetland denitrification model

	Parameter	Value used	Sensitivity*	Comment
1.	Bank full maximum depth	0.85 m	NA	Needs to be verified with measurements of berm and spillway height.
2.	Wetland area (when full)	105400 m^2	NA	Derived from Australian Wetlands Consulting survey
3.	Wetland volume (when full)	63251 m^3	NA	Derived from Australian Wetlands Consulting survey
4.	Sedimentation rate in flood water	see Appendix A1	2.4%	Rate of TSS decline taken from Wallace et al., 2008. Needs updated using *in situ* TSS measurements
5.	Duration when DO below 2 mg L^{-1}	30%	3.4%	From McJannet et al., (2012b). Needs replaced with *in situ* DO measurements
6.	Initial total nitrogen concentration	4000 µg L^{-1}	6.6%	From McJannet et al., (2012b). Needs replaced with *in situ* TN measurements
7.	Particulate fraction of TN	30%	1.9%	From McJannet et al., (2012b). Needs replaced with *in situ* PN measurements
8.	Denitrification rate in water	0.34 to 768 µg N L^{-1} day^{-1}	3.4%	From literature, see Table 2.1
9.	Soil denitrification rate	33 to 78 g N m^{-2} $year^{-1}$	NA	From literature, see Table 2.1
10.	Evaporation rate	4.6 mm day^{-1}	0.16%	Calculated using SILO data in a wetland evaporation model (average could be replaced with actual daily values)
11.	Drainage rate	$DR_w = 0.034\ d$	3.80%	Derived from depth (d) changes when there was no rainfall

* Sensitivity is the percentage change in water column denitrification over 60 days for a 10% change in the given parameter.

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APPENDIX 1: Sediment concentration in Tully-Murray floodwater.

Measurements of the concentration of suspended sediments (TSS) in the flood waters on the Tully-Murray floodplain were made during three flood events in 2007 (Wallace *et al*., 2008). These data are used here to estimate the initial TSS and subsequent rate at which this falls for overbank flood events generated by high rainfall in catchments in northern Queensland which have rainforest covered headwaters and extensive sugar cane agriculture on the floodplain.

Figure A1 shows the time trend in TSS for three flood events in February and March 2007. Measurements were made in the overbank water that was on the floodplain during each event.

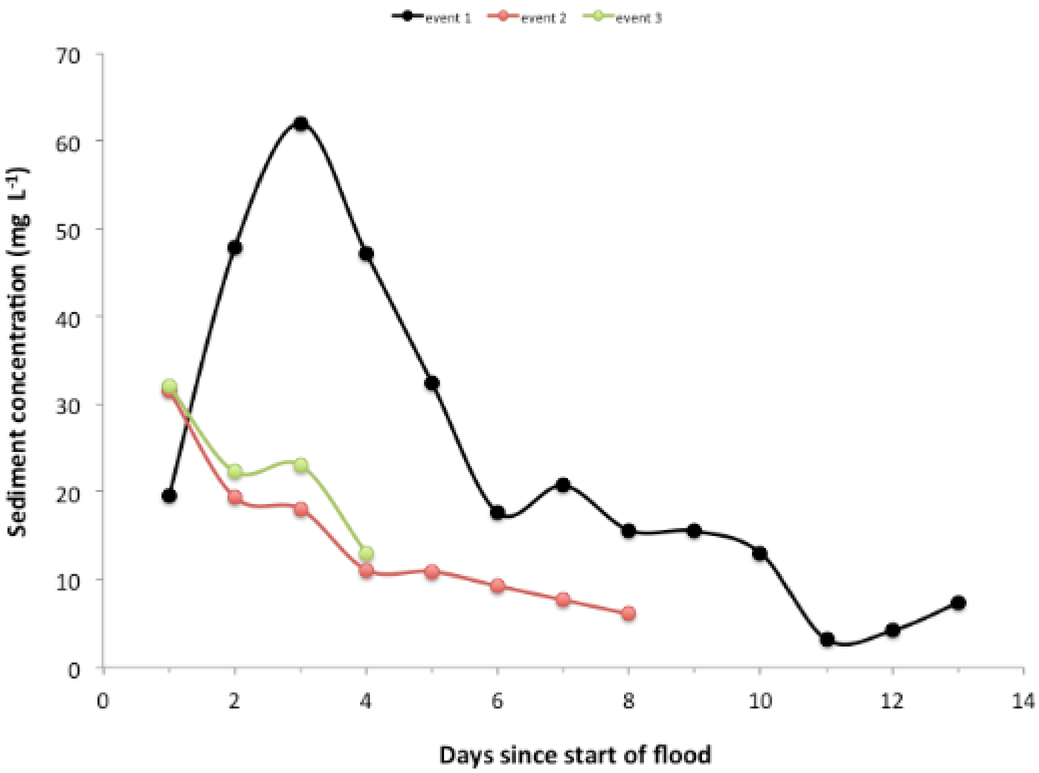

Figure A1.1: The change in total suspended sediment (TSS) concentration with time after the start of an overbank flood for three events in 2007 (30/01 to 12/02/2007; 19/02 to 26/02/2007 and 21/03 to 24/03/2007)

The highest TSS concentration, 62 mg L^{-1}, was recorded 3 days after the start of the first flood of the season. This is referred to as the 'first flush' and after this peak, TSS concentration fell steadily over the next 10 days. Subsequent floods produced a lower peak TSS, 32 mg L^{-1} , and this occurred on the first day of the flood. This suggests that at the start of the first event a large store of labile sediment was available for incorporation into runoff waters. After the first flush, this sediment store is depleted, hence the lower TSS concentrations in subsequent floods.

Previous studies of sediment concentrations in 'first flush' river flows in the Daintree, Mossman and Saltwater Creeks range from 150 to 700 mg L^{-1} (Wallace et al., 2008). However, these measurements come from water contained within the rivers banks during high flows and Wallace et al (2008) explain the lower TSS concentrations recorded on the Tully-Murray floodplain as being due to dilution by the very large amounts of overbank water discharged during floods.

The data in Figure A1 can be combined to give an estimate of how rapidly TSS concentration declines after reaching its peak value in any flood. This can be done by plotting the relative TSS (i.e. the ratio of TSS to peak value TSS_p) against the number of days since the peak TSS occurred, as shown in Figure A2.

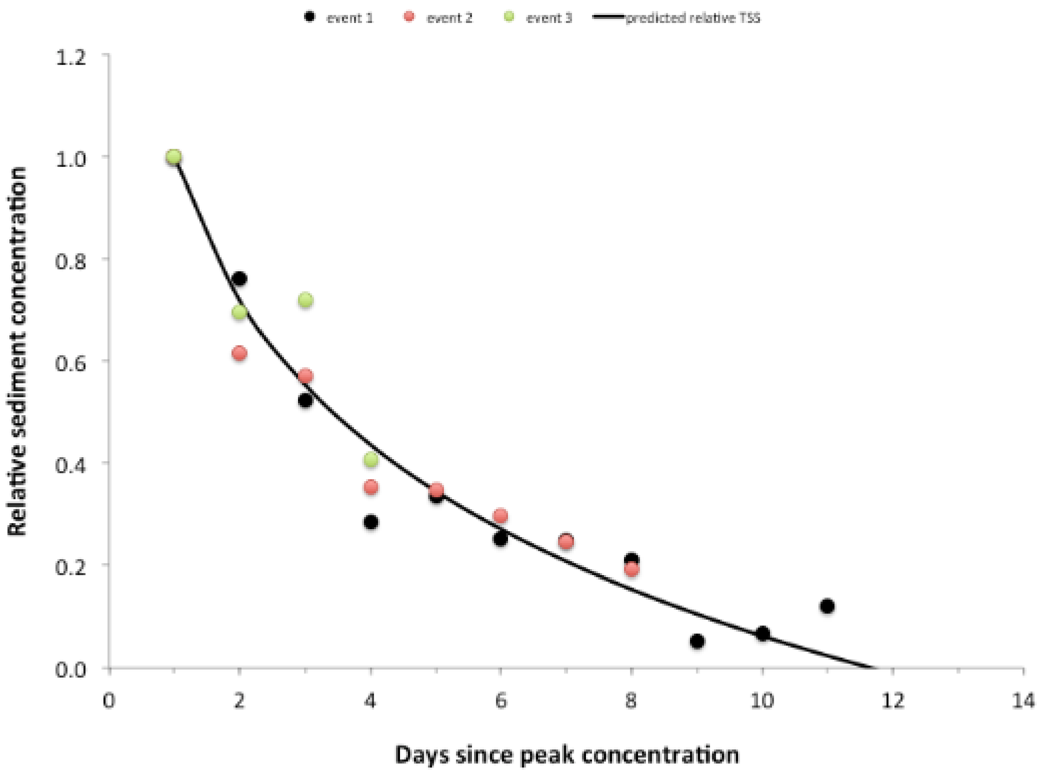

Figure A1.2: The decline in relative TSS concentration with days (*t*) after the TSS peak concentration. The fitted line has the form TSS_{rel} = -0.407*ln (*t*) +1 ; r^2 = 0.95

In all 3 flood events, relative TSS declines at a similar rate, which is quite rapid initially as the larger sediment particles settle out of the floodwater. Smaller particles settle more slowly, and so TSS concentration falls less quickly and it takes ~ 12 days for all of the sediment to precipitate from the water.

The logarithmic line fitted through the data points for all three events has the form;

$$TSS = -0.407 * TSS_p * \ln(t) + 1, \quad \text{(A1)}$$

and this equation can be used to estimate the actual value of TSS on any day after the initial flood, if the peak TSS concentration is known.

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