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On the potential for improving water quality entering the Great Barrier Reef lagoon using constructed wetlands

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ABSTRACT

The Reef 2050 Plan has identified a range of measures aimed at reducing end-of-catchment loads of sediment and nutrient and recognizes the role that freshwater wetlands may have in achieving this. However, quantitative information on the potential for tropical wetlands to filter agricultural runoff is scarce, so this paper describes a study that combines field data from a 10 ha wetland constructed on land previously used for sugar cane near Babinda, north Queensland with a water balance and denitrification model. During the 12-month monitoring period (from October 2017 to September 2018) we estimate that the nitrogen filtering capacity of the wetland was 52% (26% lost as gaseous denitrification from the water and soil, and 26% as sedimentation of particulate nitrogen, PN). The remaining nitrogen (48%) left in the drainage water and this emphasises the importance of the wetland hydrology in determining denitrification and filtering. The current estimates are highly variable, so we have also identified the key parameters that need to be measured in order to improve long-term wetland filtering capacity estimation. Babinda is in the Mulgrave-Russell catchment, where the Reef 2050 Water Quality Improvement Plan has set target reductions in DIN of 300 t and PN of 53 t by 2025. 10% of the DIN reduction target could be achieved from ~593 ha of wetland with the same mean denitrification properties as currently estimated for the Babinda wetland (i.e. 51 kg N ha^{-1} $year^{-1}$). This amounts to 2.3% of the total sugarcane area in this catchment that, as wetland, would also remove 56% of the 2025 PN reduction target.

1. Introduction

The modification and loss of catchment and waterway 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 degradation and loss of freshwater wetlands, forested floodplains, rainforests and other terrestrial and aquatic ecosystems in the catchment (GBRMPA, 2009; Brodie et al., 2013; Waltham et al., 2019). These ecosystems are essential for a healthy, resilient GBR because they reduce pollution loads by slowing surface water flow and trapping catchment sediments and nutrients (Fisher and Acreman, 2004; Mitsch et al., 2005, Adame et al., 2019a). They also improve coastal hydrological connectivity (Karim et al., 2010) and provide habitat for a range of freshwater and marine species (Sheaves et al., 2012; Great Barrier Reef Marine Park Authority, 2014).

Intensive agriculture dominates the coastal floodplains of the Wet Tropics NRM Region and grazing is the major land use in the drier, western parts of the region (Terrain NRM, 2015). Sugarcane production is the dominant intensive agricultural land use on the coastal floodplains 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 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).

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 filtering capacity of the floodplain and increased water velocity, erosion and pollutant transport (Brodie et al., 2004; Great Barrier Reef Marine Park Authority, 2014).

The Wet Tropics Water Quality Improvement Plan identifies that

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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. There is therefore 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). There is mounting evidence that some freshwater wetlands can remove sediments and DIN, thereby reducing the pollution load to the GBR lagoon. For example, in wetlands in the Wet Tropics Region in northern Queensland, Adame et al., 2019a&b found denitrification 'hot spots' where instantaneous nitrogen losses (from water and soil) were between 0.26 and 4.1 (mean 1.5) kg N ha^{-1} day^{-1}. In contrast, a 3 year study of a riverine wetland in the same area, by McJannet et al. (2012) reported lower monthly mean denitrification values of between 0.1 and 1.7 (mean 0.3) kg N ha^{-1} day^{-1}, mainly occurring in the wetland soil (see Table 2 in Wallace et al., 2020). Given the significant potential for sediment and nutrient removal by tropical wetlands, this project was set up to examine the filtering capacity of a constructed wetland in a sugar cane dominated catchment in the wet tropics of north Queensland.

2. Wetland location and methods

2.1. Russell catchment and weather

The constructed wetland is situated in the Russell catchment near Babinda, north Queensland. Prior to development the vegetation was dominated by rainforest, with small areas of melaleuca and eucalypt woodlands and forests. However, current land use on the floodplain is dominated with sugarcane (~13%), grazing (~8%), urban (~3%) and minor areas of other cropping and services (https://wetlandinfo.des.qld.gov.au/wetlands/). Babinda has a tropical rainforest climate (*Af*) with hot, humid and persistently wet weather. It is the wettest town in Australia, with an annual average rainfall of 4287 mm. The wet season lasts from December to May, while the 'dry season' occurs from June to November. During the wet season, heavy monsoonal downpours occur almost daily and monthly totals over 1000 mm are not uncommon.

Soils in the catchment are highly variable and across the floodplain mainly fertile alluvial soils (Killin, 2011). Adjacent to Babinda Creek they are quite poorly drained Tully, Coom and Timara soil types. Further details of the soil at the constructed wetland site are given by Wallace et al. (2020).

2.2. Babinda treatment wetland

The Babinda wetland is an off channel facility, where water flowing down the Babinda drain (Fig. 1) can be diverted into the wetland using a one-way manual value once the drain level is above the inlet pipe height. Water can also enter the wetland as rainfall and catchment runoff from the west or by overtopping the drain banks, or crossing paddocks from Babinda creek (AWC, 2015; Wallace et al., 2020). There are 12 outlet pipes located along the northern wall of the wetland (Fig. 1) in addition to an outlet weir. The 12 outlet gates are also manually operated to control the release of water from the wetland.

Water depths were monitored every 20 min by loggers (CTD-Diver, Eijkelkamp Soil & Water, Netherlands) located in two positions in the wetland, from September 2017 to June 2018 (Fig. 1). The loggers were downloaded every few months as part of routine maintenance visits. Denitrification was measured on 14 June 2018 using 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. A total of 12 samples were taken from two locations (one wet and one dry) from within the wetland. Further details of the isotopic measurements and their uncertainty are given by Adame et al., 2019a and Wallace et al. (2020). The values obtained in this study were used along with those found in three other studies of tropical wetland denitrification to estimate the Babinda wetland nitrogen losses (see Section 2.4).

2.3. Wetland water balance

A schematic diagram of the key components of the Babinda wetland water balance is shown in Fig. 2.

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



Fig. 1. 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, the outlet weir, and the twelve manually control outlet pipes.

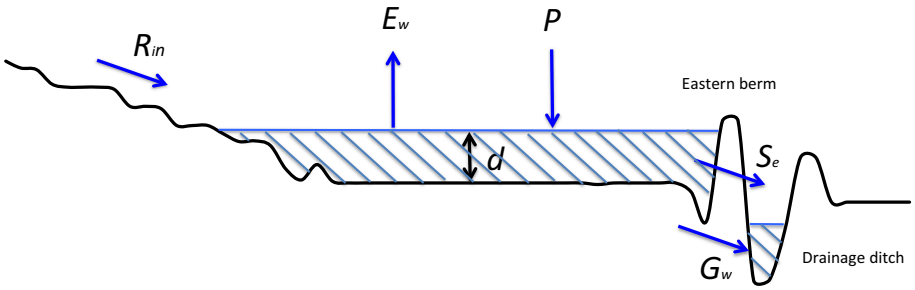


Fig. 2. Schematic representation of the Babinda wetland water balance. When the inlet and outlet pipes are closed, water runs in from the higher land to the west and is contained by the constructed eastern berm wall. There is also a deep drainage ditch next to the berm. The symbols are explained in the text.

as:

$$\delta d = (P + Rin) - (DRw + Ew) \quad (1)$$

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. E_w is rate of evaporation from the inundated part of the wetland, which is dominated by open water, hence we use a method to estimate evaporation from open water described by McJannet et al., 2008 and Wallace et al., 2015. The main input of energy to the evaporation 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/).

Drainage (DR_w) from the wetland was calculated from the daily decrease in wetland depth recorded in the rain free periods following rain events, Wallace et al. (2020). These authors also describe how 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.

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 (Fig. 1). Here we estimate this filtering 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 in combination estimates of particulate settlement and water and soil denitrification. Each of these is described in more detail below.

When water enters the wetland the suspended sediment (TSS) and particulate nitrogen (PN) adhered to it will settle to the bottom of the water column over time. The rate at which this occurs varies in time as larger particles settle more quickly than smaller particles. In the absence of TSS measurements in the Babinda wetland, we have estimated sediment and PN losses using the rate at which TSS declined in the overbank floodwater on the nearby Tully-Murray floodplain (see Wallace et al., 2020). 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.

To calculate the loss of nitrogen due to sedimentation it is also necessary to know the dissolved nitrogen (DN) and PN concentrations of the water entering the wetland. Unfortunately, this was not measured in this study, so an initial estimate has been made here using the typical total (DN 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}. As the concentration of nitrogen in flood pulses can vary with discharge and nitrogen surplus conditions of the surrounding catchment, we have calculated the sensitivity of our results to changes in nitrogen concentration of ±50%.

Another potential pathway for nitrogen loss from a wetland is through gaseous nitrogen (N_2) loss from the water through the process of denitrification. 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).

As the small number of denitrification estimates made in the current study were highly uncertain (coefficients of variation in the range 52 to 76%), we combined these data with values that have been reported in three other studies of tropical wetland denitrification in Australia (McJannet et al., 2012; Adame et al., 2019a, 2019b), details of these are summarized in Table 2 in Wallace et al., 2020. To allow for water column nitrogen losses to vary with wetland depth, we converted the denitrification rates in the above studies to a volumetric basis. This gave a wide range of values varying from 0.34 (McJannet et al., 2012) to 768 μg N L^{-1} day^{-1} (Wallace et al., 2020), with an average value of 427 μg N L^{-1} day^{-1}. These daily rates were converted to the total denitrification loss (kg) from the wetland water column using the calculated volume on each day. The resultant nitrogen concentration at the end of each day was then calculated allowing for the loss due to denitrification, in addition to the losses due to sedimentation and in drainage water. As denitrification is often assumed to follow first order kinetics, the denitrification rate on each day was calculated as $k_w * N_{conc}$, with the value of k_w derived from the ratio of the initial water denitrification rate (427 μg N L^{-1} day^{-1}) and initial water N_{conc} (4000 μg L^{-1}) as 0.107 day^{-1}.

Denitrification can also occur in the water 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. For the Babinda wetland we have used the range of soil water denitrification values found in four tropical wetland studies in north Queensland of 33 to 74 (average 48) mg N m^{-2} day^{-1} (see Table 2 in Wallace et al., 2020). These daily rates were converted to the total nitrogen lost from the water column using the wetland area on each day. As with the water column, the soil water denitrification rate was calculated as $k_s * N_{conc}$ with the value of k_s derived from the ratio of the initial soil denitrification rate (48 mg N m^{-2} day^{-1}) and the initial water N_{conc} (4000 μg L^{-1}) as 0.012 day^{-1}.

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) summarized 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 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). DO is usually lower towards the bottom of the water column, so in our estimate of denitrification from the water within the soil we have assumed the average amount of time DO was below 2 mg L^{-1} was 67%. This figure was calculated from 3 months of hourly DO data recorded 10 cm above the bottom of a wetland system near Mackay, north Queensland (Waltham and Butler, 2020).

3. Results

3.1. Water balance

Water first entered the wetland on 19 October 2017, following heavy rainfall (300 mm) on this and the previous day (Fig. 3). The daily average water depth at site 2 increased rapidly, reaching a maximum of 140 cm, well above the top of the berm wall at ~60 cm.

Once the heavy rainfall on 19 October 2017 ceased, the water level 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 depth was ~20 cm. Rain on 9 and 10 November 2017 (80 mm) caused a small rise in water level (~5 cm), after which it stayed around 25 cm due to a further series of small rainfall inputs in November 2017 (Fig. 3b).

After draining the wetland on 23 November 2017 (point B Fig. 3), water depth was below 5 cm (the lowest level the sensors can record) throughout December and early January 2018. The water level increased rapidly after 343 mm of rain on 16–17 January 2018 (Fig. 3b), reaching ~2 m before falling rapidly again. There were a further 4 occasions after this when water depth rose to over 2 m and during the last three events it stayed fairly constant for up to 7 days. During these periods the entire area around the Babinda wetland was inundated to a depth in excess of the berm wall height. These events were characterised 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.

3.2. Wetland denitrification

Our estimates of nitrogen loss from the Babinda wetland are given below. The first estimate of nitrogen losses are for conditions in which the wetland depth decreases in time by allowing for losses due to evaporation and drainage. We have allowed for denitrification from both the water column and water within the soil beneath it. 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.

3.2.1. Denitrification after a flood event

The first denitrification estimate is for a simulated single flood event, which completely fills the wetland, following which there is no further rainfall or run-in. Subsequent depth decreases, due to losses from evaporation and drainage, 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 its relationship with depth as $DR_w = 0.034\ d$ (Wallace et al., 2020). With these evaporation and drainage rates, it takes ~60 days for the wetland to empty, Fig. 4.

In this simulation the average initial rates of denitrification from the

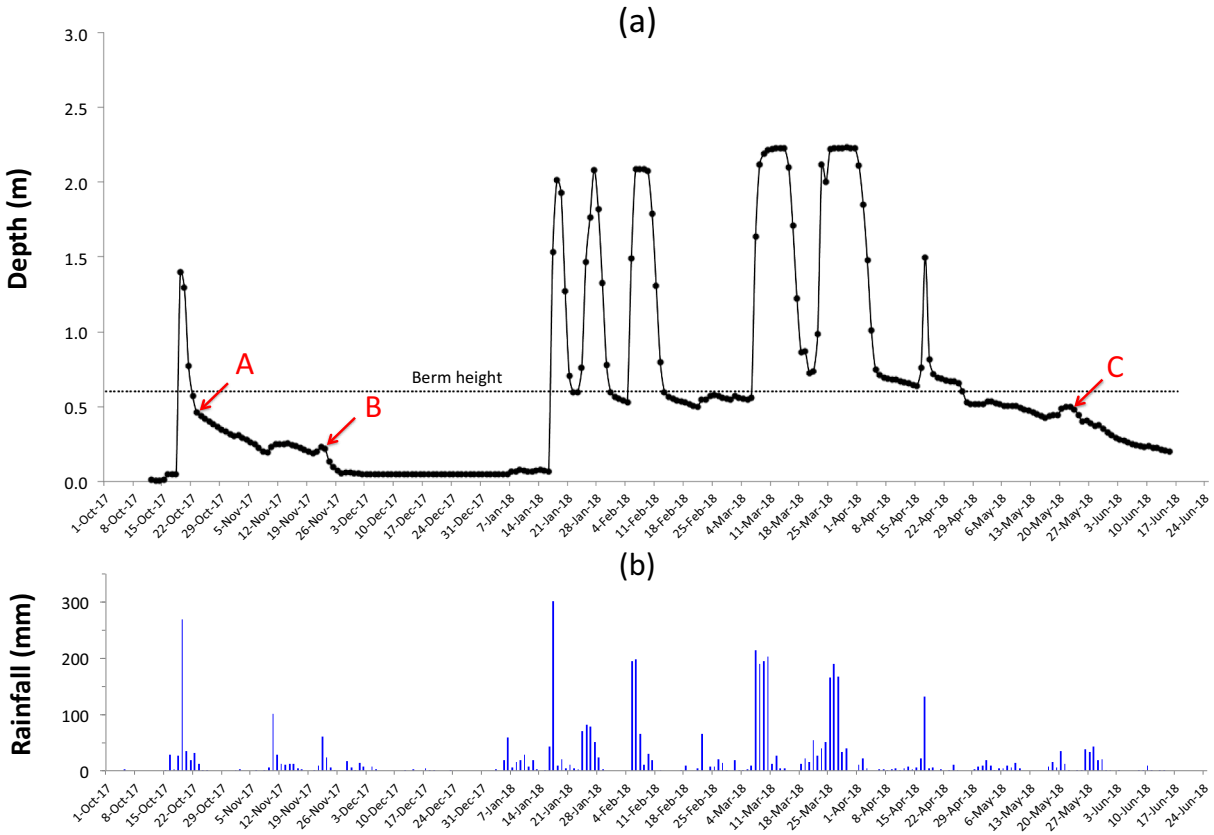


Fig. 3. (a) Changes in daily average depth at wetland site 2 between October 2017 and June 2018 and (b) Daily rainfall during the same period. The height of the berm wall spillway is shown by the dotted line. Sluice gates were operated on three occasions; (A) 23 October 2017 – inlet gate closed, (B) 23 November 2017 – outlet gates opened, (C) 23 May 2018 – outlet gates opened.

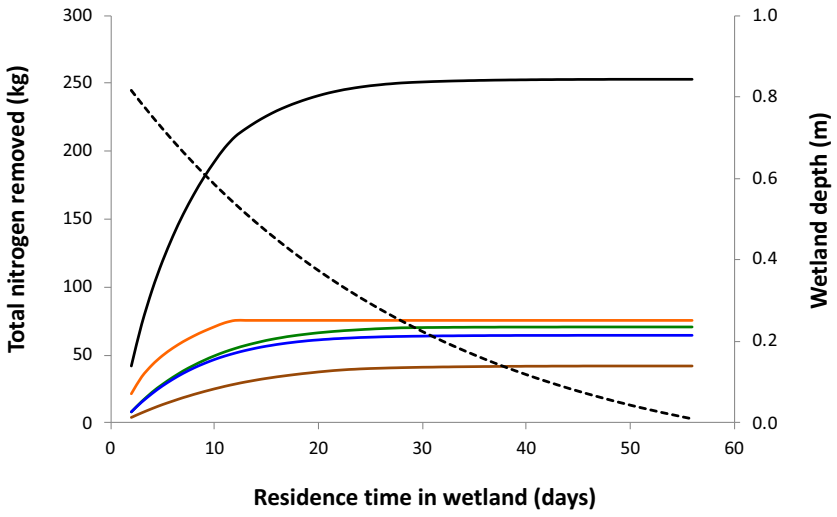


Fig. 4. The total amount of nitrogen (kg) removed from the wetland (solid black) as a function of the length of time the water is in the wetland (residence time). Losses due to PN sedimentation (orange), water column denitrification (blue), soil water denitrification (brown) and loss of drainage water (green) are also shown. The change in water depth is also shown (black). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

water column (427 μg N L^{-1} day^{-1}) and water in the soil (48 mg N m^{-2} day^{-1}) were used and this led to a steady increase in the amount of nitrogen removed from the wetland over the following 60 days. The largest individual loss of nitrogen, 30% (76 kg), occurred as PN settled to the bottom of the water column in the first 12 days. The next largest loss of nitrogen was due to drainage, which accounted for 28% (70 kg) of the total nitrogen loss from the wetland. Gaseous denitrification from the water column accounted for 26% (65 kg) of the total nitrogen loss, with denitrification from the soil water at 17% (42 kg). Combined gaseous denitrification from the water and soil was 42% (107 kg) constituting the largest nitrogen loss from the wetland.

The above denitrification estimates are very dependent on the rate of denitrification in the water column. For example, at the lower initial rate of denitrification (0.34 μg N L^{-1} day^{-1}), gaseous denitrification is almost entirely from the water in the soil at 32% (72 kg) and drainage losses increase to 34% (76 kg). Conversely, at the higher initial rate of water column denitrification (768 μg N L^{-1} day^{-1}), total denitrification increases to 48% (122 kg) and drainage losses decrease to 32% (55 kg). There are no changes in the losses due to sedimentation.

3.2.2. *Water nitrogen concentration*

The way in which the concentration of nitrogen in the wetland water column changes with time is shown in Fig. 5. Again, in this simulation the average initial denitrification rates in the water column (427 μg N L^{-1} day^{-1}) and soil water (48 mg N m^{-2} day^{-1}) were used. The initial concentration of total nitrogen (DN + PN) is assumed to be 4000 μg L^{-1} and this decreases quite rapidly in the first few days as the PN fraction precipitates from the water column and denitrification occurs. After 12 days there is no PN in the water and further decreases in nitrogen concntration are due to denitrification only.

The way in which the concentration of nitrogen varies with residence time is very dependent on the water column and soil water denitrification rates. This is illustrated in Fig. 6, which shows the predicted time course of nitrogen concentration at three initial denitrification rates, (1) Low; water column 0.34 μg N L^{-1} day^{-1} and soil water 33 mg N m^{-2} day^{-1} (2) Average; water column 427 μg N L^{-1} day^{-1} and soil water 48 mg N m^{-2} day^{-1} and (3) High; water column 768 μg N L^{-1} day^{-1} and soil water 73 mg N m^{-2} day^{-1}. As expected, nitrogen concentration is lowest at the highest denitrification rates and approaches zero after ~30 days. At the average denitrification rates nitrogen concentration decreases more slowly, taking ~50 days to approach zero. Simulations at the lowest denitrification rates show the slowest drop in nitrogen concentration which stays above 800 μg L^{-1} . Interestingly, there is a small rise in concentration after day 26 and day 50. This occurs because volume decreases at these times that concentrate the nitrogen are greater than the slower loss of nitrogen due to the very low denitrification rate. Most of this volume decrease is due to evaporation (>75%) and this illustrates the potential for nitrogen to become more concentrated in the small pools that form as the wetland empties.

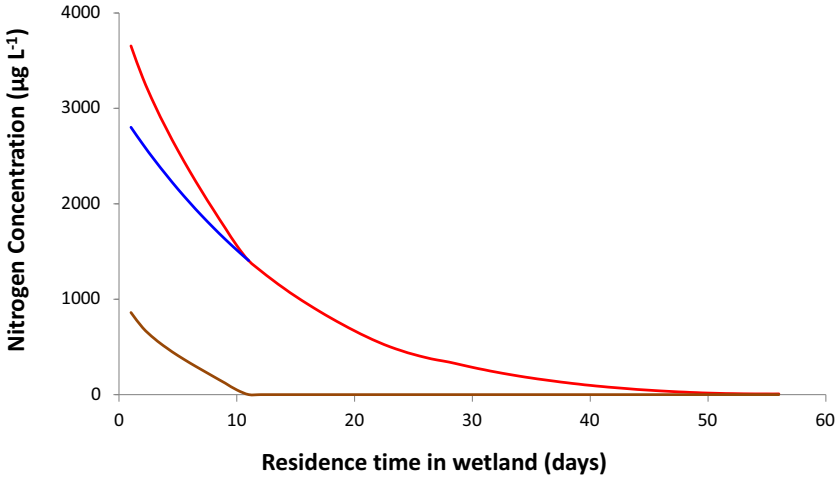


Fig. 5. The change in total nitrogen concentration in the wetland (red) as the wetland volume (black) decreases due to evaporation and drainage. Also shown are the dissolved nitrogen concentration (blue) and particulate nitrogen concentration (brown). In this simulation the average initial rates of denitrification in the water column (427 μg N L^{-1} day^{-1}) and soil water (48 mg N m^{-2} day^{-1}) were used. (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

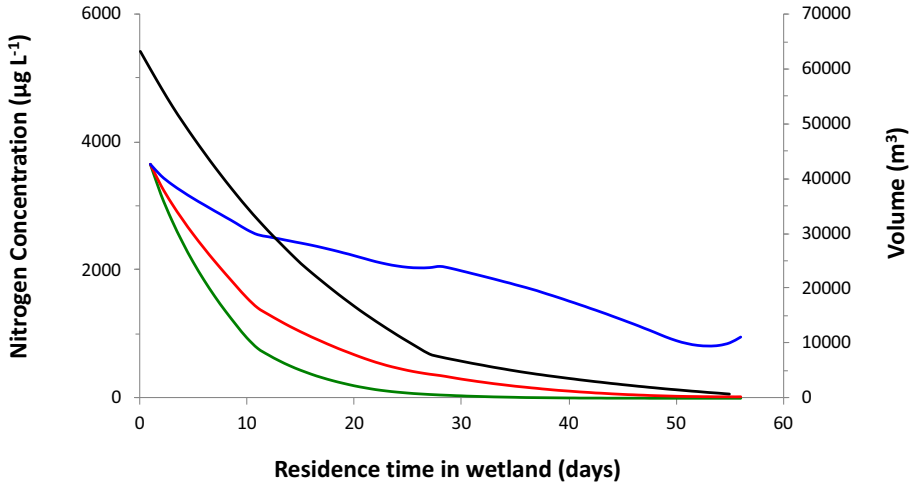


Fig. 6. The change in nitrogen concentration in the wetland with time at the low water and soil denitrification rates (blue); average denitrification rates (red) and high denitrification rates (green). The decrease in wetland volume due to evaporation and drainage is also shown (black). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

3.2.3. Denitrification from water within the soil

Estimates of the cumulative denitrification from water within the wetland soil over time are shown in Fig. 7. This simulation is also for a single flood pulse that fills the wetland and uses the average initial water column denitrification rate of 427 µg N L^{-1} day^{-1} in all three simulations. At the lowest initial soil water denitrification rate (33 mg N m^{-2} day^{-1}), 32 kg of nitrogen is lost from the anaerobic wetland soil over the whole simulation period, with 90% of the loss occurring in the first 25 days. Total nitrogen loss from the soil water is dependent on the anaerobic soil area, which is assumed to be that which is covered by water. As the wetland soil surface slopes (downwards) irregularly from west to east, the area of the wetland decreases non-linearly as the wetland depth decreases over time, Fig. 7. Total denitrification from water within the soil increased to 42 kg when the average initial soil water denitrification rate was used (48 mg N m^{-2} day^{-1}) and reached 55 kg at the high initial soil water denitrification rate (73 mg N m^{-2} day^{-1}).

3.2.4. Annual nitrogen filtering of the wetland

Our final estimate of wetland nitrogen losses has been made over the full year October 2017 to September 2018. Measured depth data were used for the monitoring period between October 2017 and 18 June 2018. For the period 18 June to 30 September 2018 the wetland water balance model was used to calculate depth (Fig. 8a). In this simulation we have used these daily depths to calculate the wetland volume and area (see Wallace et al., 2020), from which the daily water column and soil water denitrification were both estimated. We used the average water column value of 427 µg N L^{-1} day^{-1} and the soil water rate of 48 mg N m^{-2} day^{-1}, both of which were reduced as the concentration of DN decreased after each flood pulse. Drainage from the wetland was also calculated from the changes in depth, allowing for losses due to evaporation and any inputs from rainfall and run-in.

Fig. 8 shows that over this year there were 8 flood pulses that raised the water level in the wetland above the berm wall. On each of these occasions the main loss of nitrogen (664 kg) was via drainage from the

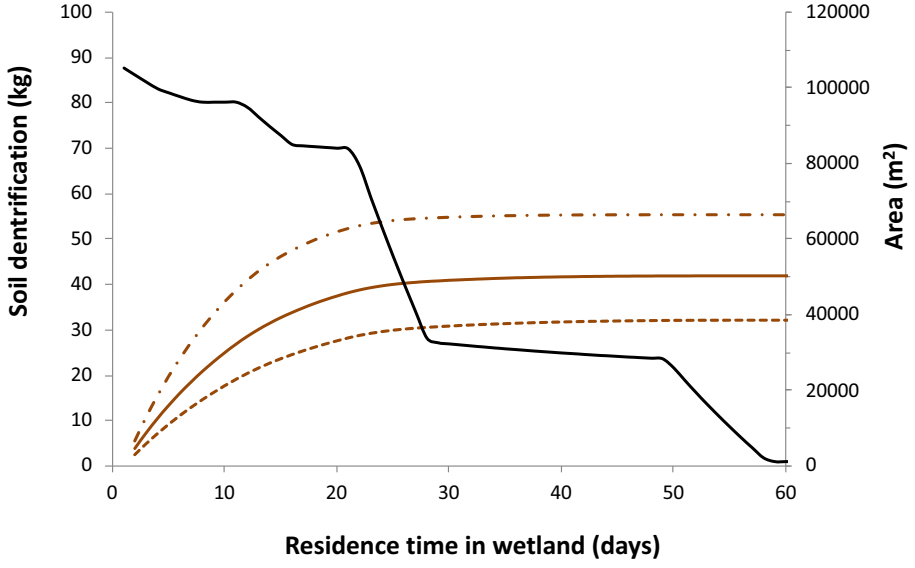


Fig. 7. The increase in denitrification from water within the soil (brown) with residence time in the wetland. Calculations are shown for three soil water denitrification rates; low, 33 mg N m^{-2} day^{-1} (brown dashed); average, 48 mg N m^{-2} day^{-1} (brown solid) and high, 73 mg N m^{-2} day^{-1} (brown dash-dot). Also shown is the decline in wetland area over time (black). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

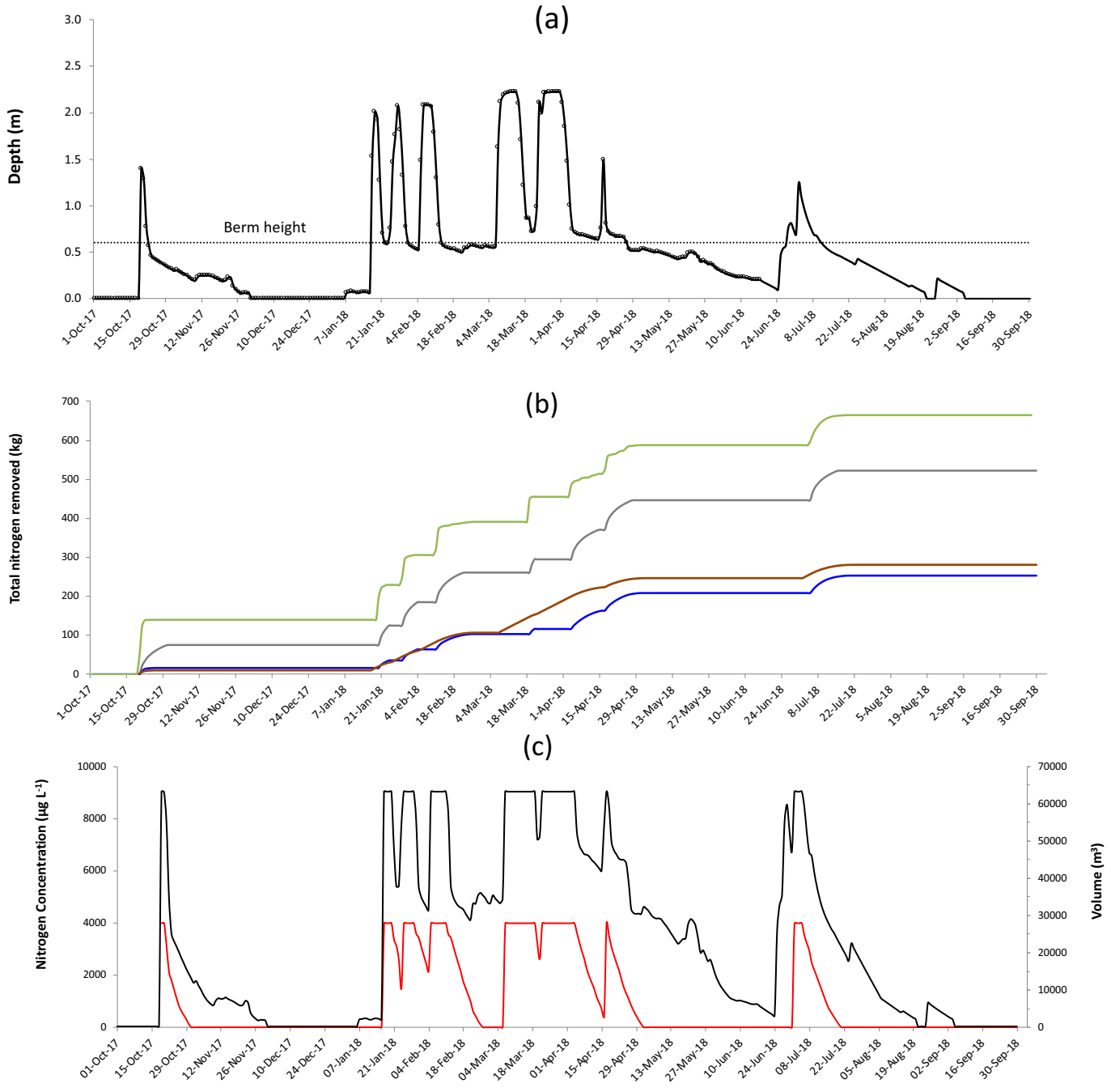


Fig. 8. Annual time series of (a) depth, (b) cumulative nitrogen losses by drainage (green), water column denitrification (blue), soil water denitrification (brown) and PN loss (grey) and (c) wetland water nitrogen concentration (red) and volume (black). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

wetland (Fig. 8b). The next largest loss of nitrogen (523 kg) from the wetland water was via the settlement of PN, which occurred within the first 12 days after each flood pulse. Denitrification in the soil water (280 kg) was less than half of that lost by drainage and continued as long as there was water containing nitrogen in the wetland. Denitrification in the water column (252 kg) is less than in the soil water, mainly because it is insignificant when deep floodwater is flowing rapidly through the wetland. However, at these times the model allows denitrification to continue in the soil water, which is more isolated from the overlying floodwater. Gaseous denitrification losses from the wetland are proportional to the volume and area of water in the wetland, which were initially low, but increased markedly during the January to April 2018 wet season.

Fig. 8c shows that with each flood pulse the concentration of nitrogen in the wetland rises to the concentration assumed for floodwater (4000 µg L^{-1}) and stays at this concentration while the depth is greater than the berm wall. This is because during this period water enters and leaves the wetland very rapidly and there is no time for any significant PN settlement or water column denitrification. Once the depth falls below the berm wall, sedimentation and denitrification in the water column begin and the nitrogen concentration decreases fairly rapidly, reaching zero when there was more than ~2 weeks between flood pulses.

Table 1 summarises the nitrogen balance of the wetland over the

Table 1
Summary of wetland annual nitrogen balance.

	kg ha^{-1}	% of input
Nitrogen input	192	
Water column particulate settlement loss	−49.7	25.9
Water column denitrification loss	−24.0	12.5
Water column total nitrogen loss	−74	38.3
Soil water denitrification loss	−26.6	13.9
Drainage loss	−63	32.8
Flood flushing loss	−29	15.0
Total nitrogen loss in drainage & flushing	−92	47.8
Total filtering of nitrogen by wetland	100	52.2

entire year. The eight flood pulses brought a total of 2024 kg of nitrogen into the wetland, equivalent to 192 kg N ha^{-1}. Much of this nitrogen left the wetland as drainage (33%), with additional amounts (15%) leaving as some of the flood pulses flushed the wetland of PN and DN before it was empty.

Total soil and water denitrification were similar and gave a combined nitrogen loss of 51 kg N ha^{-1}, (26% of the nitrogen input), with an additional loss due to the settlement of PN of 50 kg N ha^{-1} (26% of the nitrogen input). The wetland was therefore able to filter 52% of the nitrogen that entered it. If the lower water denitrification rate (0.34 μg N L^{-1} day^{-1}) derived from the study of a riverine wetland by McJannet et al. (2012) is used the wetland filtering drops to 45%. In contrast, if the higher water denitrification rate measured in the Babinda wetland (768 μg N L^{-1} day^{-1}) is used, then the nitrogen filtering increases to 56%.

4. Discussion

4.1. Water balance

The data recorded in the Babinda wetland show that large rainfall events are sufficient to raise the water depth in the constructed wetland to over 2 m. This is well above the level of the berm wall access roads (0.6 m) and so water initially drains very rapidly (~100–700 mm day^{-1}) from the wetland (Wallace et al., 2020). This rapid drainage can last for up to 7 days until the water level drops below the height of the berm spillway. During this time deep water is flowing very rapidly across the wetland, and so there is little/no opportunity for it to filter any sediment or nitrogen.

Once the water level is below the berm wall height drainage is slower, but continues between 3 and 20 mm day^{-1}. In our study 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 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 et al. (2007) reported values of the recharge rates from constructed recharge pits on the Burdekin delta of 1.4 to 3.4 mm day^{-1}. As drainage is a large component of the wetland water balance (over 80% of the water loss), it is important to take this into account when estimating the filtering capacity of a wetland.

4.2. Wetland denitrification

The values of water denitrification found in Australian tropical wetlands vary widely, ranging from 0.34 (McJannet et al., 2012) to 768 μg N L^{-1} day^{-1} (Adame et al., 2019a, 2019b; Wallace et al., 2020) for the water column and 33 (McJannet et al., 2012) to 73 mg N m^{-2} day^{-1} (Adame et al., 2019a, 2019b; Wallace et al., 2020) for the water within the soil. Some of the difference in the above studies are due to the very different methods used; i.e. actual monthly and annual rates from a water and nutrient balance taking into account periods of low dissolved oxygen (McJannet et al., 2012) compared to potential denitrification rates over periods ~6 h measured using isotopic tracing (Adame et al., 2019a, 2019b). The much lower denitrification rates in the former study also reflect the short residence times of water in their riverine wetland lagoon.

Using the above range of water and soil denitrification rates gives annual denitrification totals in the Babinda wetland that vary from 27 to 67 kg N ha^{-1} year^{-1}. These annual amounts are higher than those found in constructed riparian wetlands in Ohio, USA, where annual rates of between 19 and 44 kg N ha^{-1} year^{-1} were found using the acetylene inhibition technique (Song et al., 2014). The same technique gave higher rates of 68 kg N ha^{-1} year^{-1} in a restored riparian forest wetland (Lowrance et al., 1995). However, Groffman and Hanson (1997) have pointed out that many factors can influence the rate of denitrification in wetlands (e.g. site quality, hydrology and vegetation, method used), which accounts for the wide range of values they reported between 1 and 130 kg N ha^{-1} year^{-1}. In general, the Babinda wetland denitrification is towards the upper end of these other reported values.

The estimation of denitrification is also affected by the initial concentration of nitrogen in the water entering the wetland. In the absence of measurements in this study we have assumed a fixed value of 4000 μg L^{-1}. Changing in this initial concentration to 2000 μg L^{-1} reduces the total denitrification from 51 to 38 kg N ha^{-1} year^{-1} but filtering increases to 66%. Conversley, if the initial concentration was 6000 μg L^{-1}, the total denitrification increases to 58 kg N ha^{-1} year^{-1} and filtering decreases to 46%.

The estimation of annual nitrogen removal by wetlands by extrapolating a limited number of measured hourly rates can be very misleading. For example, the mean total denitrification hourly rate in the Babinda wetland was measured as 8.1 mg N m^{-2} h^{-1} (see Wallace et al., 2020). This can be scaled up using the wetland area of 10.54 ha to an annual denitrification rate of 710 kg N ha^{-1} year^{-1}. However, this is nearly seven times the annual rate derived using the same hourly rate (split as water column 6.4 mg N m^{-2} h^{-1} and soil water 1.7 mg N m^{-2} h^{-1}) in our water balance and denitrification model, i.e. 104 kg N ha^{-1} year^{-1}. 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. Furthermore, denitrification in the year we studied could not take place for 68% of the time as there was no water in the wetland and when there was, a further 14% of time the nitrogen concentration was zero.

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 33% of the annual loss of nitrogen from the wetland, and with an additional 15% loss due to flushing during floods, means that the filtering capacity of the wetland was 52%. Other studies of constructed wetlands have also found drainage (as groundwater losses) to be a major nitrogen loss mechanism (e.g. Batson et al., 2012). Nitrogen dissolved in drainage water is not part of the wetland filtering, as it will remain in solution outside the wetland unless conditions become suitable for further gaseous denitrification off site (e.g. in surrounding ditches, streams and aquifer). If the drainage mainly appears in nearby ditches and streams, 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 ~25 days to lose over 90% of the nitrogen in the drainage water, by which time it may well have travelled to the main river and even into the nearby ocean.

All of the suspended sediments entering the wetland are retained provided there is more than 12 days between flood pulses. However, in the year studied several flood pulses occurred with less than this time between them and the residual sediment was flushed out. Over the year simulated this amounted to 14% of the total sediment load, hence only 86% was retained in the wetland. As PN sedimentation adds to the nitrogen stored in the soil, the net change in soil nitrogen is given by the PN deposition less that removed by flushing (8 kg N ha^{-1}) and soil denitrification. This amounts to 15 kg N ha^{-1} over the year studied (see Table 1).

5. Conclusions

This study has demonstrated the value of combining a hydrological water balance model with a simple first order rate based denitrification model to estimate the long term nitrogen and sediment filtering capacity of wetlands. From a time series of wetland depth measurements it is possible to derive a water balance for the wetland, which quantifies the key parameters that are needed to estimate denitrification over weeks, months and years. We have also shown the vital importance of knowledge of the wetland hydrology in estimating denitrification as this has a major effect on the presence and residence time of water in the wetland that is highly variable during the year.

The current model estimates of denitrification and filtering in the Babinda wetland vary widely as they are based on the wide range of denitrification rates found in tropical wetlands in Australia. The higher values are found in denitrification 'hot spots' where wetland conditions at certain times of the year can support rapid conversion of dissolved nitrogen into nitrogen gas. In contrast, there are also wetland denitrification 'cold spots', where hydrological conditions for most of the year do not allow much denitrification. In any tropical catchment floodplain there is likely to be a range of wetland types, some hot spots and some cold spots. The best way to identify which wetlands have significant filtering capacity is to make periodic measurement of dissolved nitrogen (as DIN, DON and ammonium) over time to test and calibrate the water balance/denitrification model. It is especially important to measure the concentration of nitrogen in the water entering the wetland as this has a major effect on the wetland filtering. When this is done, estimation of wetland denitrification over long-term current or future periods should be much more reliable.

The duration of low DO also needs to be monitored in the wetland, as this is a critical parameter in determining gaseous denitrification. This should be done throughout the water column and at hourly intervals so as to capture the variation in DO with depth and diurnally. These data can then be used to update the daily amount of time DO (within the water column and near the soil surface) is below the critical threshold for denitrification to occur. It would also be useful to determine the relationship between DO concentration and denitrification rate in the tropical waters present in the north Queensland wetlands.

Sedimentation of particulate nitrogen is a significant part of the filtering effect of the wetland; so to improve our current estimates PN concentration should be measured along with the rate at which it declines after the wetland is filled with floodwater. Sedimentation adds to the nitrogen load in the wetland soil, where denitrification can occur at rates comparable with those in the wetland water column. This soil water denitrification can reduce the amount of nitrogen stored in the soil, and can reduce the concentration of nitrogen in the drainage water which constitutes a major loss mechanism in the wetland. In this study drainage is thought to mainly occur as comparatively rapid seepage through the berm walls rather than deeper groundwater movement. However, 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.

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 a reduction in DIN of 300 t and PN of 53 t by 2025 is recommended. 10% of the DIN reduction target could be achieved from 593 ha of wetland with the same mean denitrification properties as the Babinda wetland; which amounts to 2.3% of the total sugarcane area in the Mulgrave-Russell catchment. This area of wetland would also remove 56% 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 525 ha, or 2.0% of the sugarcane area. There may also be additional nitrogen removal benefits from pumping water into the wetland during the dry season.

CRediT authorship contribution statement

Jim Wallace: Project Conceptualisation, data compilation and curation, formal analysis, investigation, methodology, project administration, original draft, writing, review and editing. Nathan Waltham: Project Conceptualisation, Funding acquisition, project administration, writing, original draft, writing, review and editing.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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