Denitrification bioreactor trial in the Russell River catchment of the Wet Tropics:

Final Report

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Executive Summary

Dissolved inorganic nitrogen (DIN) in runoff from agricultural land is considered to have a significant detrimental impact on the Great Barrier Reef (GBR). Losses of DIN to runoff can be reduced by good agricultural practices, but they cannot be eliminated entirely in the Wet Tropics due to the need for adequate nitrogen supply to crops, the high solubility of DIN, particularly nitrate, and high rainfall. Thus, it is inevitable that DIN concentrations are higher in runoff from agricultural land than from forested areas. Some of this DIN is removed from the water as it moves through aquifers, creeks, rivers, and wetlands on its way to the sea, through the process of microbial denitrification. Denitrification involves the conversion of nitrate and nitrite (NO_x-N) to dinitrogen (N_2) gas, which is lost to the atmosphere.

Denitrification requires NO_x-N, organic matter, and low oxygen concentration. Wetlands provide these conditions, so DIN concentrations decline in water moving through them. Similarly, denitrifying bioreactors are designed to treat water by passing it through a porous organic material, typically woodchips. The woodchips provide organic matter for the microorganisms, which in turn lower the oxygen concentration, providing ideal conditions for denitrification.

Denitrifying bioreactors are now widely used to remove the NO_x-N component of DIN from agricultural runoff water elsewhere, but they have not yet been evaluated in the Wet Tropics. The Wet Tropics pose a challenge for efficacy due to the large volumes of water moving through the landscape. The objective of this project was “to establish the effectiveness of denitrifying bioreactors as a remediation technology for excess DIN in agricultural runoff within the Babinda Swamp Drainage Area (BSDA) of the Russell catchment”. The Russell River exports a disproportionate amount of DIN to the GBR lagoon because of the high rainfall and high proportion of agriculture, mostly sugarcane, in its catchment.

To establish the context in which the bioreactors were to operate, the amounts of nitrogen entering and leaving the BSDA in surface water were determined over the course of a year (1/11/2018 to 31/10/2019). Discharge and nitrogen concentrations were measured in the main inflow to the BSDA, Niringa Creek, and the outflow at Christiano Access road, and loads were calculated by multiplying the discharge by concentration. Over the 2018-2019 year the total dissolved nitrogen (TDN) load entering the BSDA via Niringa Creek was 3,604 kg yr^{-1}, including 2,650 kg yr^{-1} NO_x-N. The TDN load leaving the BSDA in the main drain was 31,505 kg yr^{-1}, including 20,714 kg yr^{-1} NO_x-N. This equates to a TDN export from the BSDA of 8.3 kg ha^{-1} yr^{-1} in drainage water.

To assess bioreactor efficacy, four bioreactors were installed on sugarcane farms in or near to the BSDA and evaluated over the 2018-2019 and 2019-2020 wet seasons. Two were ‘Inline Beds’ in which woodchips were placed in a bed below the downslope end of a farm drain, in a bed approximately 20 m long, 1 m wide and 0.8 m deep, covered with 0.2 m of soil. Water flowing down the drain entered the bed through a gravelled inlet structure, flowed through it, and left via a gravelled outlet structure and pipe. The third was an ‘Offline

Bed', which had similar design except that it was adjacent to a drain rather than underneath it. The fourth was a 'Wall' designed to intercept shallow groundwater. It was installed along the downslope edge of a paddock, and was 48.1 m long, 0.65 m wide and 1.09 m deep, covered with 0.5 m of soil. Concentrations of nitrogen in water entering and leaving the bioreactors was determined using grab sampling and automatic sampling.

The bed bioreactors removed 41 % of the nitrogen that entered them, on average. The mean concentrations of NO_x-N were 0.20 mg L^{-1} in the inlets and 0.10 mg L^{-1} in the outlets. During periods of high NO_x-N concentrations, which coincided with periods of high flow at the start of the wet season, the reduction in concentration by the bioreactor beds was limited by hydraulic residence time. A useful rule of thumb arising from this work is that NO_x-N concentration is reduced by approximately 1 mg L^{-1} for every hour of residence time within a woodchip bioreactor. A total water and nitrogen budget of Inline Bed 1 showed that 7 % of the nitrogen load in the drain entered the bioreactor and that the NO_x-N load from the contributing paddocks was reduced by 0.11 kg ha^{-1} yr^{-1} in the 2018-2019 year. Most of the nitrogen load was exported during a 'first-flush' period of approximately 10 days in the 2018-2019 season. However, very little nitrogen was exported at the beginning of the 2019-2020 wet season, presumably because sufficient rain to cause runoff occurred much later after fertiliser application than in the 2018-2019 wet season.

The maximum daily denitrification rate, per unit bioreactor volume, as measured in the inline bioreactor beds was 8.7 g-N m^{-3} s^{-1}, close to the maximum rates observed in parallel laboratory studies. However, for much of the monitoring period, the bioreactors as N-limited, resulting in average denitrification rates of just 0.1 g-N m^{-3} day^{-1}.

The wall bioreactor reduced the concentration of TDN in shallow groundwater from a mean of 0.81 mg L^{-1} (including 0.59 mg L^{-1} NO_x-N) to a mean of 0.24 mg L^{-1} (including 0.01 mg L^{-1} NO_x-N). Wall bioreactors effectively treat water flowing through them in all likely scenarios of hydraulic conductivity and head, but significant amounts of groundwater are treated only if the soil has high hydraulic conductivity (e.g. sandy texture or high macroporosity). Electromagnetic induction surveys can be used to identify these areas of preferential flow, e.g. paleochannels in the landscape.

The cost-effectiveness of bioreactors was poor for beds but reasonable for walls. Using the costs of construction in this trial, the cost of nitrogen removal by paddock-scale bioreactor beds was estimated at \$1,409 kg^{-1} nitrogen. However, the modelled potential cost-effectiveness of bioreactor walls was found to be reasonably high (e.g. <\$100 kg^{-1}, not counting costs of site surveys or project management) under conditions of high soil hydraulic conductivity, or in areas where preferential flow paths allowed the targeting of walls of limited length to capture shallow lateral nitrogen losses.

The main unknowns for calculating cost-effectiveness, applying to beds and walls, were the lifespan of the bioreactors (assumed to be 10 years) and the change in interception and efficacy over time (assumed to be zero). For the bioreactor wall, additional important unknowns were variability in hydraulic conductivity in the contributing area (assumed to be

uniform), and the dimensions of high conductivity zones and their contributing areas, which determine the amount of nitrogen passing through the wall.

The likely effect of installing paddock-scale bed bioreactors throughout the BSDA was assessed using a drainage model. The drainage network was mapped and 127 suitable locations were identified, with a total contributing area of 733 ha, being 19 % of the area contributing to the main BSDA drain. Installing bioreactors at all 127 sites would result in an estimated nitrogen load reduction of 81 kg yr^{-1}. Exploratory modelling suggests that substituting one large bioreactor bed for multiple paddock-scale beds would increase nitrogen removal and cost-effectiveness, but the assumptions need to be tested before such an option was considered.

To identify factors that might influence uptake of bioreactors by sugarcane farmers, eight farmers in the Russell River catchment were interviewed. The farmers regarded bioreactors positively, should the trial prove their effectiveness within the climatic and soil constraints of the area. The best setting for uptake involves: a) Conclusive evidence that the Russell River catchment has a nitrogen pollution problem related to runoff from individual farms, b) Promotion of environmental benefits, c) Guidelines for design and installation, and d) Financial assistance for implementation.

We conclude that, while the bioreactors assessed in this project significantly reduced the nitrate concentrations of water moving through them, paddock-scale beds had little impact on nitrogen loads passing downstream, due to the hydrology of the Wet Tropics and issues surrounding 'first flush' from sugarcane paddocks . The use of large-scale bioreactor beds lower in the catchment may be more effective because water flow and nitrate concentrations are maintained for a higher proportion of the year. They can be considered akin to augmented or enhanced landscape wetlands. It would be worthwhile investigating them further, particularly the hydraulic constraints that may limit their design and deployment. We also conclude that the relatively low construction costs of bioreactor walls, and the fact that even thin walls are capable of treating shallow groundwater under most hydraulic conditions mean that they should be considered for large-scale deployment in soils with high hydraulic conductivity or in a targeted fashion in areas of preferential shallow groundwater flow.

Building on the results of this project, it became clear that further research is needed to:

- Determine the change in bioreactor effectiveness over time, including those installed in this project, which are equipped for the purpose and have now been in place for 2 years. This would have two purposes: first, to determine their lifespan and second, to determine their effectiveness under variable prevailing climatic and management conditions.
- Continue water quality and flow monitoring across the BSDA to allow for determination of the nitrogen budget across variable climatic conditions. This would help reconcile the significant difference found between this study and Queensland DNRME (Department of Natural Resources, Mines and Energy)-modelled DIN loads per unit area of sugarcane.

- Evaluate the use of high resolution soil mapping (e.g. using electromagnetic induction) to help determine suitable locations for bioreactor walls, in addition to informing better in-field nutrient management.
- Evaluate 'controlled drainage' as a way of delaying the first flush of runoff to enhance loss of DIN via denitrification in paddock soils, drainage systems and inline bioreactor beds.
- Determine the effectiveness of the novel bioreactor configurations and designs conceived during this project. For example, 'hybrid walls' and the use of woodchips in interceptor drains should be evaluated. Such novel designs may provide the compromise needed between concentrating NO_x-N flows (an advantage of beds) and higher hydraulic residence time (an advantage of walls) required to achieve substantial reductions in nitrogen load. Initial evaluations have commenced in collaboration with Terrain NRM, but further replicated studies are required to ascertain their true efficacy in nitrogen removal.

Project Overview

The project sought to establish the effectiveness of denitrification 'bioreactors' as an on-farm technology for removing dissolved inorganic nitrogen (DIN) and specifically nitrate and nitrite (NO_x-N, Figure 1) from agricultural runoff under the conditions of the Australian Wet Tropics. The trial was conducted in the Babinda Swamp Drainage Area (BSDA), an area of predominantly sugarcane farms in the Russell River catchment, where it was also possible to estimate total nitrogen load and level of interception from a broader roll-out of bioreactors across the wider catchment.

The trial was funded through the Queensland Government's Reef Water Quality Program – Great Barrier Reef Innovation Fund. Being one of the first trials of bioreactors in the Wet Tropics, the bioreactors form part of a broader integrated treatment system to improve water quality leaving the BSDA. A corresponding trial was established in the Dry Tropics and administered by Queensland University of Technology and the Queensland Department of Agriculture and Fisheries (DAF).

The trial also contributes to establishment of the 'Bioreactors for the Great Barrier Reef' (B4GBR) network established by DAF. This network facilitates the exchange of data and ideas leading to the production of State guidelines on the design and use of denitrifying bioreactors as a possible on-farm strategy for improving runoff water quality.

Background

The Russell River catchment is part of the Russell-Mulgrave basin. To improve the health and resilience of the Great Barrier Reef (GBR), the Russell-Mulgrave basin has an end-of-catchment anthropogenic DIN reduction target of 70% by 2025[1]. The high target reflects the significant DIN load the basin discharges annually to the GBR lagoon. In particular, the Russell River catchment is a DIN hotspot, accounting for 11% of DIN export to the GBR lagoon[2]. Modelled DIN loads for agricultural areas within the Russell River catchment range between 13.93 kg ha^{-1} yr^{-1} and 20.21 kg ha^{-1} yr^{-1}. [3] This, combined with a high proportion of agricultural land in the catchment leads to the Russell catchment producing the highest DIN yields per unit area across the GBR catchment at 590 kg km^{-2}.[4]

[1] State of Queensland, 2018. Reef 2050 Water Quality Improvement Plan, 2017-2022, Table 2. End-of-catchment anthropogenic water quality targets for the Reef catchments by 2025 and relative priorities for water quality improvement, p 18. See: https://www.reefplan.qld.gov.au/

[2] The State of Queensland (Department of Environment and Science), 2017. Total suspended solids, nutrient and pesticide loads (2015–2016) for rivers that discharge to the Great Barrier Reef, Great Barrier Reef Catchment Loads Monitoring Program, p. 34.

[3] Source: Terrain NRM, https://terrainnrm.maps.arcgis.com/apps/MapSeries/index.html?appid=85b6f348f55643e78240ce1a16c91062 Accessed: 9/05/2020.

[4] The State of Queensland (Department of Environment and Science), 2017. Total suspended solids, nutrient and pesticide loads (2015–2016) for rivers that discharge to the Great Barrier Reef, Table 3.3 Total suspended solids and nitrogen yields calculated for the 2015-2016 monitoring year, p. 44.

With the support of the Babinda Swamp Drainage Board, Jaragun NRM is establishing a treatment system to reduce DIN loads from the BSDA. The focus to date has been managing surface runoff through several Commonwealth and Queensland Government initiatives, comprising a constructed wetland, sediment traps and revegetation programs. However, given the extent of land under sugarcane and hydrological characteristics (e.g. prone to flooding) of the BSDA, Jaragun NRM and the Babinda Swamp Drainage Board are committed to trialling additional strategies, such as denitrifying bioreactors to establish their contribution to achievement of the 70% DIN reduction target.

Denitrification bioreactors seek to facilitate and accelerate the natural microbial-mediated conversion of nitrate and nitrite (NO_x-N; a component of DIN (Figure 1) to benign N_2 gas in the atmosphere by passing water containing NO_x-N through a carbon substrate under anaerobic conditions (Figure 2). In the absence of O_2, native microbes use NO_x-N as a terminal electron acceptor for ATP production as they decompose the carbon substrate. This same process occurs naturally in wet soils and wetlands; bioreactors are used to accelerate this process to remove excess nitrogen found in agricultural effluent water.

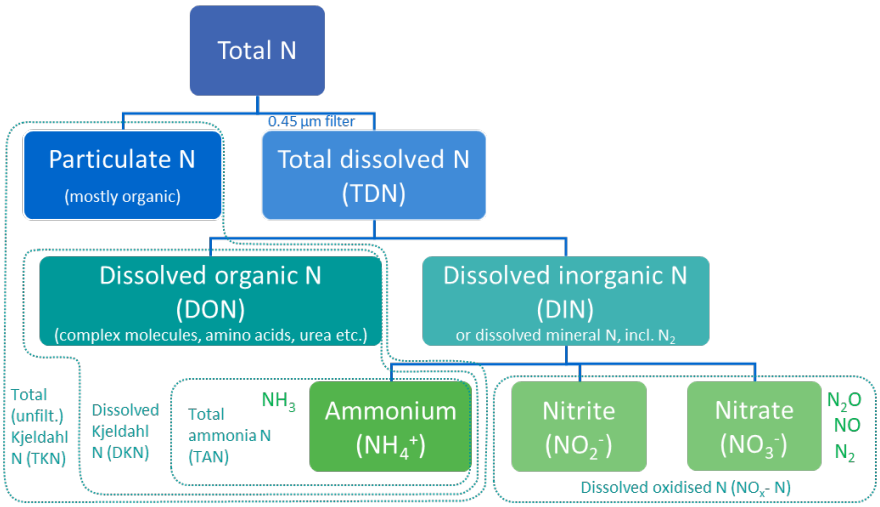

Figure 1: Overview of nitrogen forms in water samples including major operationally defined forms and trace gases (e.g. N_2O, NO, N_2 NH_3). In environmental water samples dissolved oxidised nitrogen (NO_x-N) is comprised predominantly of nitrate, with nitrite concentrations normally being negligible.

The use of bioreactors is a well-established on-farm technology in other parts of the world[5] [6] with various configurations being recognized in the USA as an integral component of both

[5] Addy, K., A.J. Gold, L.E. Christianson, M.B. David, L.A. Schipper and N.A. Ratigan. 2016. Denitrifying Bioreactors for Nitrate Removal: A Meta-Analysis. Journal of Environmental Quality. 45:873-881.
[6] Christianson, L.E. and L.A. Schipper. Ibid.Moving Denitrifying Bioreactors beyond Proof of Concept: Introduction to the Special Section:757-761.

state [7] [8] [9] and federal nutrient management strategies[10]. The use of bioreactors has been identified as a possible management strategy to remediate excess nitrogen in Queensland[11]. Typical configurations (Figure 2) include routing drainage water through a lined trench (bed) filled with high carbon material or establishing an unlined trench (wall) perpendicular to the direction of groundwater flow[12]. While the former can be deployed within existing drainage networks, the latter allows for the in-field interception of groundwater leaving farm paddocks. In both cases, temperature and solute residence time are key determinants of bioreactor efficacy[13] [14] [15], with working lifespan determined by the type and volume of carbon substrate used in relation to nitrogen loads[16] [17].

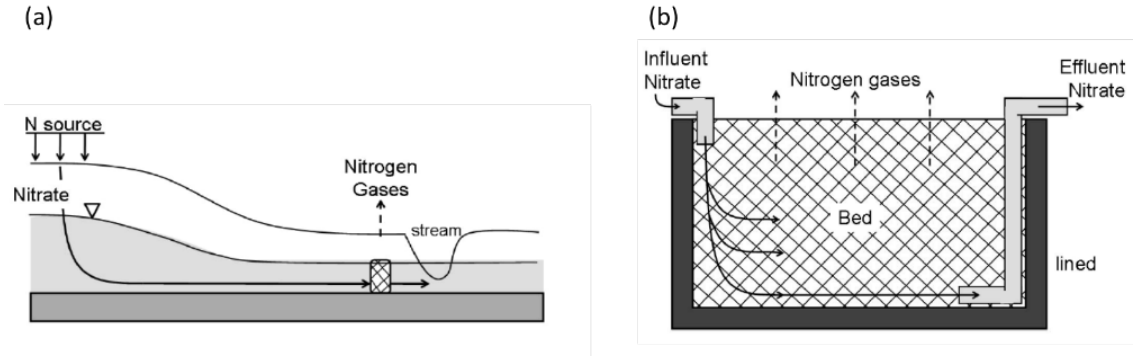

Figure 2: Schematic diagram of denitrifying bioreactor configurations including a) cross-section of a wall intercepting nitrate in shallow groundwater, and b) cross-section of bed treating effluent or drainage water. Images adapted from Shipper et al. (2010)[18]

[7] Iowa nutrient reduction strategy. http://www.nutrientstrategy.iastate.edu/documents (accessed 6th May 2020).
[8] Illinois Nutrient Loss Reduction Strategy. https://www2.illinois.gov/epa/topics/water-quality/watershed-management/excess-nutrients/Pages/nutrient-loss-reduction-strategy.aspx (accessed 6th May 2020)
[9] The Minnesota nutrient reduction strategy. https://www.pca.state.mn.us/sites/default/files/wq-s1-80h.pdf (accessed 6th May 2020)
[10] USDA-NRCS. 2015. Conservation practice standard denitrifying bioreactor code 605 (605-CPS-1). USDA-NRCS, Washington, DC.
[11] Queensland Dept. of Environment and Science Wetland Info https://wetlandinfo.des.qld.gov.au/wetlands/management/treatment-systems/for-agriculture/treatment-sys-nav-page/bioreactors/
[12] Schmidt, C.A. and M.W. Clark. 2012. Efficacy of a denitrification wall to treat continuously high nitrate loads. Ecological Engineering. 42:203-211.
[13] Pluer, W.T., L.D. Geohring, T.S. Steenhuis and M.T. Walter. 2016. Controls Influencing the Treatment of Excess Agricultural Nitrate with Denitrifying Bioreactors. Journal of Environmental Quality. 45:772-778.
[14] Hoover, N.L., A. Bhandari, M.L. Soupir and T.B. Moorman. Ibid.Woodchip Denitrification Bioreactors: Impact of Temperature and Hydraulic Retention Time on Nitrate Removal:803-812.
[15] Rosen, T. and L. Christianson. 2017. Performance of Denitrifying Bioreactors at Reducing Agricultural Nitrogen Pollution in a Humid Subtropical Coastal Plain Climate. Water. 9:16.
[16] David, M.B., L.E. Gentry, R.A. Cooke and S.M. Herbstritt. 2016. Temperature and Substrate Control Woodchip Bioreactor Performance in Reducing Tile Nitrate Loads in East-Central Illinois. Journal of Environmental Quality. 45:822-829.
[17] Long, L.M., L.A. Schipper and D.A. Bruesewitz. 2011. Long-term nitrate removal in a denitrification wall. Agriculture, Ecosystems & Environment. 140:514-520.
[18]Schipper, L.A., W.D. Robertson, A.J. Gold, D.B. Jaynes and S.C. Cameron. 2010. Denitrifying bioreactors—An approach for reducing nitrate loads to receiving waters. Ecological Engineering. 36:1532-1543.

The translation of bioreactor design and implementation to the BSDA and Wet Tropics more broadly must account for the area's climatic, hydro-geomorphic and edaphic setting, as follows:

- The Russell catchment receives some of the highest average rainfall in Australia, with the small size of the catchment and steep topography in its headwaters making the system very responsive to high intensity rainfall events. This produces rapid changes in water height, frequent flooding of low-lying areas and poor distinction between surface runoff and groundwater flows (e.g. flood bypass).
- The topography and geology have resulted in high level of soil complexity across the floodplain. The poorly drained alluvium, which is comprised of clay, silt, sand and gravel, is characterised by waterlogging and rapid surface runoff from waterlogged soils. It has a high water table.
- The BSDA drainage network is incised, with many of the drain beds below the level of the water table. This means that the drainage network collects both surface-runoff and groundwater, with some drains flowing all year due to groundwater seepage.
- The timing of the wet season (from December to March) and/or the preceding wetting up period coincide with or immediately follow fertiliser application after the harvesting period. This produces a distinct 'first flush' of nitrogen from agricultural runoff[19].
- The BSDA produces sediment from bank erosion of the drainage network and farm runoff.
- The 1:50,000 scale agricultural soil survey of the Babinda-Cairns area (BCC polygon) denotes the area to be a complex of Organosols (e.g. Babinda and Sumalee) and Hydrosols (e.g. Timara and Coom). The former is likely to have a high inherent denitrification potential.

These characteristics make the choice and siting of bioreactor types (in-drain 'bed' or end-of-field 'denitrification wall') critical to scientific evaluation of the suitability and effectiveness of bioreactor technology to the conditions of the BSDA, Russell River catchment and Wet Tropics more broadly. For example, 'bed' reactors will be capable of treating both surface runoff and collected groundwater and may perform better at treating DIN, or more specifically NO_x-N during first-flush events but, as they are susceptible to clogging, they are not suited to drains with high sediment loads unless integrated as part of a treatment train e.g. through placement immediately following a sediment trap[20]. Similarly, while denitrification walls provide a low-maintenance option for treating the NO_x-N

[19] Davis, A.M., B. Taylor and S. Fielke. 2019. Engaging with farmers and demonstrating water quality outcomes to create confidence in on-farm decision-making ("Project 25"), p 37.
[20] Christianson, L.E., C. Lepine, K.L. Sharrer and S.T. Summerfelt. 2016. Denitrifying bioreactor clogging potential during wastewater treatment. Water Research. 105:147-156.

component of DIN within groundwater, they may not offer much of an advantage when deployed in soils with an inherently high denitrification potential.

This report addresses the following questions (from project plan):

- What level of confidence is attached to the observed reduction in DIN for each of the bioreactor configurations, as tested?
- Are denitrifying bioreactors a cost-effectives strategy for reducing DIN loads under the conditions of the Wet Tropics, based upon our experience?
- Did the project adequately address possible contra-indications (e.g. pollution swapping) from bioreactor deployment?
- What level of confidence is attached to the extrapolation of findings from a two-year trial to an expected bioreactor lifespan of 10+ years?
- How useful is the integrated modelling of bioreactor performance and landscape drainage pattern in quantifying potential for DIN reduction across the landscape?

Developing a Nitrogen Budget for the BSDA

The study established a nitrogen budget for the BSDA as a means of determining how much nitrogen was suitable for remediation by bioreactors. Establishment of the nitrogen budget involved modelling and combining the results of two discrete areas of research.

1. Developing a water balance for the BSDA.
2. Calculating nitrogen concentrations in the BSDA.

For the purposes of this report, the "BSDA" refers to the catchment of the main drain exiting the BSDA, excluding the catchment of Niringa Creek upstream of the Niringa Creek monitoring station as shown in Figure 3. This area of 3,380 ha includes 2,347 ha managed by the Babinda Swamp Drainage Board.

The BSDA[21], which is predominantly comprised of sugarcane paddocks (Figure 3, Table 1), has a drainage network developed from circa 1950s. This network incorporates a modified Niringa Creek as a main drain that flows through the middle of the BSDA. The 433-ha area upstream of the BSDA includes 41.0% 'conservation and natural environments' (forest) and 30.7% sugarcane (Table 1).

The BSDA can therefore be considered hydrologically isolated, with the only major surface flow entering the BSDA at '*Niringa Creek'* (UTM 55K 384440.11 E, 8077341.34 S) and exiting at *'Christiano Access'* (UTM 55K 388740.36 E, 8081727.64 S, Figure 3). This allowed estimation of the surface water and solute balance for the BSDA by measuring water entering and exiting the BSDA.

[21] The main drains of the BSDA are managed by the Babinda Swamp Drainage Board, established as a water authority under Water Regulation 2016 (Water Act 2000) for the purpose of providing a coordinated drainage system for removal and disposal of excess water from agricultural lands. See: https://www.legislation.qld.gov.au/view/pdf/inforce/current/sl-2016-0216 Accessed: 9/05/2020.

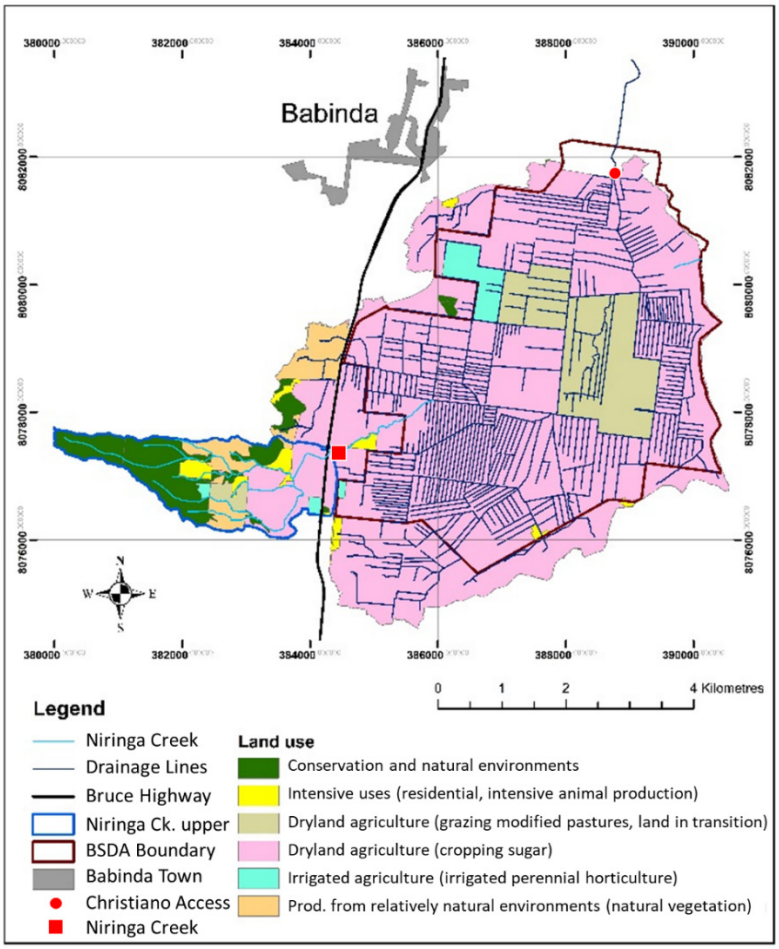

Figure 3: Map of the drainage network and land use of the Babinda Swamp Drainage Area (BSDA) and its catchment. The coloured area is the catchment (delineated from a 1x1 m digital elevation model) of the main BSDA drain at the point where it crosses Christiano Access road. Land use is the Primary land use from the most recent (2015) Queensland Land use Mapping Program (QLUMP) data set produced by the Queensland Government, with secondary land use given in brackets. See Table 1 for areas of each land use. Monitoring stations at Niringa Creek and Christiano Access are shown.

Table 1: Land use in the catchment of the main drain at Christiano Access (Total) and within it, the area upstream of Niringa Creek monitoring station (Upper) and downstream of Niringa Creek monitoring station (Lower), which is slightly larger than the BSDA but is hereafter referred to as the BSDA. Areas were derived from the most recent (2015) Queensland Land Use Mapping Program (QLUMP) data set and are mapped in Figure 3.

Primary Classification	Secondary Classification	Upper area (ha)	Lower area (ha)	Total area (ha)
Conservation and natural environments	Nature conservation	41.6	0.0	41.6
Conservation and natural environments	Other minimal use	135.8	27.1	162.9
Prod. from relatively natural environments	Grazing native veg.	64.8	72.5	137.2
Prod. from dryland agric. and plantations	Grazing modified past.	0.0	372.6	372.6
Prod. from dryland agric. and plantations	Cropping - Sugar	133.0	2810.5	2943.5
Prod. from dryland agric. and plantations	Land in transition	18.2	0.0	18.2
Prod. from irrigated agric. and plantations	Irrigat. perenn. hortic.	9.8	68.8	78.6
Intensive uses	Intensive animal prod.	9.7	0.0	9.7
Intensive uses	Residential	20.4	28.4	48.8
Total		433	3380	3813

Water movement into the BSDA (via Niringa Creek)

Flow in Niringa Creek at the point where it enters the BSDA was determined using continuous measurements of water velocity and depth and a surveyed cross-section. An ultrasonic doppler instrument (Model 6526, Starflow, O'Conner, WA, Australia) was deployed (Figure 4) to provide depth and average water velocity measurements at 10-minute intervals. The stream cross-section profile was measured using a Trimble R8 GNSS RTK (Figure 5). Daily discharge was calculated by summing discharge over the measured 10-minute intervals.

Over 675 days of monitoring between the dates 27/05/2018 and 31/03/2020, discharge flow at this entry point to the BSDA averaged 35.5 ML day^{-1} (Figure 6) but with substantial daily variation. Flows were usually below this long-term average (median 12.3 ML day^{-1}), although a maximum flow of 764 ML day^{-1} was recorded on the 27/01/2019.

Over 2018-2019 year, defined for study purposes as 1/11/2018 to 31/10/2019 and thus incorporating the 2018-2019 wet season, discharge at Niringa Creek equated to 12,784 ML or 2,952 mm yr^{-1} across the (433 ha) catchment, while rainfall was estimated at ~4,358 mm yr^{-1} (from SILO-Australian Climate Data, 17.35°E, 145.95°E), i.e.~ 68% of rainfall was measured as runoff. This leaves the remaining ~1,406 mm as lost from the catchment by evapotranspiration and deep drainage. Evapotranspiration of forests in the region is in the range of 591-1535 mm yr^{-1} [22] [23] and that of sugarcane (the main crop in the catchment) is approximately 1,000 mm yr^{-1} [24]. Given that forest covers approximately 41.0% of the catchment and sugarcane approximately 30.7% (Table 1), it appears that deep drainage was in the order of ~500 mm yr^{-1} over the whole upper catchment.

[22] Connor, S., P.N. Nelson, J.D. Armour and C. Hénault. 2013. Hydrology of a forested riparian zone in an agricultural landscape of the humid tropics. Agriculture, Ecosystems & Environment. 180:111-122.
[23] McJannet, D., P. Fitch, M. Disher and J. Wallace. 2007. Measurements of transpiration in four tropical rainforest types of north Queensland, Australia. Hydrological Processes. 21:3549-3564.
[24] Thorburn, P.J., Biggs, J.S., Webster, A.J., Biggs, I.M. 2011. An improved way to determine nitrogen fertiliser requirements of sugarcane crops to meet global environmental challenges. Plant and Soil 339: 51-67.

Figure 4: Installation and maintenance of Niringa Creek gauging station

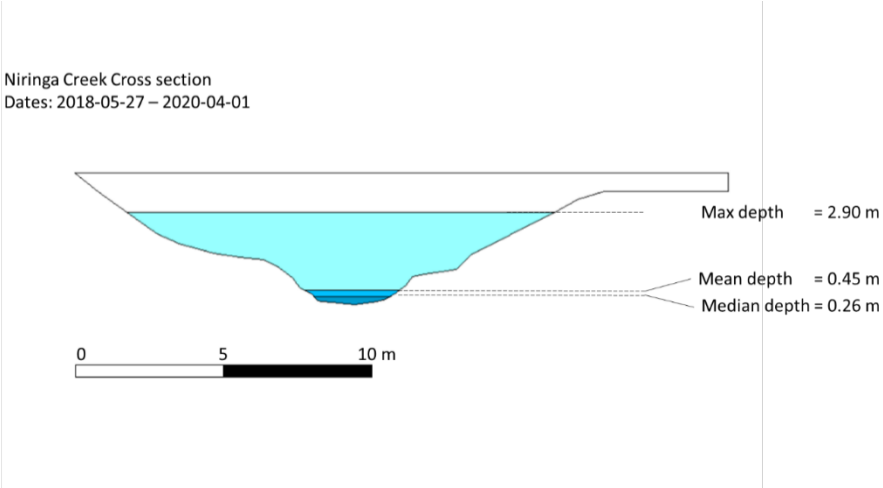

Figure 5: Surveyed stream cross-section at Niringa Creek Monitoring station, including water levels recorded on site.

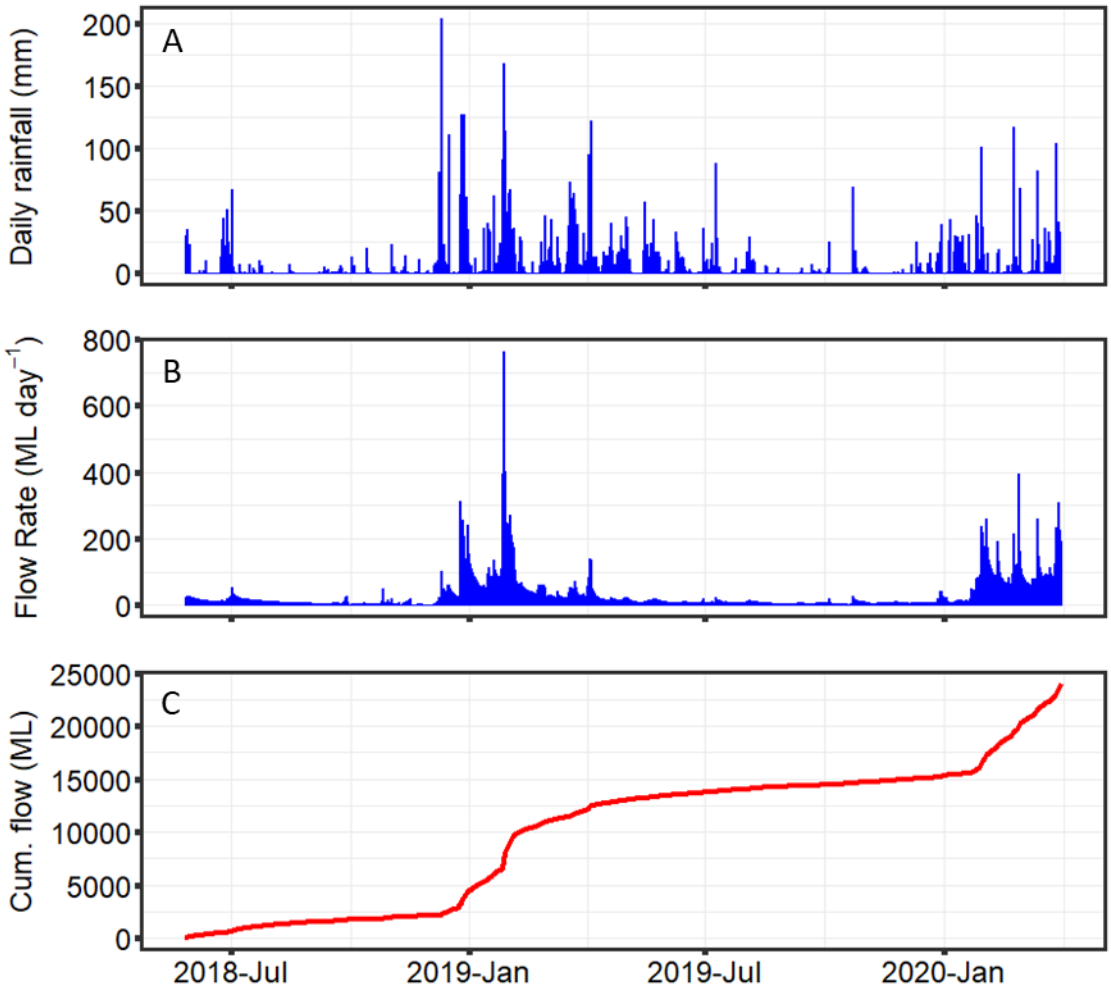

Figure 6: Major water inputs to the BSDA, being A) Rainfall (from SILO-Australian Climate Data grid), and B) Daily and C) Cumulative flow at Niringa Creek monitoring station.

Water movement out of the BSDA (at Christiano Access)

Discharge at Christiano Access was measured as part of *'Engaging with farmers and demonstrating water quality outcomes to create confidence in on-farm decision-making ("Project 25")'*, a project funded by the Department of Agriculture, Water and the Environment - National Environmental Science Program (NESP) - Tropical Water Quality Hub (TWQ Hub)[25] and made available by Aaron Davis (2019). Project 25 uses the station name "Dickson Road" for this site.

Stream stage was measured continuously using a Campbell Scientific CS451 pressure transducer hardwired to a CR1000 data logger (Campbell Scientific, Inc., Logan, UT, USA). Due to minor tidal effects on stream stage, a simple low-pass Butterworth filter (BWF) was applied to remove semi-diurnal tidal signal from the measurements[26]. A stage-

[25] Davis, A.M., B. Taylor and S. Fielke. 2019. Engaging with farmers and demonstrating water quality outcomes to create confidence in on-farm decision-making ("Project 25"), p 37.
[26] Pagendam, D.E. and D.B. Percival. 2015. Estimating freshwater flows from tidally affected hydrographic data. Water Resources Research. 51:1619-1634.

discharge rating curve for the site was derived from repeated discharge measurements over a range of stream stage heights. Instantaneous discharge measures for the rating curve were calculated by conventional current meter methods for wade-able streams at lower stream depths (digital water velocity flow meter, Global Water – FP 311, Xylem, USA) and stream cross-sectional areas. Discharge at higher river stages was calculated from water depth and velocity measurements through a multiple road culvert control section under Christiano Access road[27].

Pre-processed data from Project 25 was collated and resampled at regular 10-minute intervals using a standard spline function. This allowed for the determination of discharge at the same resolution as Niringa Creek as well as the summing of daily total discharge through Christiano Access. Daily discharge was calculated by summing discharge over the measured 10-minute intervals. Daily discharge at Christiano Access, over the period for which corresponding flow data at Niringa Creek was collected (27/5/2018 to 31/3/2020), averaged 150 ML day^{-1}, with a maximum of 1,611 ML day^{-1}.

Over the 2018-2019 year, discharge at Christiano Access (Figure 7) was 82,831 ML. Based on the catchment area of 3,813 ha, this equates to 2,172 mm yr^{-1} runoff across the entire catchment. Given rainfall over the same period was estimated at ~4,358 mm yr^{-1} (SILO grid), runoff was approximately 50% of rainfall. Over this same period, the inflow via Niringa Creek represented 15.4% of outflow at Christiano Access, the remainder (i.e. 70,046 ML) originating within the BSDA.

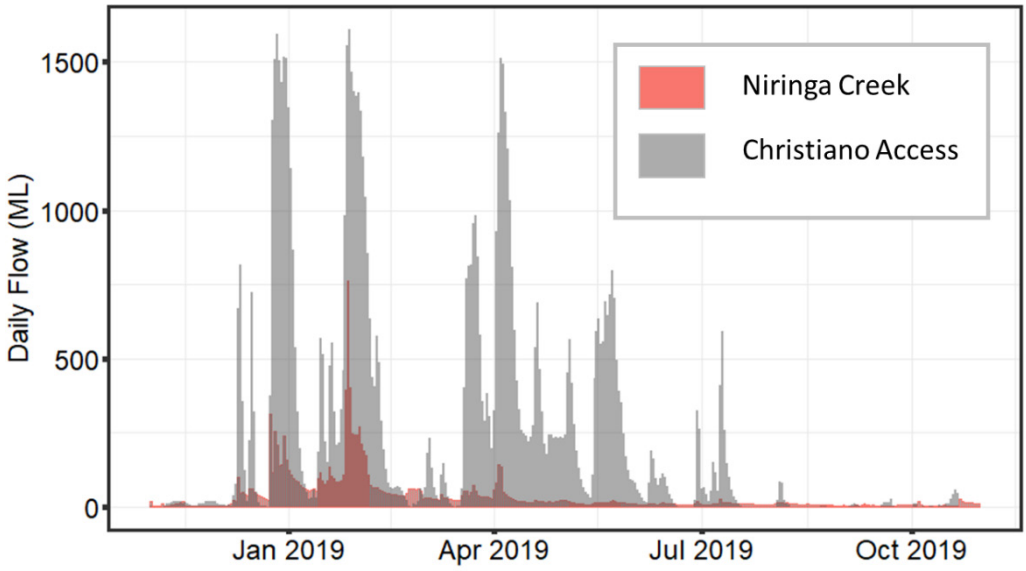

Figure 7: Annual hydrographs over the 2018-2019 year for Niringa Creek and Christiano Access.

Calculating nitrogen concentrations in the BSDA

Development of the nitrogen budget required establishment of water sampling sites to account for nitrogen entering and leaving the BSDA. Three water quality sampling locations

[27] Bodhaine, G.L. 1968. Measurement of peak discharge at culverts by indirect methods. *In* Techniques of Water-Resources Investigations.

were established. Two were the same locations for measuring flow entering (Niringa Creek) and exiting (Christiano Access) the BSDA, while a third site was established at Pughs Creek (UTM coordinates 382717 E, 8076037 N, 55K) to examine water quality of flows leaving the forested portions of the catchment[28].

Grab samples were taken at the three sites approximately every 2 weeks, from 22/10/2018 to 20/3/2020 (Figure 3, Table 3). Sampling methodology, sample bottles, preservation techniques and analytical methodology was in accordance with standard methods[29]. All samples were collected in duplicate, filtered (0.45 µm) and immediately frozen prior to analysis. Procedural control water samples were incorporated into each sampling batch, consisting of a field blank water sample of deionised water, in order to perform a check for laboratory contamination. Samples were analysed for the parameters shown in Table 2.

Table 2. Water quality variables measured and mentioned.

Acronym	Name	Description
TDN	Total dissolved nitrogen	All the nitrogen in a filtered (<0.45 µm) sample, comprised of dissolved inorganic nitrogen (DIN) and organic nitrogen (including urea). DIN consists of NO_x-N and ammonium N (NH_4-N)
NO_x-N	Oxidised nitrogen	Nitrate (NO_3^-) plus nitrite (NO_2^-, which is usually negligible), all of which is dissolved, so filtration is irrelevant

For all samples, NO_x-N was on average 61% of TDN, but samples with high TDN usually had a higher proportion of NO_x-N (Figure 8). Stream water leaving the forested catchment had significantly (p<0.001) lower TDN and NO_x-N concentrations than either Niringa Creek or Christiano Access (Table 3). Samples taken at the edge of the forest averaged 0.10 ± 0.08 mg TDN L^{-1} (n=41) of which, on average, 57% was NO_x-N. Samples at Niringa Creek (n=43) averaged 0.39 ± 0.17 mg TDN L^{-1}, with a maximum value of 0.68 mg TDN L^{-1}. Water quality at Christiano Access (n=43), while not significantly different to Niringa Creek (TukeyHSD, p=0.99), was more variable, with an average 0.39 ± 0.37 mg TDN L^{-1} and a maximum of 1.5 mg TDN L^{-1}. Daily mean concentrations at Niringa Creek and Christiano Access were estimated by linear interpolation between measured values from the grab samples (Figure 9).

[28] Pughs Creek, which is in the adjacent catchment and of similar forested area, was used instead of Niringa Creek. This was because Niringa Creek was not accessible at the point where it leaves the forest and water quality collected any further downstream would have been influenced by a commercial plant nursery.
[29] DES. 2018. Monitoring and Sampling Manual: Environmental Protection (Water) Policy. Department of Environment and Science Government, Brisbane.

Table 3: Mean dissolved nitrogen concentration in grab samples taken from water entering and leaving the BSDA. Samples were collected on an approximately fortnightly schedule between 22/1/2018 and 20/03/2020. Locations were compared using one-way ANOVA. TDN and NO_x-N concentrations both differed significantly ($p<0.001$) between locations. Superscript letters denote significantly different means as determined by post-hoc Tukey HSD.

	#	TDN (mg L^{-1})				NO_x-N (mg L^{-1})			
Location	samples	mean	±	sd	max.	mean	±	sd	max.
Forest (upstream)	41	0.10^a	±	0.08	0.46	0.03^a	±	0.03	0.19
Niringa Creek (BSDA inlet)	43	0.39^b	±	0.17	0.68	0.30^b	±	0.16	0.53
Christiano Access (BSDA outlet)	43	0.39^b	±	0.37	1.5	0.27^b	±	0.39	1.46

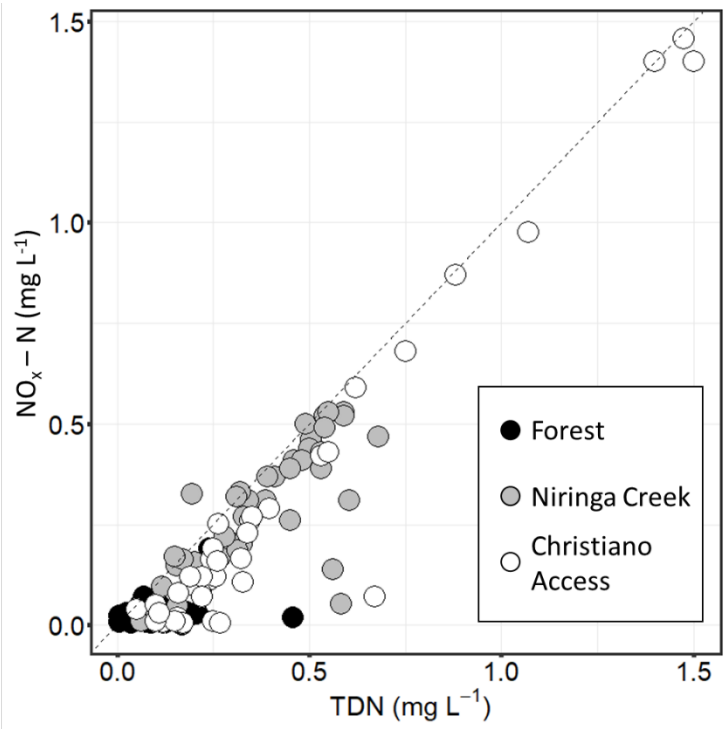

Figure 8 Comparison of TDN (Total dissolved nitrogen) and NO_x-N in grab samples taken from water entering the BSDA (Niringa Creek and Forest) and leaving the BSDA (Christiano Access).

Nitrogen movement into the BSDA (via Niringa Creek)

Nitrogen load in Niringa Creek was calculated on a daily time step by multiplying daily discharge by daily mean concentration. Over the 2018-2019 year an estimated 3,604 kg TDN, including 2,650 kg NO_x-N, passed into the BSDA via Niringa Creek. The catchment of the Niringa Creek monitoring station is 433 ha (Figure 3) resulting in an annual load over the 2018-2019 year of 8.3 kg ha^{-1} yr^{-1} of TDN including 6.1 kg ha^{-1} yr^{-1} of NO_x-N. This catchment (Table 1) includes 41% land used for 'conservation and natural environments' (forest), and 39% used for some form of intensive or modified agricultural production (e.g. intensive animal production, sugarcane and irrigated fruit trees).

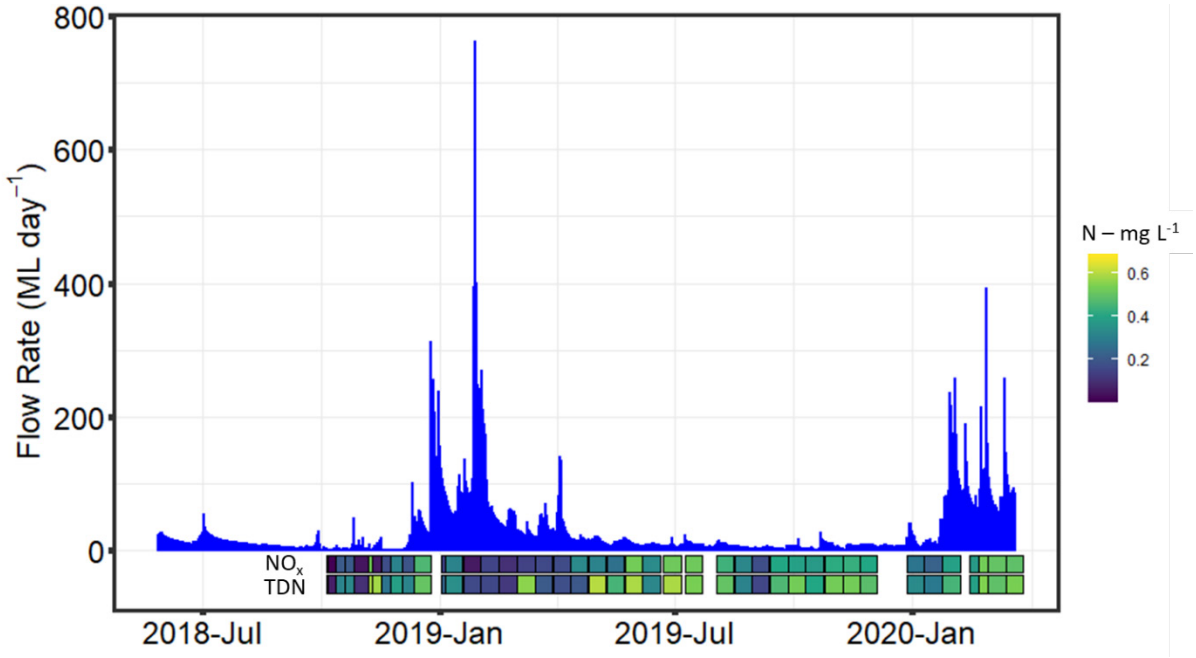

Figure 9: Daily flow and concentration of dissolved nitrogen forms in grab samples collected at Niringa Creek monitoring station.

Nitrogen movement out of the BSDA (at Christiano Access)

Nitrogen load at Christiano Access was calculated on a daily time step by multiplying daily discharge by daily mean concentration. Over the 2018-2019 year, an estimated 31,505 kg TDN, including 20,714 kg NO_x-N, left the BSDA via the main drain. Given the 3,813 ha of the catchment (Figure 3), this equates to an average load per unit catchment area of 8.26 and 5.43 kg ha^{-1} yr^{-1} of TDN and NO_x-N, respectively. The load derived from the 3,380-ha portion of the catchment in the BSDA (i.e. subtracting the load entering the BSDA from the load leaving the BSDA) equates to 8.25 and 5.34 kg ha^{-1} yr^{-1} of TDN and NO_x-N, respectively.

Of the nitrogen leaving the BSDA over the 2018-2019 year, 11.4% of TDN and 12.8% of NO_x-N was contributed by the area above Niringa Creek monitoring station, which was less than its relative 15.4% of total discharge. Therefore, it can be concluded that while water entering the BSDA from upstream areas is heavily impacted by human activities, most of the nitrogen leaving Christiano Access originates from the BSDA.

It should be noted that while standard methods were used to interpolate nitrogen concentrations between analyses, this may poorly represent the dynamic nature of water quality on site. The use of higher frequency analysis (either through more discrete sampling, or use of real-time sensors) may improve the accuracy of these load estimates.

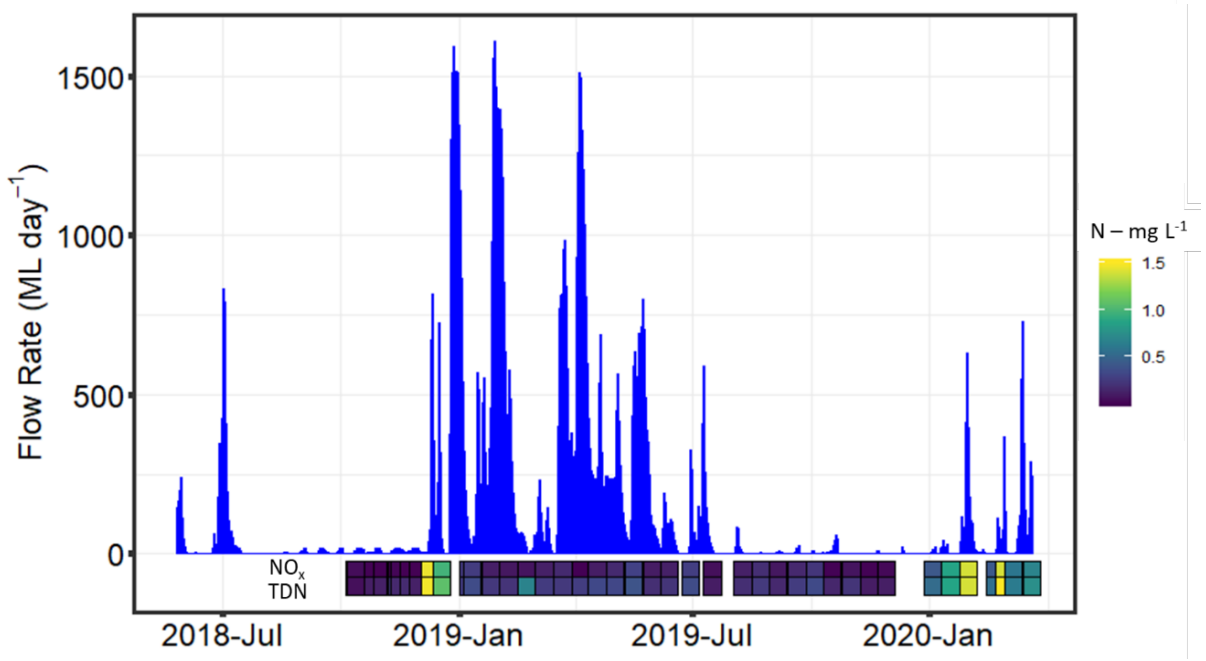

Figure 10: Daily flow and concentration of dissolved nitrogen forms in grab samples collected at Christiano Access monitoring station

Efficacy of Denitrifying Bed Bioreactors

Design and installation

Three trial sites were selected for installation of three bed bioreactors, two having 'inline' configuration, and the third having 'offline' configuration. Site selection followed detailed prospective appraisal of numerous sites within the BSDA that was undertaken in consultation with Australian Wetland Consultancy and landowners. The appraisal involved consideration of site characteristics such as soil and surface topography (see Appendix 1, & 2) as well as initial assessment of water quality across the landscape (Appendix 3).

Two 'inline' bioreactor beds were installed on 'Farm 1' in the southern portion of the BSDA on 29/08/2018 to 1/09/2018. They were installed below the base of parallel drains in adjacent paddocks, just upstream of where the drains joining the larger drain (Figure 11, Figure 13). The beds consisted of 17.5 m^3 of woodchips (20 m long, 0.99 m wide and 0.88 m deep) with a gravel plug at either end (0.5 m^3 each, with 100mm aggregate at the inlet and 20-30 mm aggregate at the outlet), surrounded by geotextile and plastic, except for the inlet ramp and the outlet pipe. The top of the bed was 0.2 m below the bottom of the drain and the bed had a fall of 0.2 m from inlet to outlet. At the inlet the gravel was built up slightly, and a small gravel dam was built ~10 m upstream from the inlet to trap sediment. The outlet pipe (100 mm diameter PVC) passed underneath the culvert and emptied into a larger drain flowing perpendicular to the bed drain.

An 'offline' bed design was installed on 'Farm 2' in the northern portion of the BSDA on 24/09/2018 to 25/09/2018. This involved routing water from a drain to a bioreactor bed constructed parallel to the drain before exiting into the main drain (Figure 12, Figure 14). The bed consisted of 15.4 m^3 of woodchips (20.0 m long, 1.08 m wide and 0.71 m deep) with a gravel plug at either end (0.5 m^3 each, with 100 mm aggregate at the inlet and 20-30 mm aggregate at the outlet), surrounded by geotextile and plastic, except for the inlet and outlet pipes (100-mm diameter PVC). The top of the bed was 1.70 m below ground surface and the bed had a fall of 0.32 m from inlet to outlet. The inlet pipe started in the drain and various types of filters were trialled in attempts to prevent it clogging. The outlet pipe emptied into a larger drain flowing perpendicular to the intercepted drain.

The woodchips used in all bioreactors were mixed hardwood (*Eucalyptus*) species and represent the same product used in bioreactor trials in SE Queensland and the Burdekin being run by other members of B4GBR network. After delivery to Babinda they were 'aged' for 10 months by being left exposed to the elements from November 2017 to September 2018. The chips were analysed for total C and N content using a LECO CNS TruMAC Analyser. Total carbon content was 48.9 % and total nitrogen content 0.10 %, giving a C:N mass ratio of 489. Total phosphorus content was 71 mg kg^{-1}.

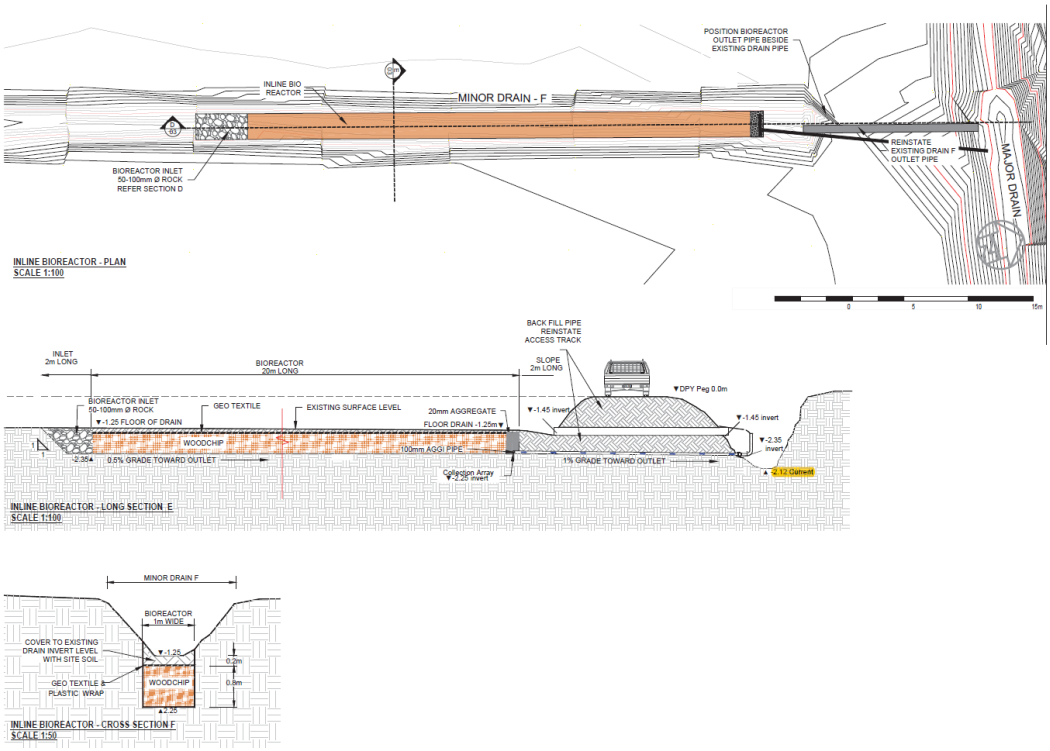

Figure 11: Design of inline bed bioreactor as drawn up by Australian Wetland Consultancy (AWC)

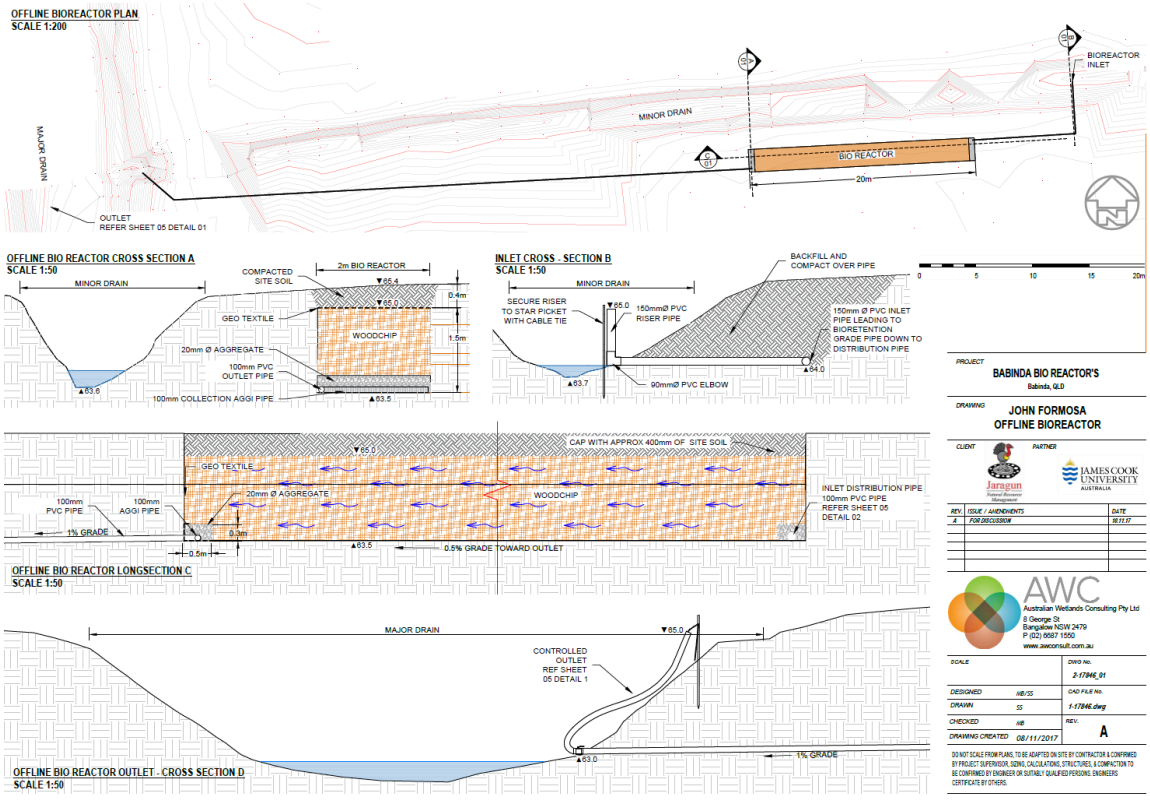

Figure 12: Design of offline bed bioreactor as drawn up by Australian Wetland Consultancy (AWC)

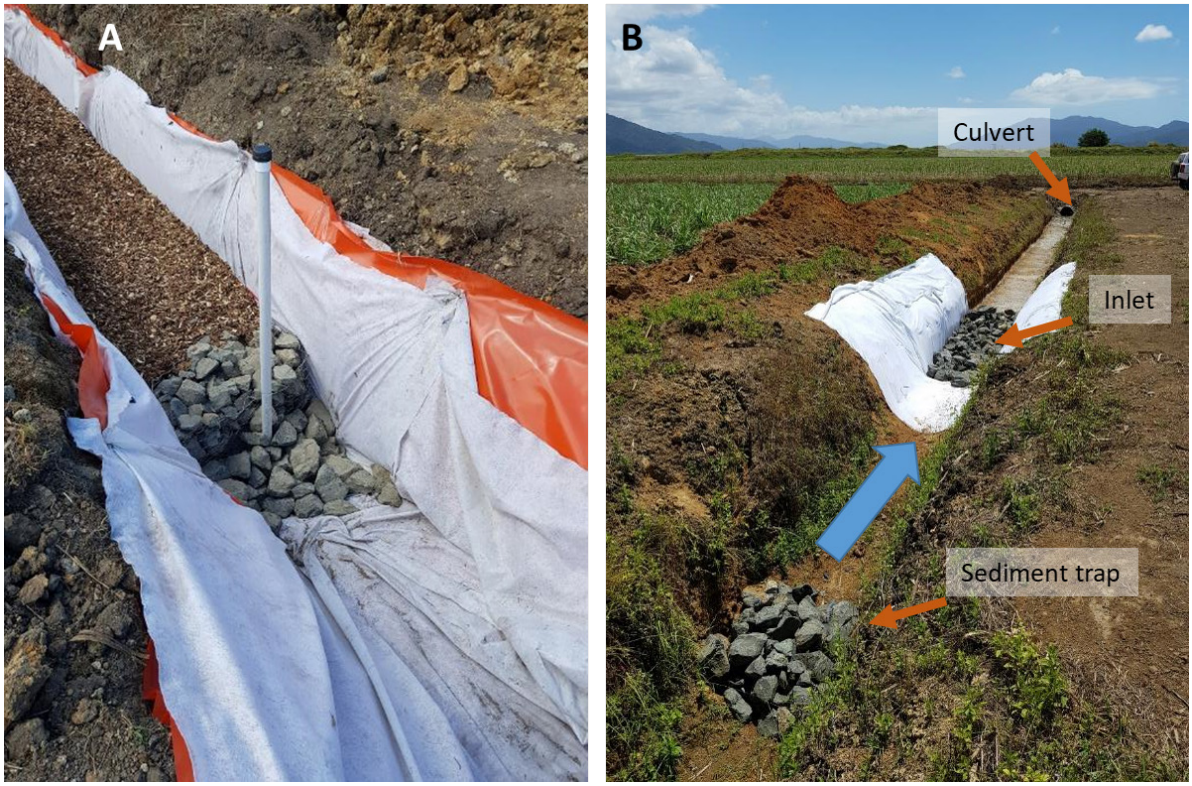

Figure 13. Installation of one of the two Inline Bed bioreactors, showing (A) gravel inlet and sampling piezometer, woodchip and surrounding geofabric and plastic before covering, and (B) completed bed, looking downslope.

Figure 14. Construction of the Offline Bed bioreactor, showing (A) geofabric lining before inserting woodchips, and (B) woodchips, before covering and sealing.

Installation costs

The construction cost of a single inline bed bioreactor within an agricultural drain in the Wet Tropics was approximately $6,653 (Table 4. Cost of constructing each inline bed bioreactor.). This is the amount needed for broader implementation of bioreactors using the design principles contained in the State guidelines and based on availability of local supply of materials and design expertise.

The cost considerations relate to topographic survey, materials and construction. Pre-construction site assessment for the purposes of the trial required topographic survey, although the above cost is not expected for future implementation. The purpose was to ensure the bioreactor bed depth was constructed to specifications, with drainage unaffected through reinstatement of the original drain bed depth following installation.

The main construction costs involved excavator hire. For drains typical in Wet Tropics sugarcane, an excavator able to straddle the width of the drains is required to dig the bioreactor bed to avoid damage to drain banks. A survey technician and assistant were needed during excavation to confirm the bed depth and gradient through the length of the bed for drainage purposes.

Rock was needed at four locations. An upstream sediment trap was required to trap and prevent silt from entering and clogging the bioreactor. Rock was required at the inlet and outlet to contain the woodchip. At the inlet, it further served to direct flow unimpeded into the bioreactor. Rock work was required on the headland, on either end of the outlet pipe to prevent erosion following soil disturbance.

To reduce erosion following construction, cement dust and a compactor were used to re-harden the drain bed.

The cost of construction given here does not include the costs of experimental monitoring features such as the plastic ‘skin’ and piezometers.

Table 4. Cost of constructing each inline bed bioreactor.

Activity	Description	Unit	Price	Total $
Construction materials				
Woodchip supply m^3	Softwood (local supply)	16	22	352
Woodchip cartage	Off-site storage	2	100	200
Float hire	Excavator float to & from site	2	150	300
Excavator hire	Bioreactor construction	8	130	1040
Geofabric	50m x 2m	1	146	146
Rock (100mm)	Riverstone m3	2	30	60
Rock (20-30mm)	Riverstone m3	0.5	30	15
Rock delivery	Delivery	1	50	50
Gabion mesh/cage	100mmx100x50mm	2	93	186
Tie wire (galvanised)	0.9mm x 150m	1	9	9
Star pickets	1.5m	2	12	24
PVC pipe	100mm x 6m	1	24	24
Agroflex draincoil	100mm x 20m	1	95	95
T-piece	100mm	1	8	8
Dirty water pump hire	Remove excess water	1	100	100
Whacker packer hire	Compacting soil	1	60	60
Cement (20 kg)	Dusting do seal drain surface	4	7.5	30
Marker	Can	1	10	10
Incidentals		1	100	100
			Sub-total	**2808**
Project management				
Designer (8 hour day)	2 planning, 1 construction	24	120	2880
General labour (8 hour day)	1 planning, 1 construction	16	45	720
Travel	0.72 cents p km	340	0.72	245
			Sub-total	**3845**
Equipment needs				
Measuring tape	60m	1		
Fencing pliers		1		
Mallet		1		
Shovels	Fine clean drain & woodchip	2		
Rakes	Level woodchip	2		
			Total	**6653**

Note: The table does not include additional costs of materials, equipment and labour for scientific assessment and monitoring purposes in this trial.

Bed bioreactor performance – influence on dissolved nitrogen concentration

The capacity for denitrifying bioreactors to intercept and remove NO_x-N was assessed by comparing concentrations of nitrogen in inlet and outlet water. That was done by periodically taking grab samples from all bed bioreactors over the period 13/10/2018 to 20/03/2020. In one of the in-line beds (Inline Bed 1), additional samples were taken over the course of specific events (e.g. first flush and flooding), using two programmed autosamplers (ISCO 3700, Teldyne ISCO, Lincoln, NE, USA) housed within a remote sampling trailer (Figure 15).

Figure 15: Remote sampling trailer with ISCO-3700 autosamplers on board installed alongside Inline Bed 1 within the BSDA

Concentrations of TDN and NO_x-N in drain water at the bioreactor sites, as measured periodically in inlet samples, was generally moderate, with means <0.5 mg L^{-1} (Table 5). These concentrations are substantially higher than in water from the forested catchment and similar to those in the main BSDA drain (Table 3). However, they are substantially lower than those in other denitrifying bioreactor deployments around the world, where 5 mg N L^{-1} has been considered as "low"[30]. Interestingly, the composition of N in these samples differs from those in the main drain; NO_x-N generally made up a lower proportion of the TDN than in the main drain. The difference may be due to unhydrolysed urea in the paddock runoff. Fertiliser had been applied to the surface as a granular blend and there was little rainfall between application and the first-flush runoff event (see 'Bed bioreactor performance' section below for details).

All three bed bioreactors reduced the concentration of NO_x-N in water flowing through them (Figure 16). NO_x-N concentration was reduced to zero on most occasions and reduced on all occasions, apart from one at the very start of deployment. The mean reduction in NO_x-N concentration across all samples and bioreactors was 41% (p<0.001, df=73, Adj-R^2=0.95, coefficient =0.59±0.015, Figure 16). There was no significant difference between bioreactors in terms of concentration reduction. However, it is apparent that the overall relationship is substantially driven by a small number of grab samples collected during periods of high inlet (and correspondingly high outlet) NO_x-N concentration.

To better assess the efficacy of the bed bioreactors, results from periodic grab sampling were combined with those collected by automated samplers for Inline Bed 1. The automatic sampling produced daily composite samples from the bed inlet and outlet, taken during periods of dynamic discharge, including first-flush and flooding events. The daily composites were comprised of 6 x 200-mL samples taken at 4-hour intervals. On days in which both composite samples and grab samples were collected the composite samples were used as a

[30] Addy, K., A.J. Gold, L.E. Christianson, M.B. David, L.A. Schipper and N.A. Ratigan. 2016. Denitrifying Bioreactors for Nitrate Removal: A Meta-Analysis. Journal of Environmental Quality. 45:873-881.

better representation of daily average conditions. The resultant data set had inlet and outlet concentrations representing 77 days between the dates 13/10/2018 and 20/03/2020.

The relationship between inlet and outlet concentrations of Inline Bed 1 (Figure 17, $p<0.001$, df = 75, Adj-R^2= 0.78, slope coefficient 0.59 ± 0.04) shows that the mean reduction in NO_x-N concentration was 41%. Given that this value is identical to that determined for all bioreactors using the grab samples (Figure 16), it appears that the value is a robust estimate for the reduction in NO_x-N concentrations given the designs used and deployment context of the BSDA. Calculation of N removal rate, in g-N m^{-3} day^{-1}, can be found below.

The change in TDN concentration through Inline Bed 1 (Figure 18) was similar to the change in NO_x-N concentration. Across all data points, TDN declined by 48% between the inlet and outlet samples ($p<0.001$, df=74, Adj-R^2= 0.51, slope coefficient 0.52 ± 0.06). Therefore, the reduction in NO_x-N concentration through the denitrifying bioreactor beds was related to a reduction in total nitrogen load and a loss to atmosphere rather than a conversion to other forms of dissolved nitrogen.

Table 5 Mean dissolved nitrogen concentration of grab samples taken from the inlet and outlet of bed bioreactors (± 1 standard deviation).

Bed & location	No. of samples*	TDN (mg L^{-1})	NO_x-N (mg L^{-1})
Offline -Inlet	21	0.32 ± 0.32	0.11 ± 0.25
Offline -Outlet	20	0.51 ± 0.29	0.02 ± 0.05
Inline 1 -Inlet	36	0.46 ± 0.91	0.28 ± 0.77
Inline 1 -Outlet	34	0.44 ± 0.76	0.17 ± 0.47
Inline 2 -Inlet	29	0.31 ± 0.75	0.22 ± 0.71
Inline 2 -Outlet	28	0.30 ± 0.67	0.10 ± 0.45

*missing samples from were due to restricted site access during dangerous conditions.

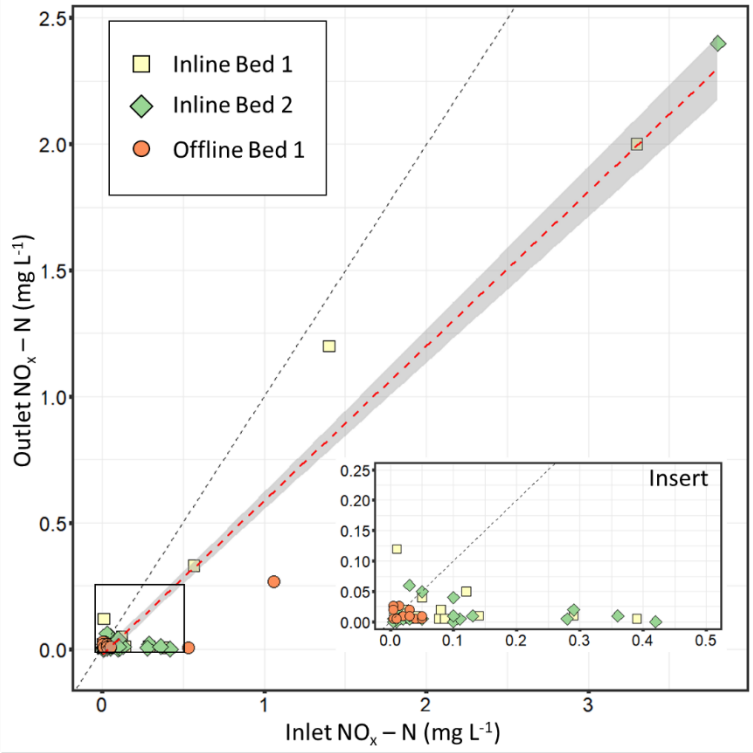

Figure 16: Comparison of NO_x-N concentration in inlet and outlet water collected as grab samples at three denitrifying bioreactor beds installed in the BSDA. The black line is the 1:1 relationship and the red line is the significant linear relationship across all data (p<0.001, df=73, Adj-R^2=0.95, slope coefficient =0.59±0.015).

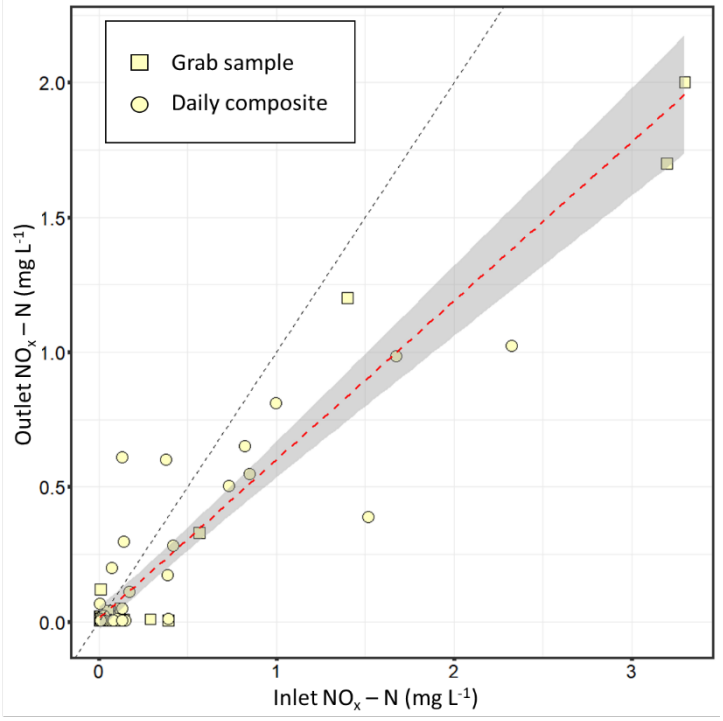

Figure 17: Comparison of NO_x-N concentration in inlet and outlet water collected as both grab samples and daily composites at Inline Bed 1 (n=77). The black line is the 1:1 relationship and the red line is the significant linear relationship across all data (p<0.001, df=75, Adj-R2= 0.78, slope coefficient 0.59±0.04).

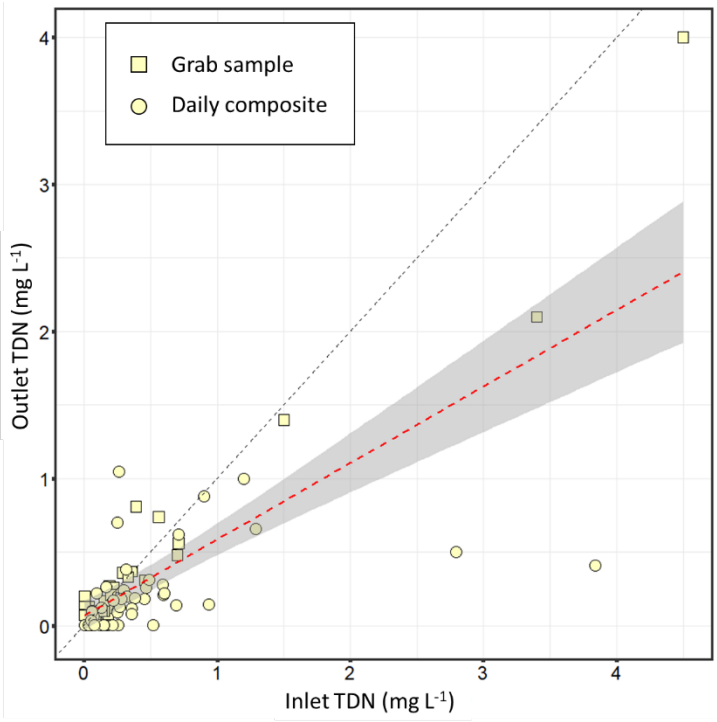

Figure 18: Comparison of TDN concentration in inlet and outlet water collected as both grab samples and daily composites at Inline Bed 1 (n=76). The black line is the 1:1 relationship and the red line is the significant linear relationship across all data (p<0.001, df=74, Adj-R2= 0.51, slope coefficient 0.52 ± 0.06).

Bed bioreactor performance – nitrogen load reduction

The nitrogen load in the drain upstream of the bioreactor was calculated by multiplying daily discharge by nitrogen concentration. The proportion of this load passing into the bioreactor was multiplied by the bioreactor efficiency (i.e. 41%) to determine the nitrogen load reduction.

Drain discharge was determined using continuous stage measurements and rating curves established using the channel cross-sectional area and occasional velocity measurements. Depth was measured using pressure transducers (CS451, Campbell Scientific, Logan, UT, USA) and recorded with solar-powered data loggers (CR300, Campbell Scientific, Logan, UT, USA). Site specific rating curves were determined by recording water velocity over 2-week deployments of a Doppler instrument (6527 Starflow QSD, Unidata O'Connor WA, Australia) and accurate surveys of the drain cross sectional area using a RTK GPS (Trimble R8 GNSS, Figure 19). Daily discharge was calculated by summing discharge over measured 5-minute intervals.

Daily average water depths and discharge were highly variable in the drains accommodating Inline Beds 1 and 2 ('drains 1 and 2', Figure 19). Both drains were dry for much of the dry season but during the wet season there were periods when water overtopped the drains and flooded the surrounding paddock (Figure 20). Drain 1's daily average depth exceeded 1.0 m (full depth) for 7 of the 545 days of monitoring, and drain 2's depth exceeded 1.1 m (full depth) for just one day. These flood days in drain 1 included 3 consecutive days in

January 2019 which recorded a 10-min maximum depth 1.96 m, although the maximum daily average was just 1.46m. Although not a regular occurrence these flooding events, resulting from intense rainfall, high Russell River stage and high tides retarding drainage across the BSDA, do make accurate estimation of drain flows problematic due to movement of water outside the channel (outside the rating curve relationship) and unknown changes to the rating curve within the channel. Therefore, during 'flood events' calculated drain discharge was assumed to remain constant at 32.1 L s^{-1} in drain 1 and 26.1 L s^{-1} drain 2, corresponding to the depth at which the culverts leaving the drain into the receiving drain were full (~0.6 m). Above this point drain discharge was restricted by the culvert dimensions.

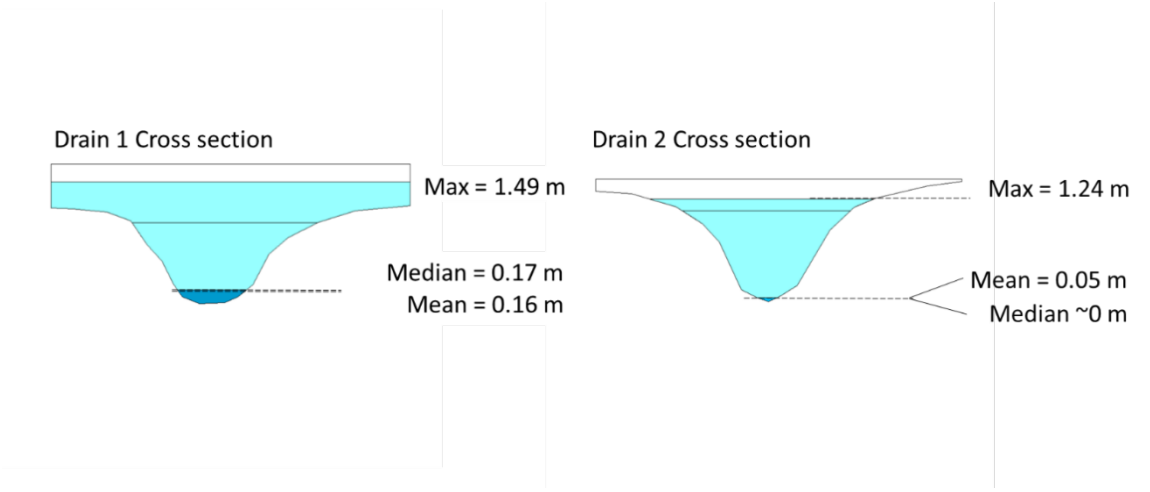

Figure 19: Cross section profile and summary of daily average water depths of drains at the inlets of Inline Beds 1 and 2.

Figure 20: Flooding of drain and surrounding paddocks observed in the BSDA. The sites were inaccessible at higher flood levels.

Daily mean concentration was estimated for each day by linear interpolation between measured values from all grab and composite samples collected. There was a high degree of temporal variation in nitrogen concentrations in drain 1 (Figure 21, Figure 22) It is worth

noting that the initial grab samples collected on 13/10/2018 and 15/10/2018 represent the initial wetting up of the bioreactor during a minor flow event (Figure 23).

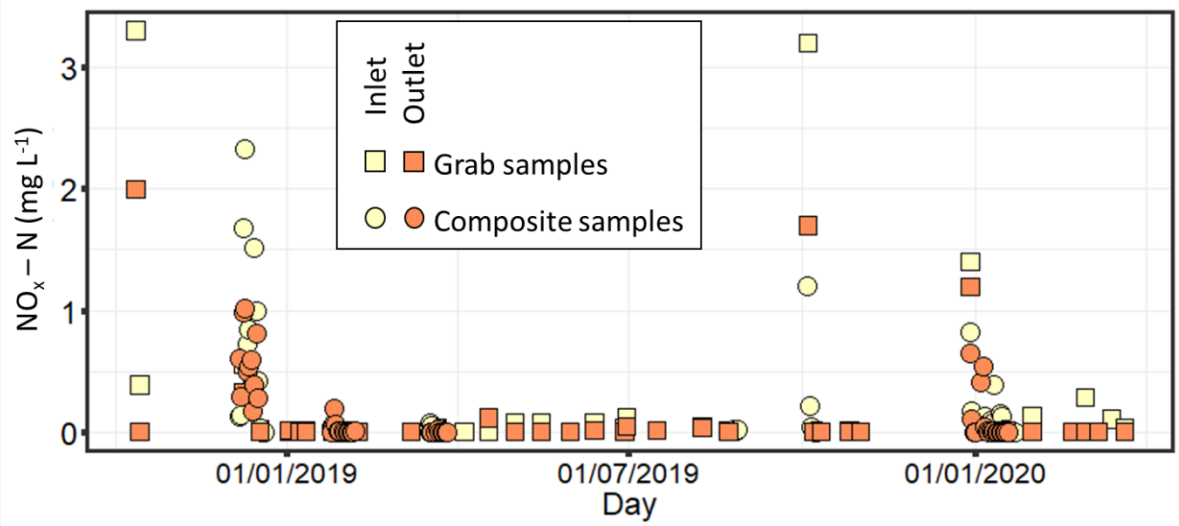

Figure 21 NO_x-N concentrations in all water samples from Inline Bed 1. Daily composite samples were collected using an autosampler.

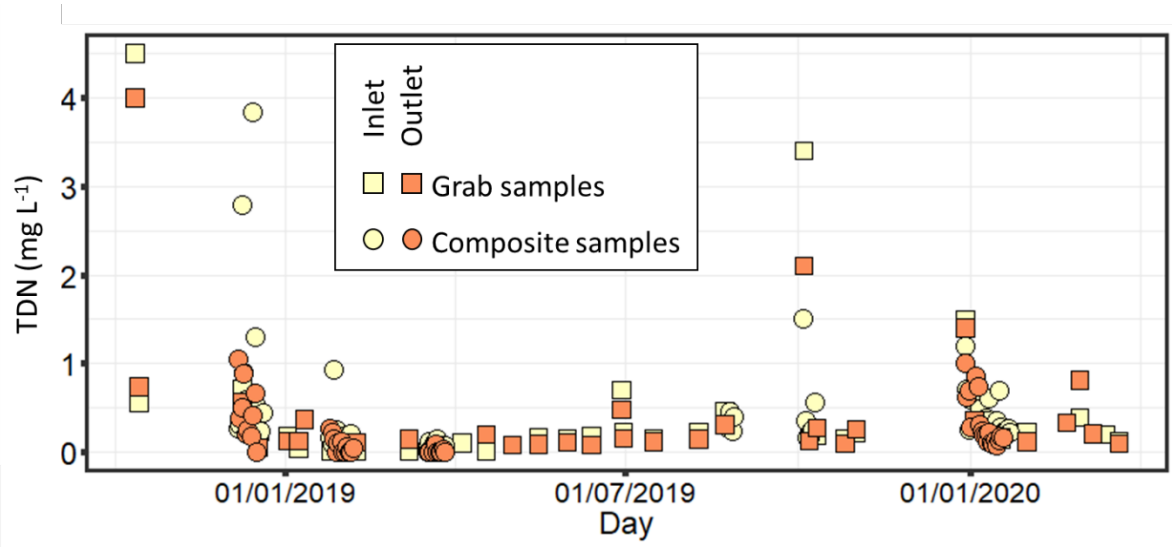

Figure 22: TDN concentrations in all water samples from Inline Bed 1. Daily composite samples were collected using an autosampler.

In the 2018-2019 year, taken from the first minor rains after paddock fertiliser application (13/10/2018 to 12/10/2019) the estimated load in the drain feeding Inline Bed 1 totalled 42.2 kg TDN, including 16.0 kg NO_x-N. Assuming a contributing area of 4.33 ha (estimated using a 1x1 m digital elevation model of the surrounding paddocks, Figure 24) this equates to an annual load of 9.7 kg TDN ha^{-1} including 3.7 kg NO_x-N ha^{-1}. A high proportion of the load occurred during a large 'first-flush' event, with a 10-day period (i.e. 08/12/2018 to 17/12/2018) responsible for 11.6 kg NO_x-N (72% of annual NO_x-N load) and 15.6 kg TDN (37% of annual load). This period was the first time that significant flow occurred in the drain, and was associated with high concentrations.

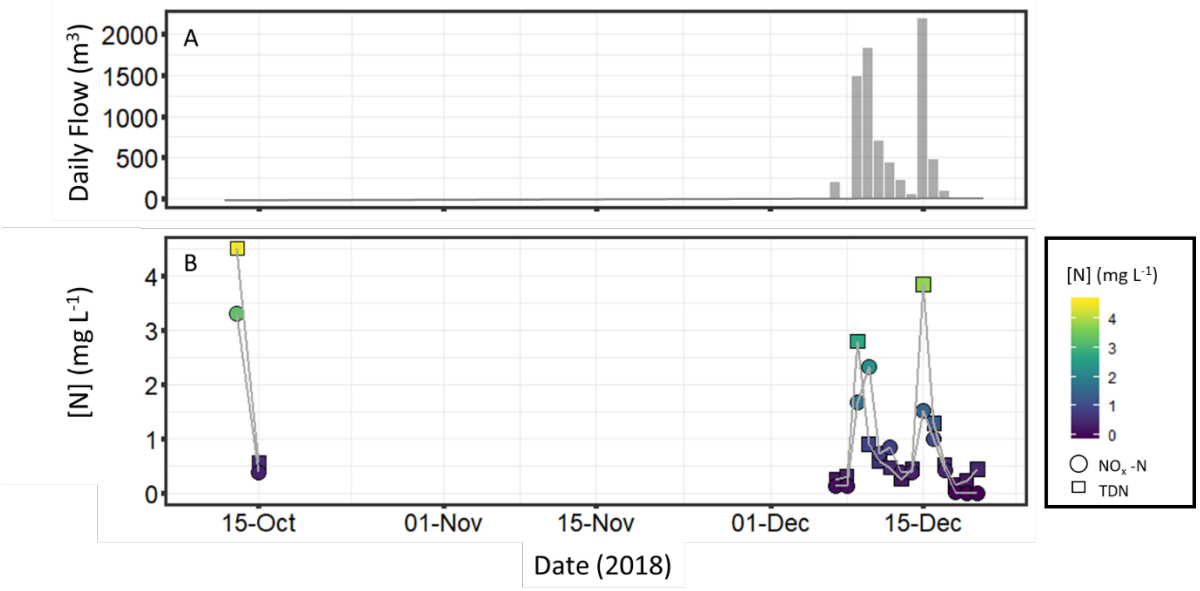

Figure 23: Daily flows (A) and concentrations of TDN and NO_x-N (B) observed in Drain 1 on Farm 1 of the Babinda Swamp Drainage Area. Note initial water samples collected on the 13th and 15th of October 2018 correspond with total daily drain flows of < 3 m^3, plotted but not visible.

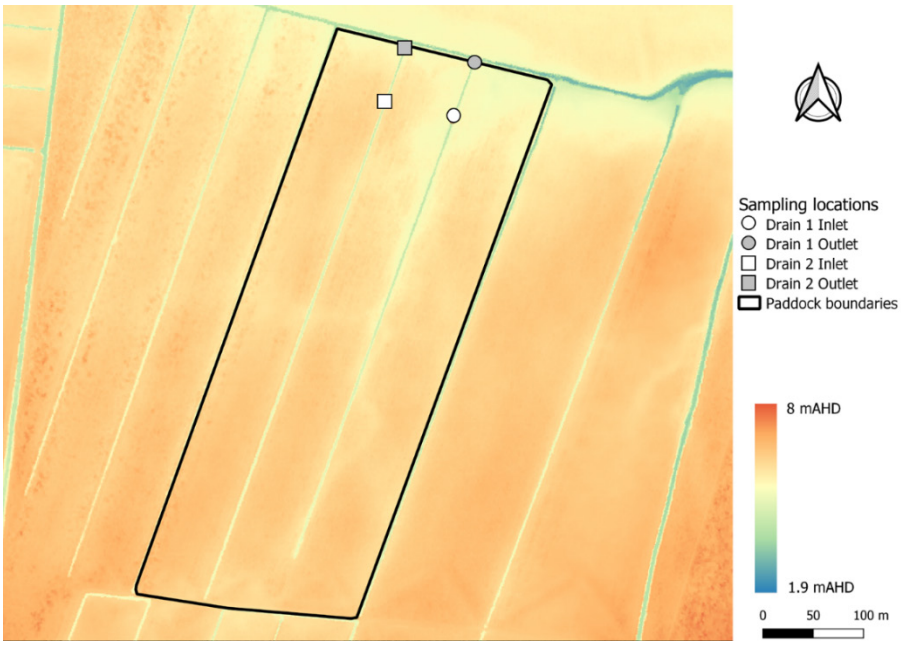

Figure 24 Surface topography of paddocks surrounding Drains 1 & 2, and locations of the inlets and outlets of Inline Beds 1 & 2. Digital elevation data sourced from the Department of Natural Resources and Mines. CRS: EPSG:28355 - GDA94 / MGA zone 55. URL: https://elevation.fsdf.org.au/.

The annual TDN loss in the 2018-2019 year was equivalent to 8.7% of the amount of nitrogen applied in fertiliser to the surrounding paddocks. Fertiliser had been applied according to 'Best Management Practice', on 10/10/2018, 59 days before the large first-flush event (although there was a small runoff event on 13-14/10/2019 which contained high nitrogen concentrations but negligible load). It was applied as a surface dressing in a

blend containing 20.2% nitrogen (17.5% as urea and 2.7% as ammonium) at a rate of 4.5 bags of fertiliser per acre, which equates to 112 kg N ha^{-1}. In the 2019-2020 season, the first-flush runoff event came 99 days after fertiliser application, which was on 21/09/2019. The longer period between fertiliser application and runoff in the second year was presumably responsible for the smaller load in the 2019-2020 first-flush event, wherein daily load leaving the 4.33 ha paddock area never exceeded 0.49 kg day^{-1} NO_x-N, or 0.79 kg day^{-1} TDN during the 182 days post fertiliser application that were included in this monitoring effort (Figure 25, Figure 26)

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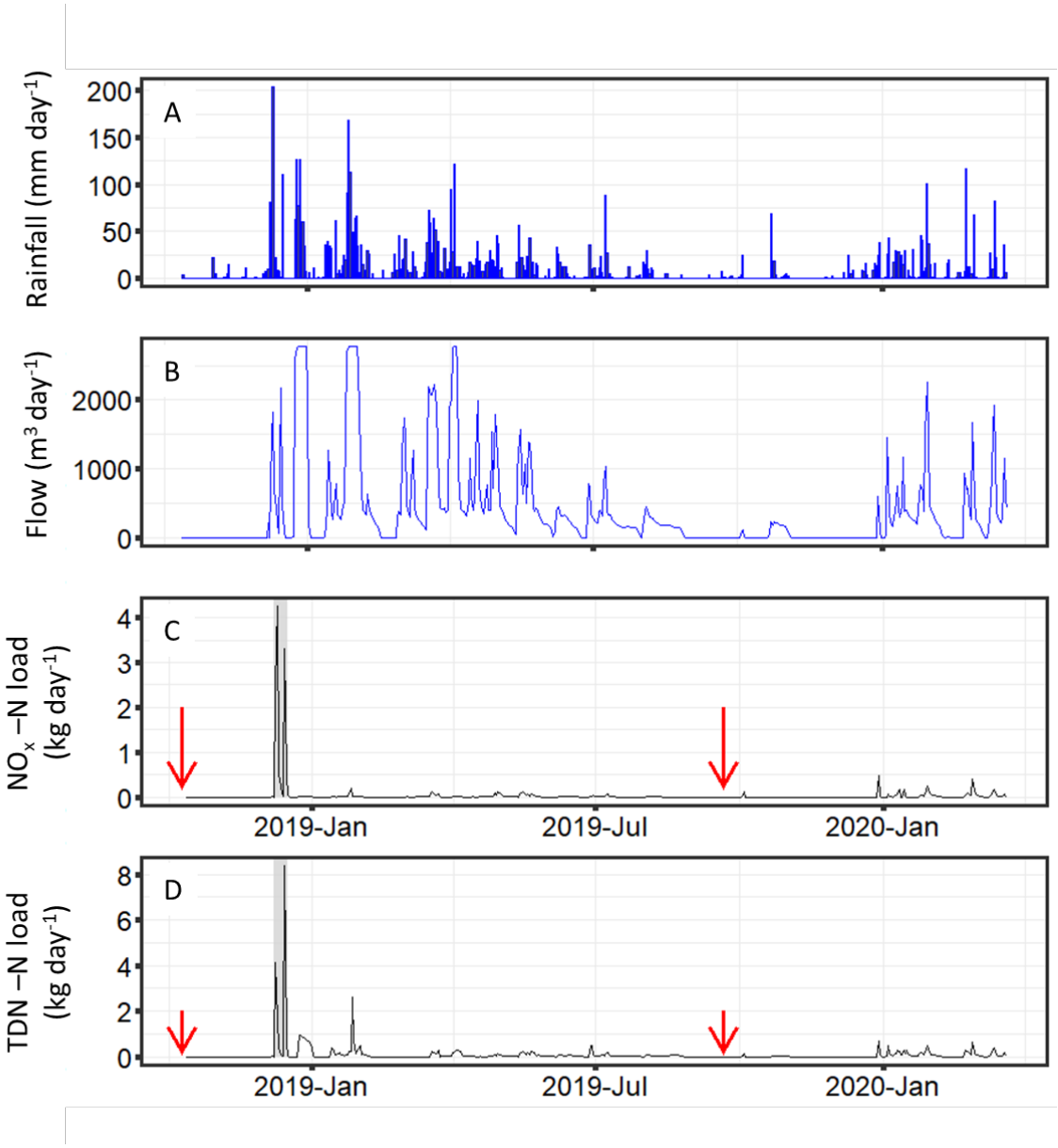

Figure 25: Daily rainfall (A) and, discharge (B), NO_x- N load (C) and TDN load (D) of the drain in which Inline Bed 1 was installed. Red arrows indicate time of fertiliser addition to contributing paddocks and grey shading represents 10-day large 'first-flush event' at the start of the 2018-2019 wet season.

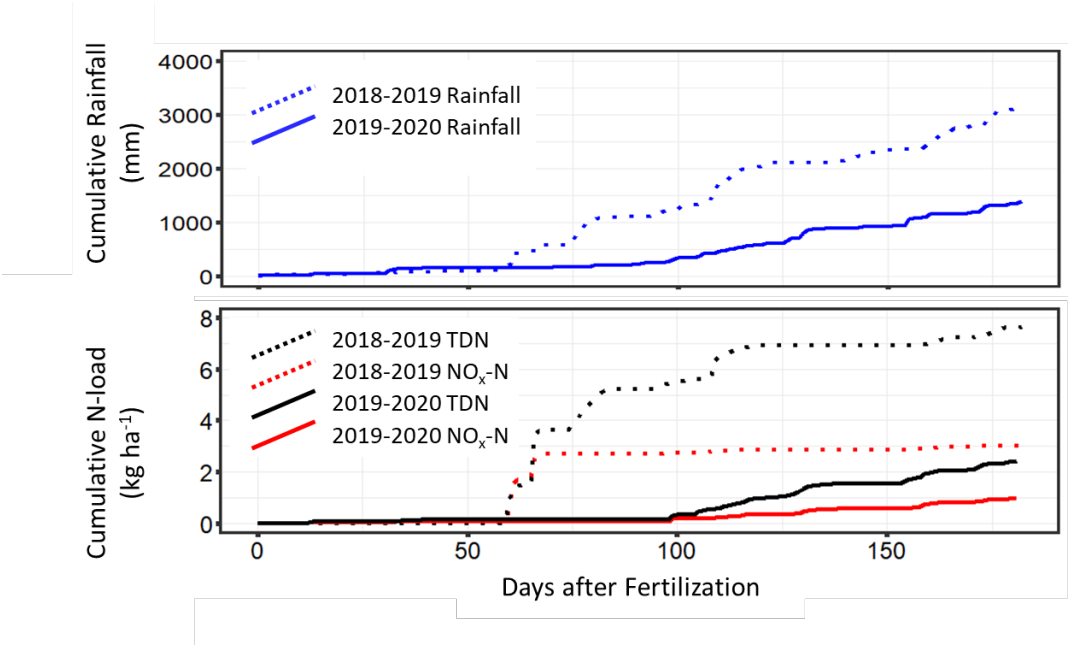

Figure 26 Comparison of cumulative rainfall and NO_x-N and TDN loads leaving paddocks drained by Drain 1 on Farm 1, over the two years of monitoring, plotted from the date of fertiliser application (10/10/2018 in the 2018-2019 year and 21/9/2019 in the 2019-2020 year).

Of the nitrogen load observed in the drains feeding the bioreactor beds, only a relatively small portion was ever directed through the bed. Initial attempts to measure this flow with an inline flow meter (Flomec DP490, Flomec, Sydney, Australia) were unsuccessful due to being below the resolution required for accurate determination. Subsequent direct measurement of bed outflow was carried out on 24 occasions when the bed was found to be flowing and the outflow was accessible. These values were used to derive a linear relationship between flow in the drain to the amount of water passing through the bioreactor bed, and thereby allow for the calculation of flows through the bioreactor bed across the annual hydrograph.

Nitrogen loads entering the bioreactor were calculated by multiplying daily flow into the bed by interpolated daily concentrations, calculated as described above. Flow rates varied from 0 L s^{-1} (when the drain was dry) to 1.76 L s^{-1} under conditions of maximum drain flow, equating to a maximum flow through the bioreactor bed of 152 m^3 day^{-1} and a theoretical minimum residence time of 1.4 h assuming a porosity of the bioreactor woodchips of 0.53. Although this is substantially below the 6-8 h often cited as optimal[31], our work suggests that a more meaningful measure for the effect of residence time is that NO_x-N concentration is reduced by 1 mg L^{-1} h^{-1} spent flowing through a bioreactor (under the generally warm temperatures and with the woodchips used here). This rate of concentration reduction applies once conditions are anaerobic, which varies according to the situation, and holds until very low concentration is reached. This provides an indicative design criteria if concentrations are known – but also highlights the problem for sizing a bioreactor when there are highly variable NO_x-N concentrations.

Over the 2018-2019 year, 3.1 kg TDN, including 1.15 kg NO_x-N, entered Inline Bed 1, representing 7.2 % of the total annual NO_x-N load in the drain at the point where it reached

the bioreactor. The annual amount of nitrogen removed from the drain was calculated by multiplying the amount of NO_x-N entering the bioreactor by the observed removal efficiency, 41% (Figure 16). This resulted in a nitrogen removal rate of 0.47 kg N over the 2018-2019 year, which when considering the 4.33-ha contributing area of the drain, equated to a NO_x-N load reduction of 0.11 kg N ha^{-1} yr^{-1}.

Denitrification efficiency of the bioreactor beds was calculated using two independent approaches. The first used calculated load reductions (as above) annualized over the year to provide an average annual denitrification rate in the woodchips of 0.08 g-N m^{-3} day^{-1}, which increased to 0.10 g-N m^{-3} day^{-1} if considering only the 283 days in which water was flowing through the woodchips. In the second approach, denitrification rates were calculated on a daily time step using interpolated NO_x-N concentrations at the bioreactor inlet and outlet, in conjunction with daily flows. For the 524 days for which data was available, a small number of days (104) were calculated to have a 'negative' denitrification rates (average denitrification -0.047 g N m^{-3} day^{-1}). This is believed to be an artefact due to the fact many [NO_x-N] values were at or below the analytical detection limit. After reassigning negative denitrification rates to 0 we see that daily denitrification rates averaged 0.1± 0.5 g-N m^{-3} day^{-1} (Figure 27), but reached a maximum of 8.7 g-N m^{-3} day^{-1} on two days.

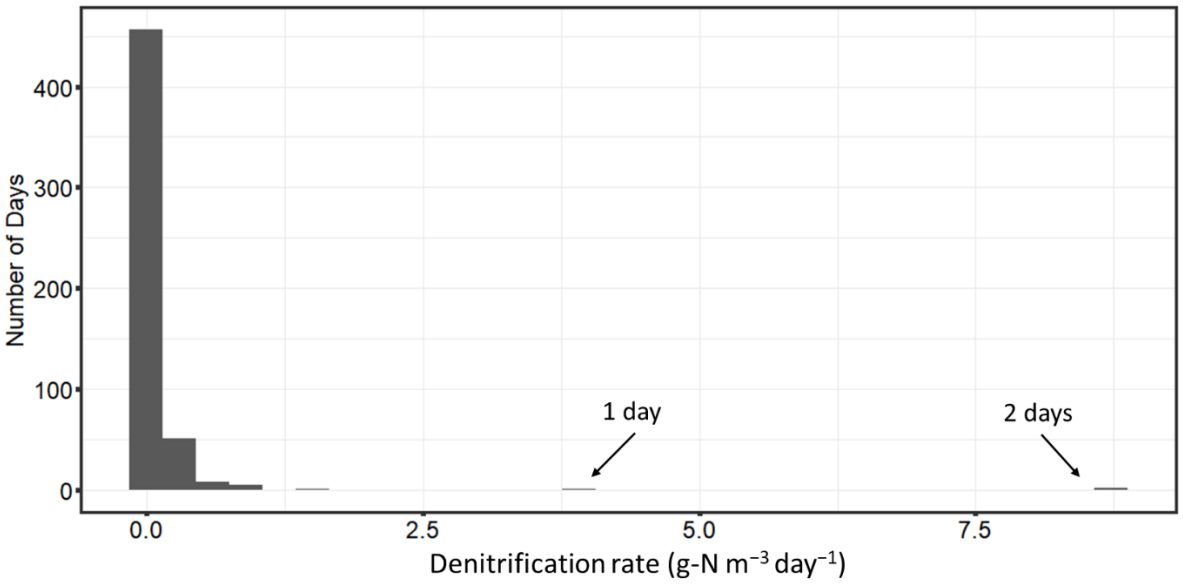

Figure 27: Daily denitrification rates observed in Inline Bioreactor Bed 1.

While maximum denitrification rates observed in the field approached the maximum potential rates observed in laboratory studies (Appendix 5), it is clear that N-limitation during much of the year results in an average rate orders of magnitudes less than that

observed in other bioreactors. It should also be kept in mind that nitrogen removal rates in bioreactors tend to be higher in the first year of operation than in subsequent years.[31]

Bed bioreactor performance – cost-effectiveness

Cost per kg of nitrogen removed is the metric currently used to compare the effectiveness of various approaches for reducing nitrogen loads carried by rivers in the GBR catchments[32]. In the case of bioreactors (and wetlands) it can be estimated as follows:

$$Cost\ per\ kg\ N\ removed = \frac{Cost}{Load \times Interception \times Reduction\ in\ [NO_x{-}N]}$$

Equ.1

Were the 'Cost' is the cost of construction, assuming a discount rate of zero, 'Load' is the NO_x-N load carried by the drain in kg (annual load x life of bioreactor in years), 'Interception' is the proportion of the load routed through the bioreactor, and 'Reduction in [NO_x-N]' is the proportion of the NO_x-N entering the bioreactor that is removed. In this trial, the construction cost was $6,653, load was 160 kg NO_x-N (16.0 kg yr^{-1} x 10-year life), interception was 0.072 (i.e. 7.2%) and the reduction in [NO_x-N] was 0.41 (i.e. 41%). Therefore, the cost of removal is $1,409 kg^{-1} nitrogen. It was assumed that bioreactor life, load, interception and reduction in [NO_x-N] do not change with time. A lifespan of 10 years is a commonly used assumption but there is little data globally and none in the tropics.[31] Bioreactor lifespan and the change in interception and removal efficacy over time must be known for a robust assessment of bioreactor cost-effectiveness to be made.

Cost-effectiveness would be improved (i.e. cost per kg nitrogen reduced) by a lowering of cost. The largest components of cost, in decreasing order of magnitude, were: labour, hire of excavator, and woodchips. All these costs might be reduced in a competitive tender process for large scale implementation. Cost also depends on design, and possible modifications to design are discussed below.

Cost-effectiveness also increases if load increases. For example, if fertiliser rate or loss in runoff was higher (e.g. due to soil type, method and timing of fertiliser application, or rainfall) then cost effectiveness would be greater. In this trial the fertiliser application rate (112 kg N ha^{-1} yr^{-1}) was relatively low for the region. Load, and hence cost effectiveness, would also increase if the contributing area was larger. In that case the bioreactor volume would need to be larger, so cost would also increase, but the cost would probably not increase in proportion to bioreactor volume.

Cost-effectiveness would increase if interception and/or the reduction in [NO_x-N] in intercepted water were increased, but there is a trade-off between them. The reduction in

[31] Addy, K., Gold, A.J., Christianson, L.E., David, M.B., Schipper, L.A., Ratigan, N.A. 2016. Denitrifying Bioreactors for Nitrate Removal: A Meta-Analysis. *Journal of Environmental Quality* 45: 873–881.

[32] Alluvium. 2019. Effective and Efficient Pathways for Investment in Improved Water Quality in the Great Barrier Reef: Final Report, A report for the Great Barrier Reef Foundation, Brisbane.

[NO_x-N] was limited at points when [NO_x-N] was high due to sub-optimal hydraulic residence time, however the only way to increase hydraulic residence time to that required for full bioremediation during these periods (i.e. achieving reduction in [NO_x-N] ≈100%), without increasing size of the bioreactor, is by decreasing flow rate, but this can be achieved (at a given drain flow) only by lowering the proportion of water intercepted. Interception and the reduction of [NO_x-N] might be both increased by design modifications, discussed below, however it is apparent that bed bioreactors as tested show only limited potential under the hydrologic and nitrogen load conditions observed.

It is also apparent that outside periods where [NO_x-N] was high the bioreactor system was substantially N-limited (Figure 27). Therefore, when concentratons were low, the bioreactor could have afforded much higher rates of drain flow interception (and lower hydraulic residence times) .

If the NO_x-N load entering the bioreactor was distributed throughout the period in which the drain is flowing, rather than arriving in a short first flush, then a higher proportion would be removed. This would occur for two reasons. First, there would be more days when NO_x-N concentration of water entering the bioreactor was significant and, hence, more days on which nitrogen was available to be removed. Second, interception is 100% of drain flow up to a threshold (approximately 0.22 L s^{-1} in Inline Bed 1, at which point hydraulic residence time is ~10.6 h), above which it declines in proportion to drain flow. Therefore, during the considerable number of days on which flow is low, the reduction in [NO_x-N] is high due to high hydraulic residence time, i.e. at the same time that interception is also high.

A more temporally distributed NO_x-N load in drains might theoretically be achieved through the use of split fertiliser applications or slow-release fertilisers. However, there is limited opportunity for split application between the earliest possible time (immediately following ratoon crop harvesting or plant crop planting) and the latest possible time (when crop is too large to drive through or soil is too wet). Furthermore, there is relatively little crop growth and nitrogen uptake during this period. Slow-release fertilisers and other enhanced efficiency fertilisers are being examined, but their main purpose is to increase uptake efficiency and thus reduce losses rather than spread the losses over time. Thus, the scope for more even, temporal distribution of load in sugarcane growing areas of the Wet Tropics is low. It is worth noting that the sugarcane cropping system is challenging for bioreactors compared to systems in which losses are inherently distributed more evenly over time, such as those with more evenly distributed rainfall and tile drainage systems.

Pollution swapping

Initial concerns have been raised in the B4GBR network about the potential for denitrifying bioreactors to be a source of pollutants to the environment. The most common concerns raised are dissolved organic carbon (DOC) leaching from the woodchips which can increase biological oxygen demand (BOD) in receiving waters, and the emission of N_2O, a powerful greenhouse gas, if denitrification was found to be incomplete.

However, DOC generation as measured by visible light absorption was negligible throughout the study. This was apart from the initial weeks following installation and the first flushing of

the woodchips. There is, however, the need to consider the impact of an increased chemical oxygen demand (COD) of water leaving the bioreactor. Highly reduced anoxic water from the woodchip bioreactors, produced during periods of low flow rapidly interacts with any oxidised substrate, in the case of the BSDA often iron (III) oxide in the drain floor and walls. The reaction is visible as the colour of the drain floor changes from shades of brown to light grey. This reaction occurs rapidly when the receiving water body is aerobic and relatively large; there was no visible trace of colour change 10 m downstream of Inline Bed 1's outlet. However, when considering the scaling-up of wood chip bioreactors, consideration should be given to the impact of increased COD and implication for the dissolution of minerals.

Emission of N_2O from the bed bioreactors was determined by measuring concentration of dissolved N_2O in the inlet and outlet and multiplying the difference between them by the flow through the bed. All N_2O leaving the experimental beds had to do so in the dissolved phase because the beds were sealed. The measurement of flow is described above. Concentration was measured using headspace-gas chromatography[33].

The concentration of N_2O-N in water passing through the bioreactor bed actually decreased from a mean of 0.42 to 0.24 µg L^{-1} during a period when mean TDN concentration of the water flowing into the bioreactor beds was 4.5 mg L^{-1} (Figure 18). The finding that the bed bioreactors were actually a sink for N_2O is not surprising, given their low redox potential and the likelihood for equilibrium kinetics to drive denitrification to completion, reducing N_2O to N_2.

More detailed examination of N_2O emission was carried out in the column study described in Appendix 4. The highest N_2O emission, achieved under conditions suboptimal for denitrification, was 0.5% of the unrecovered (i.e. denitrified) nitrogen. Similar values have been reported by others[34] [35]. These values are less than the amount of N_2O-N estimated to ultimately arise from NO_3^- leaving farms, irrespective of its pathway; the IPCC assume that 0.75% of the nitrate N in runoff is eventually converted to N_2O[36] [37]. Therefore, there is no evidence that bioreactors would generate more N_2O than would otherwise be emitted.

Bed bioreactors – considerations for future design

Limitations of the tested systems

The effectiveness of bioreactors is determined by the amount of the flow they intercept and the amount of the nitrogen removed from the intercepted water. Interception and the reduction in [NO_x-N] of intercepted water were both low in this trial, due to most of the

[33] Well, R., Myrold, D.D. 1999. Laboratory evaluation of a new method for in situ measurement of denitrification in water-saturated soils. *Soil Biology and Biochemistry* 31(8): 1109-1119.
[34] Elgood, Z., Robertson, W.D., Schiff, S.L., Elgood, R. 2010. Nitrate removal and greenhouse gas production in a stream-bed denitrifying bioreactor. *Ecological Engineering*, 36, pp.1575-1580.
[35] Moorman, T., Parkin, T., Kaspar, T. and Jaynes, D. (2010). Denitrification activity, wood loss, and N2O emissions over 9 years from a wood chip bioreactor. *Ecological Engineering*, 36(11), pp.1567-1574.
[36] Mosier, A., Kroeze, C., Nevison, C., Oenema, O., Seitzinger, S., van Cleemput, O. (1998). Closing the global N_2O budget: nitrous oxide emissions through the agricultural nitrogen cycle. *Nutrient Cycling in Agroecosystems* 52, pp.225-248.
[37] *Nitrous oxide emissions from Bioreactor, crops and waterways*. Bioreactor network factsheet Nov 2019. QUT and Queensland Govt.

NO_x-N load occurring over short high-flow 'first flush' periods, while for large periods of time when flow was low (and therefore interception was high) there were generally low NO_x-N concentrations.. This section considers possible improvements to bioreactor beds in or adjacent to, farm drains, and another option for increasing denitrification through management of the drain systems themselves. In subsequent sections, we consider the possibility of much larger offline beds associated with larger drains, and the modification of the design of bioreactor walls to include 'hybrid' bed-wall systems.

A key design criterion arising from this work is that NO_x-N concentration is reduced by approximately 1 mg L^{-1} for each hour of residence time in the bioreactor, under non-nitrate-limited conditions (see 'Modelling bioreactor wall efficacy' for derivation). Therefore, to optimise NO_x-N removal, residence time would ideally be either a) always sufficiently high to deal with the highest [NO_x-N] reached, or b) varied to match the [NO_x-N] observed. For example, at the maximum [NO_x-N] observed in Farm 1 drain 1 of 4.5 mg L^{-1}, a residence time of 4.5 h would be required for full bioremediation, yet during most of the year, when [NO_x-N] <0.5 mg L^{-1}, residence time could be as low as 30 min and still be sufficient for full bioremediation. Although it is worth noting, there is a likely minimum residence time required to first achieve anaerobic conditions in the bioreactor before denitrification can occur and so a minimum residence time of ~1 h is more likely required.

Given the passive control of flow through the tested bioreactor beds, hydraulic residence time varied greatly, from periods of high flow where it may have been as low as 1.4 h, through periods of full interception of the drain flow (at a flow rate of 0.22 L s^{-1}) where it was 10.6 h, to very low-flow conditions where hydraulic residence time would have been far longer. It is therefore likely that, during periods of high flow (which during 'first flush' coincide with high [NO_x-N]), residence time was below optimal for achieving full NO_x-N removal, whereas for most of the year (with low flow and low [NO_x-N]), residence time was longer than necessary.

Design modifications to alter hydraulic residence time will by necessity also alter interception, i.e. as hydraulic residence time goes down the amount of water intercepted goes up. Any passive design modification that increases efficacy during the first-flush period will result in a bioreactor that is massively over-designed for most of the year. However, the use of controlled drainage may allow residence time to be increased during the important first-flush period.

Increasing flow interception

The volume of flow intercepted by a given volume of bioreactor material might be increased by increasing the hydraulic head across the system or the cross-sectional area of the bioreactor relative to its length. The former might be achieved by lowering the outlet level relative to the inlet and the latter by having a shorter, wider bioreactor bed. However, in both cases, physical limitations of the drain systems in the BSDA and elsewhere make this challenging.

Interception might also be increased by reducing biological fouling and clogging of inlet and outlet structures, which restricted flow rates through some of the beds. Specifically, the

inlet of the offline bed clogged frequently, due mainly to algal growth, but also floating sugarcane trash. However, different intake structures could also help, e.g. ensuring shaded conditions to reduce algal growth. To reiterate though, any increase in interception must be balanced against considerations of reduced hydraulic residence time or the requirement for a larger bioreactor volume. It should be kept in mind that the improvements to bioreactor beds suggested here are unlikely to result in large improvements to the amount of nitrogen removed, because of the substantial environmental limitations that remain.

'Controlled drainage'

Another option for using the drainage system to remove nitrogen from runoff water is the consideration of modified drain management including 'controlled drainage'. Controlled drainage involves slowing the flow of water through the drain system at key times and places in order to raise the water table in the paddock and thereby enhance denitrification in the soils and drains[38]. It is widely used in the USA where it has provided an average net decrease in DIN loads of approximately 30% at a cost of US$2.10 ± 1.53 per kg N [39 40 41 42]. Drainage can be controlled in open drain systems by installing control structures in key drain locations and controlling flow by inserting or removing riser boards. The idea would be to partially hold up water in key drains during the dry season so that the first flush of runoff is held back, raising the water level in the drains and soils to approximately 600 mm below ground level. This level does not restrict sugarcane growth, but prevents a large volume of water leaving the field, allowing denitrification to occur during the time when nitrate concentrations are at their maximum. After several weeks, or when rainfall and soil moisture become excessive, the gates would be opened and the drain system allowed to operate at full capacity. Research has shown that with proper management, controlled drainage systems can also conserve water in the soil profile and alleviate drought stress, which may result in increased crop yields[43].

Controlled drainage could also enhance the efficacy of inline bioreactor beds. A gate installed just upstream of an inline bed during the first-flush period would reduce and delay the flow of water to that leaking past or overtopping the gate. Therefore, both interception and residence time could increase during this critical time.

[38] Poole, C., Burchell, M. Youssef, M., 2018. Controlled Drainage- An important Practice to Protect Water Quality That Can Enhance Crop Yields. https://content.ces.ncsu.edu/controlled-drainage

[39] Christianson, L.E., Frankenberger, J., Hay, C., Helmers, M.J., Sands, G. 2016. Ten Ways to Reduce Nitrogen Loads from Drained Cropland in the Midwest. Pub. C1400, University of Illinois Extension.

[40] Skaggs, R.W., Breve, M.A., Gilliam, J.W. 1994. Hydrologic and water quality impacts of agricultural drainage. Critical Reviews in Environ. Sci. and Tech. 24: 1-32.

[41] Woli, K. P., David, M.B., Cooke, R.A., Mclsaac, G.F., Mitchell, C.A. 2010. Nitrogen balance in and export from agricultural fields associated with controlled drainage systems and denitrifying bioreactors. Ecological Engineering 36: 1558- 1566.

[42] Saadat, S., L. Bowling, J. Frankenberger and E. Kladivko. 2018. Nitrate and phosphorus transport through subsurface drains under free and controlled drainage. Water Research. 142:196-207.

[43] Poole, C. A., R. W. Skaggs, G. M. Cheschier, M A. Youssef, and C. R. Crozier. 2013. Effects of drainage water management on crop yields in North Carolina. Journal of Soil and Water Conservation, 68: 429-437

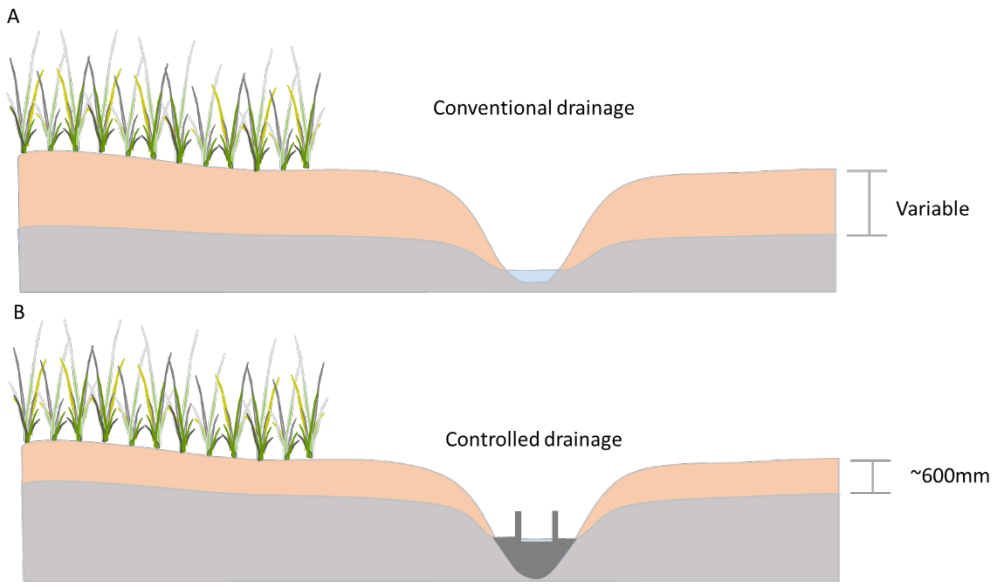

Figure 28: Conventional (A) as compared to controlled (B) drainage. Under controlled drainage water levels in drains are regulated through the use of adjustable or removable control gates. This allows farmers to maintain the water table in paddocks below the main sugarcane rooting zone but high enough to promote in-situ denitrification, especially at the start of the wet season when the first flush of nitrogen loss is expected.

Efficacy of Bioreactor Beds in Landscape Context

The likely effectiveness of installing bed bioreactors across the BSDA catchment was established through development of a hydrological model of the drainage network to determine the location and number of bioreactors feasible to implement across the landscape. The reduction in [NO_x-N] data from the trial bioreactors could then be applied to estimate interception of total nitrogen load generated from the BSDA catchment.[44]

The hydrological model was delineated from a digital elevation model (DEM) derived from LiDAR imagery at a resolution of 1m by 1m. The flat topography of the area necessitated 'burning in' some drains to ensure connectivity of flows towards the Christiano Access monitoring location. Stream or drain burning was done using the ArcHydro tool in ArcGIS. ArcSWAT was then used to identify potential locations of bed bioreactors from an analysis of the terrain and drainage network.[45]

Two criteria were used to determine suitable bioreactor locations across the drainage network. The first criterion required the end of a drain to have an elevation drop of 0.5 m or more into the receiving drain, which was deemed as sufficient head differential to enable installation of an effective bed bioreactor. This was on the basis that, for bed bioreactors to work, whether inline or offline, a head is required to drive water through the bed. In this flat landscape, sufficient head is practically available only near the end of drains that terminate above the level of the larger receiving drain. Otherwise, sufficient head would be available only with long, large-diameter inlet or outlet pipes, which would inordinately increase

[44] The area excludes the 433.5 ha in the upper area of the Niringa Creek catchment.
[45] ArcSWAT is the ArcGIS extension or interface for the Soil and Water Assessment Tool and was run using ArcGIS version 10.5. Details of this model can be found at this website: https://swat.tamu.edu/software/arcswat/

disturbance and cost. The 0.5-m drop in elevation was determined from the DEM. The second criterion was a contributing area >2 ha, as this resolution was fine enough to identify the drain outlets in this flat terrain.

Based on these two criteria, 127 sites were deemed suitable. These had a combined contributing area of 733 ha, comprising 21% of the BSDA. Of the sites, 117 were single drains and the remainder were higher-order drains (Figure 29).

If bioreactors were to be installed at all 127 sites, the amount of nitrogen removed was calculated at 81 kg yr^{-1} (Figure 30). This amount took account of the size of the contributing area, i.e. the larger the contributing area for a given bioreactor, the more nitrogen will be intercepted, assuming the size of the bioreactor was scaled accordingly. Sites were ordered in priority from largest to smallest contributing area. For example, the 1st to 4th bioreactors each have contributing areas >20 ha and the 57th to 60th bioreactors 4.3 ha, similar to the trial for Inline Bed 1. Their corresponding removal rates are 12 kg N yr^{-1} (mean of 3.0 kg yr^{-1} each) and 1.9 kg N yr^{-1} (mean of 0.47 kg yr^{-1} each), respectively once fully operational.

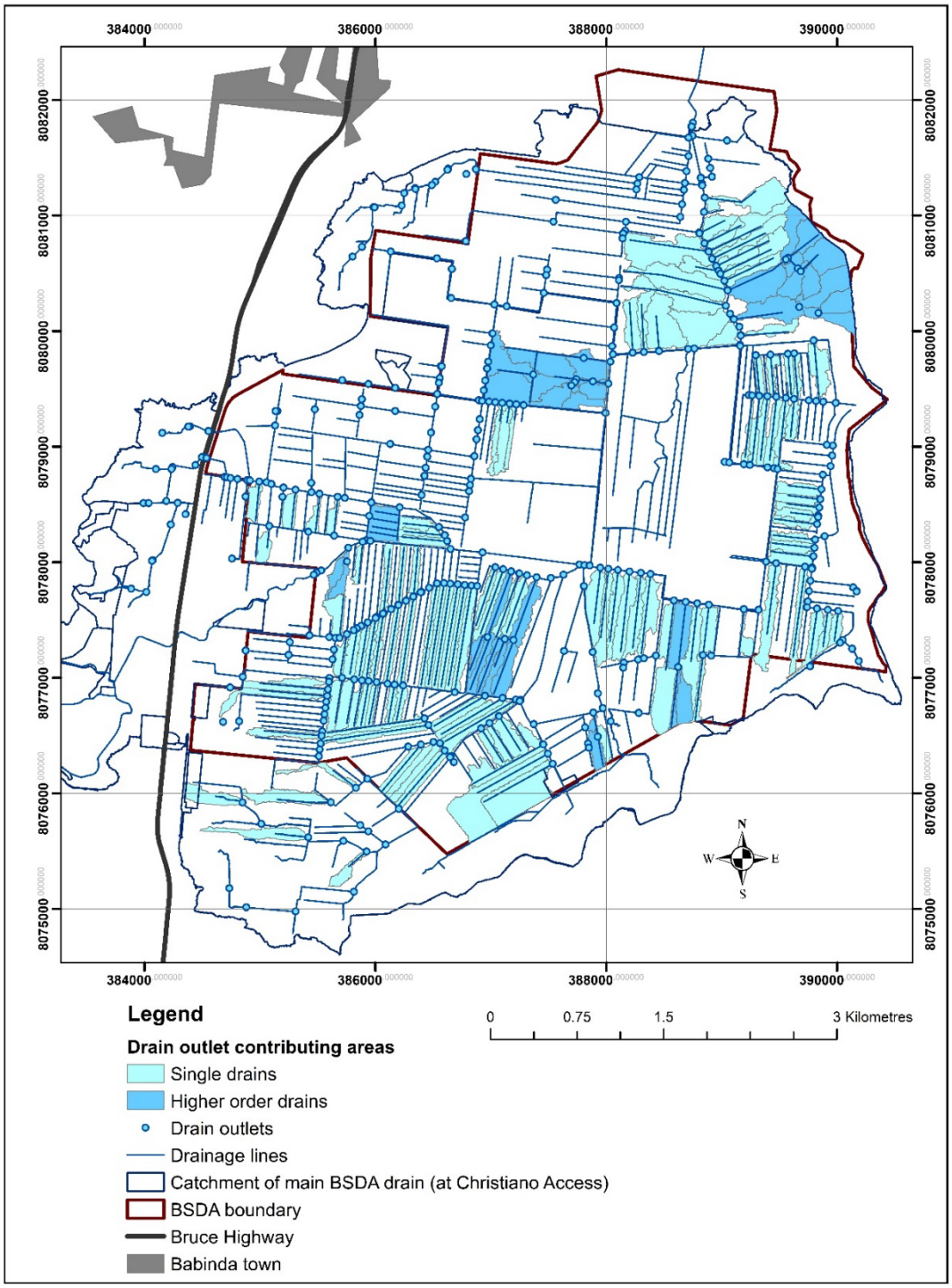

Figure 29. Potential sites for bed bioreactors in the Babinda Swamp Drainage Area (BSDA) and its catchment. Drain outlets with contributing area >2 ha, identified using ArcSWAT, are shown. Drain outlets with a drop of 0.5 m or more into the receiving drain are potential sites for bed bioreactors and their contributing areas are shown.

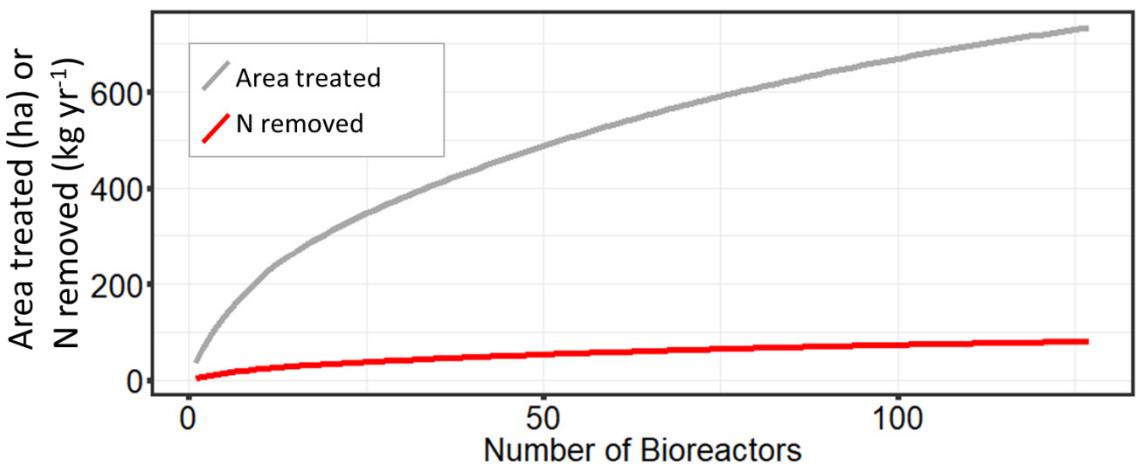

Figure 30: Cumulative area treated and estimated nitrogen removal if bed bioreactors were installed at the sites shown in Figure 29, in decreasing order of contributing areas.

Since nitrogen interception in the bed bioreactors was limited for much of the year, by both a lack of water and NO_x-N, likely nitrogen removal rates from multiple paddock-scale bioreactor beds across the landscape were compared with one 'large bioreactor bed' connected to a large drain (assuming a suitable location were found in the landscape), and using the nitrogen concentrations measured at Christiano Access. This was done using an exploratory model, and was considered useful as flow is more consistent in larger drains, with water being present at all times.

In modelling flows into a large bioreactor, the assumed hydraulic conditions (i.e. hydraulic head and hydraulic conductivity of woodchips) were the same as those for maximum flow in the paddock-scale bioreactor beds, which achieved flow rates of 1.76 L s^{-1} through a 16 x 1 x 1 m bed. While these factors were held constant in the model, it was assumed that any large bed would be built as a large square of 1 m depth (Figure 31). This was modelled up to a volume of woodchips equal to that required for the 127 identified paddock-scale sites (e.g. 2710 m^3, representing a square 52 x 52 m). Water was assumed to flow into one edge of the large bioreactor bed and flow out of the opposite edge. Other configurations would be possible, and would change the efficacy, but this one was used because it was simple and readily conceivable.

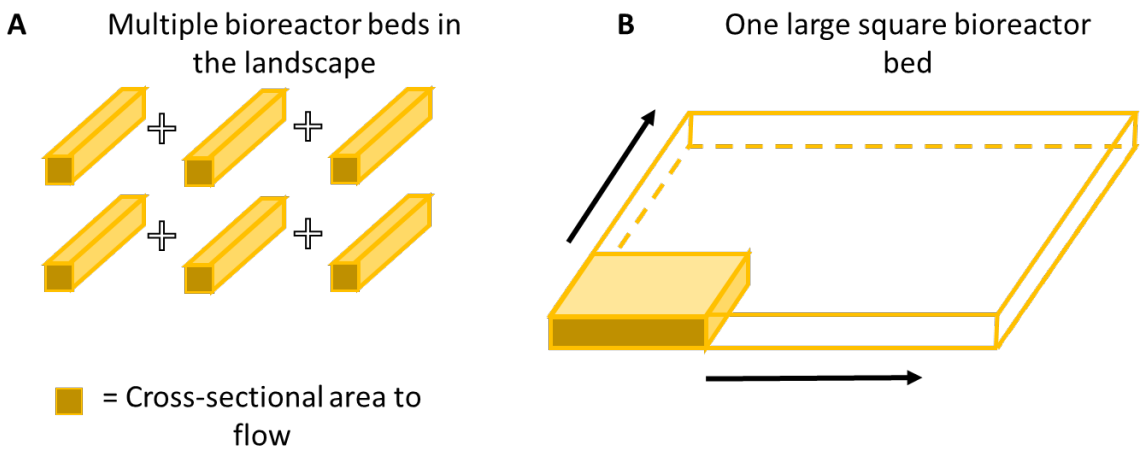

Figure 31: Configuration and dimensions of multiple paddock-scale (inline or offline) bioreactor beds with one large bioreactor bed.

The flux of NO_x-N into the large bed was calculated by multiplying the daily flow capacity through the bed or the measured flow at Christiano Access, which ever was less, by the daily NO_x-N concentrations at Christiano Access, interpolated from the measured values.

Nitrogen removal in the large bioreactor was then calculated by multiplying NO_x-N influx by the 41% reduction in [NO_x-N] measured in the paddock-scale bioreactor beds. In the modelled scenario, the large bioreactor bed removed more nitrogen than the multiple paddock-scale beds, for any given volume of woodchips (Figure 32). It should also be noted that the cost per kg N removed is likely to be less in such a large scale bed installation.

The difference in N removal occurred primarily because the large bioreactor was engaged for more days of the year than the smaller ones, and because NO_x-N load is spread out over a longer period in larger drains than in individual paddock-scale drains. For example, taking the volume of woodchips required for the 4 largest paddock-scale bioreactors (i.e. 412 m^3), one large square bioreactor bed (i.e. 20 x 20 x 1 m) is estimated to remove 72 kg N yr^{-1}, compared with 12 kg N yr^{-1}.

However, at larger woodchip volumes, the amount of nitrogen removed per increment of woodchip volume (slope of lines in Figure 32) approaches the same value for both scenarios, as the increase in hydraulic constraint of the large bed counters the effect of continuous engagement. These limitations may be countered by engineering or design (e.g. increasing hydraulic head) and model outputs should not be considered accurate but merely indicative of possible limitations when comparing paddock-scale and large bioreactors with continuous flow. Although this exploratory modelling suggested that substituting one large bioreactor bed for multiple paddock-scale beds would increase nitrogen removal and cost-effectiveness, the assumptions underlying this approach would need to be tested and design options explored before such an approach was considered.

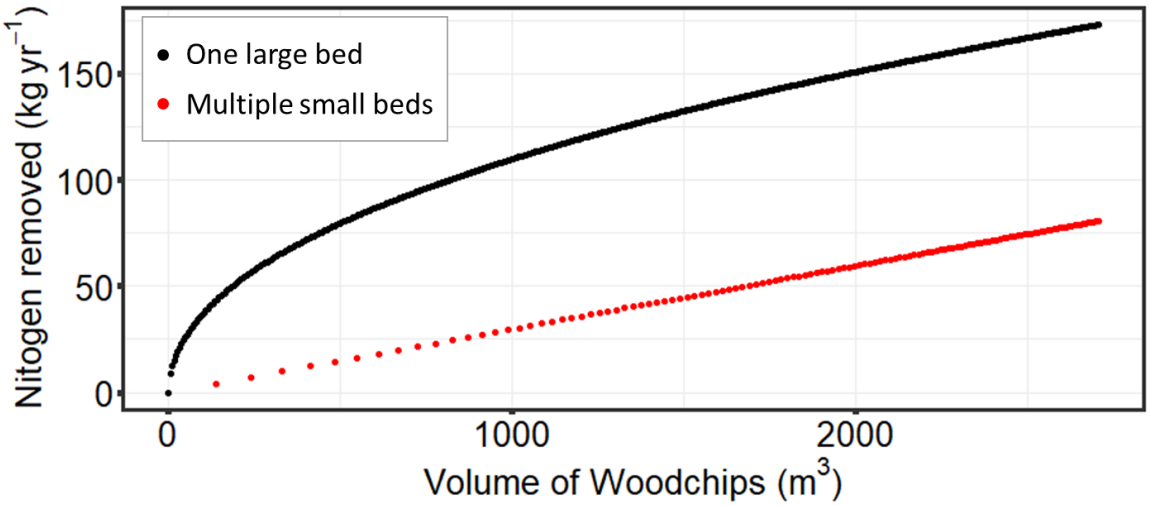

Figure 32: Comparison of N removal by denitrifying bioreactor beds in the landscape, comparing multiple paddock-scale beds with one large square 1-m deep bed having the same total volume of woodchips.

Efficacy of Denitrifying Wall Bioreactor

A denitrifying bioreactor wall was installed on 'Farm 3', near the township of Bellenden Ker, within the Russell River catchment but outside the BSDA. The site was selected following detailed appraisal of prospective sites at several locations within the BSDA, each of which was found unsuitable due to the soil types having high denitrification potential.

Design and installation

The wall was installed along the bottom edge of a sugarcane paddock, parallel to a perennial stream, on 31/10/2018. The soil was sandy loam to approximately 1.2 m depth, below which it was clay (Figure 33). The bioreactor wall was 48.1 m long, 0.65 m wide and 1.09 m high (34.0 m^3 volume), covered in 0.5 m of soil (Figure 33). During installation an agricultural pipe drain was struck. It was broken so water flowing through it would flow into the wall.

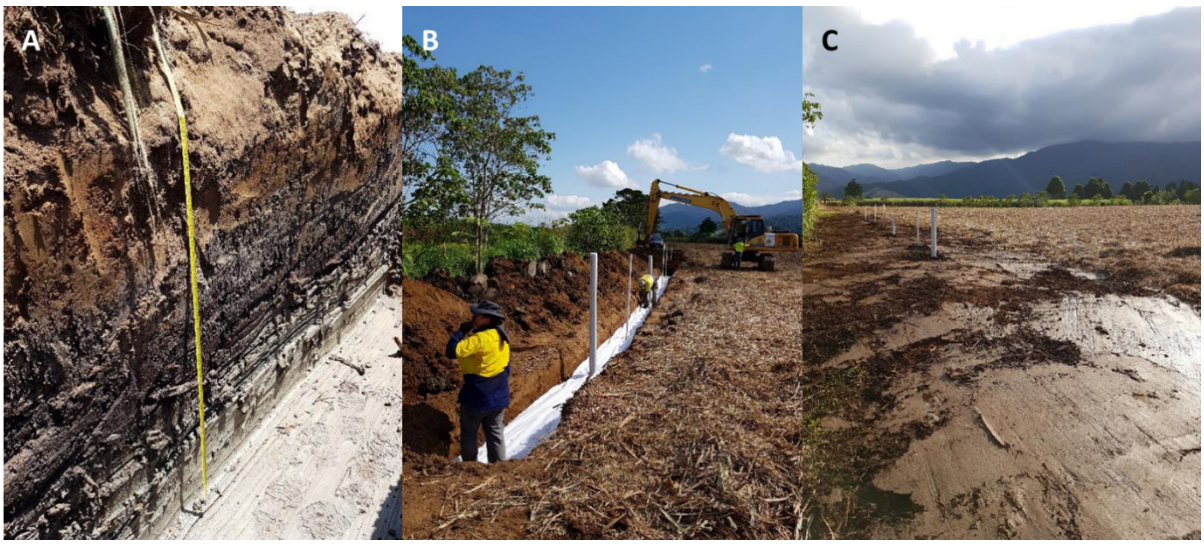

Figure 33. Installation of the bioreactor wall, showing (A) profile of sandy soil over clay in the trench, (B) preparation for returning the soil cap over the woodchips, which are covered in geofabric, and (C) completed wall, with piezometers protruding.

Installation costs

Given our findings on required wall width (see below) it is clear that the 0.65-m wide wall as originally installed at our site was wider than that required to remove all the NO_x-N from water flowing through it. We therefore obtained a commercial quote for a modified design based upon standard excavation of a 2-m deep trench, 0.45 m wide, containing 1.5 m of woodchip and 0.5-m cover of topsoil. Using local suppliers and contractors, and including the full cost of constructing a 32-m length of wall over an eight-hour day, this was found to be \$133 m^{-1} (Table 6)

Table 6. Cost of constructing the bioreactor wall.

Activity	Description	Unit	Price	Total $
Construction materials				
Woodchip supply (m3)	Softwood (local supply)	21.6	22	475
Excavator hire	Bioreactor construction	8	115	920
Dirty water pump hire	Remove excess water	1	100	100
Geofabric	50m x 1m	1	60	60
Marker	Can	1	10	10
			Sub-total	**1565**
Project management				
Designer (8 hour day)	1 x planning, 1 x construction	16	120	1920
General labour (8 hour day)	0.5 x planning, 1 x construction	12	45	540
Mileage	0.72 cents p km	340	0.72	245
			Sub-total	**2705**
Equipment needs				
Measuring tape	50m			0
Star pickets	1.5m			0
Mallet				0
Shovels	Fine clean drain & woodchip			0
Rakes	Level woodchip			0
			Total	**4270**
			per m	**133**

Wall bioreactor performance

Shallow piezometers (~1.8 m deep) were installed at three locations in the contributing paddock and both within and close to the finished wall (Figure 34). Water levels in all piezometers were routinely monitored by hand, with depth to water measurements being taken with a water level indicator (Heron Water Tape, Heron Instruments Inc., Ontario, Canada) with reference to piezometer top of casings (TOC), which were accurately located using a RTK GPS (Trimble R8 GNSS). Water table level was monitored continuously in several of the piezometers (F1, F2, F3, B2, and B4) using pressure transducers (Hobo, U20L-02, Onset, Bourne, MA, USA). The readings were converted to elevation by correcting for air pressure, measured continuously in a nearby (~50 m away) pump shed and corrected for surveyed ground elevation. Readings were cross-validated with the manual measurements.

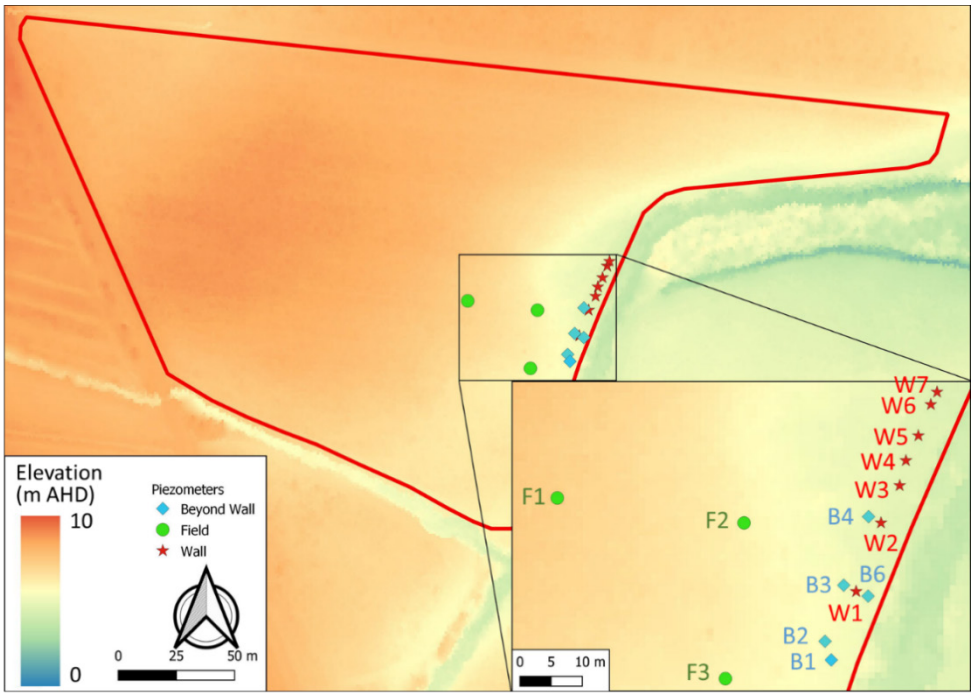

Figure 34: Topography of the area surrounding the bioreactor wall site (from 1x1m DEM), and location of piezometers. Elevation data was sourced from the Department of Natural Resources and Mines. CRS: EPSG:28355 - GDA94 / MGA zone 55. URL: https://elevation.fsdf.org.au/. Tile id: Cairns_2010_Prj_SW_385000_8090000 & Cairns_2010_Prj_SW_386000_8090000

Table 7: Characteristics of piezometers installed in the vicinity of the bioreactor wall.

Piezometer*	TOC** (m AHD)	Ground (m AHD)	Depth (m)
W1	3.87	3.95	1.714
W2	3.63	3.72	1.645
W3	3.68	3.75	1.650
W4	3.59	3.66	1.579
W5	3.58	3.66	1.770
W6	3.45	3.53	1.687
W7	3.55	3.63	1.777
B1	4.11	4.19	1.693
B2	4.09	4.22	1.475
B3	3.90	4.00	1.690
B4	3.68	3.74	2.511
B6	3.41	3.47	1.613
F1	5.18	5.23	2.164
F2	4.43	4.49	1.912
F3	4.51	4.57	2.169
SM	3.62	NA	NA

* Piezometer W# = installed in denitrifying wall, B# =installed in close proximity to wall, F#= installed in field, SM = Survey marker

** Top of casing

The water table was highly dynamic, with for example F2 seeing a fluctuation of 1.6 m between the dry and wet season. However, differences in water table level across the paddock were limited, with all piezometers moving in concert (Figure 35). The maximum hydraulic head observed between the paddock piezometers and bioreactor wall was 0.0226 m m^{-1} (1.12 m fall over 49.4 m) when considering F1, or 0.0175 m m^{-1} (0.30 m fall over 17.1 m) when considering F2 and F3.

After establishment, and once the bioreactor wall has settled, 19 periodic measurement of groundwater quality across the paddock and within the bioreactor wall were made between 2/5/2019 and 20/3/2020 (Figure 36). Sampled piezometers were initially purged for 5 well volumes (or until dry) and allowed to equilibrate for 24 h before sampling. Samples were then collected (using a bailer or 12V inline pump), filtered (0.45 µm, PES) and frozen prior to analysis for TDN and NO_x-N. Water quality within paddock piezometers showed a high degree of temporal variation with high concentrations (up to a maximum of 5.1 mg NO_x-N L^{-1} on 1/02/2020) coincident with the return of rains and a high water table after harvesting and fertiliser application in the paddock.

Across the 19 sampling events groundwater collected from field piezometers contained an average of 0.81 ± 1.03 mg L^{-1} TDN and 0.59 ± 1.00 mg L^{-1} NO_x-N, whereas water collected within the bioreactor wall contained significantly less nitrogen with just 0.24 ± 0.17 mg L^{-1} TDN and negligible concentrations of NO_x-N (Figure 37).

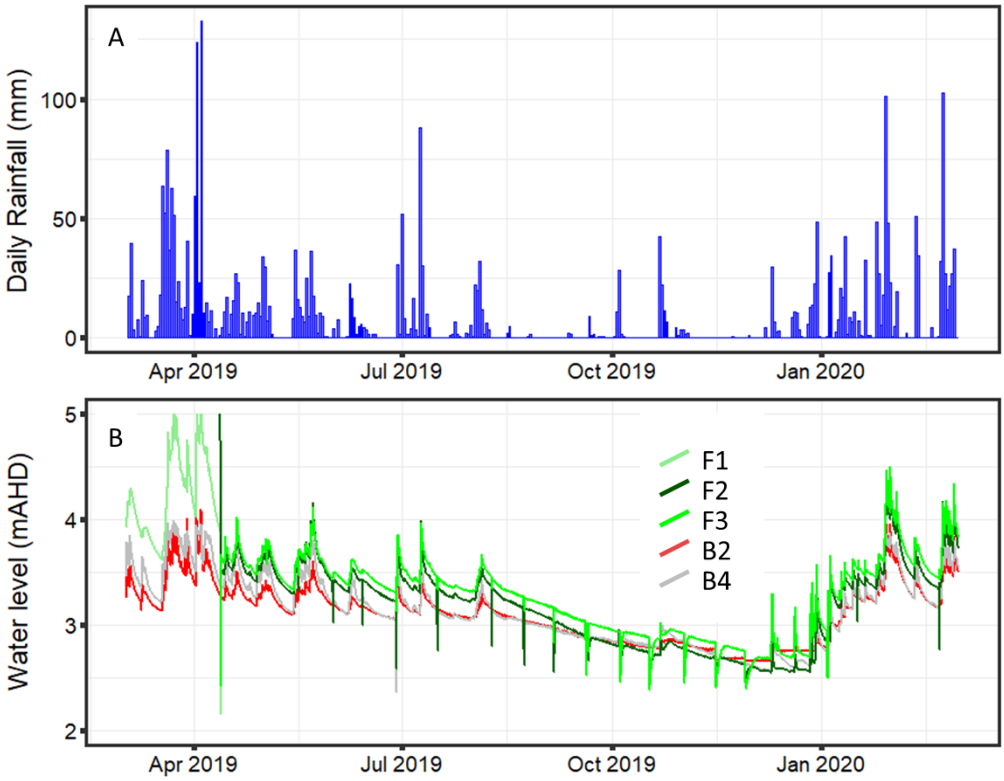

Figure 35: Daily rainfall[46] A) and calibrated water levels in continuously monitored piezometers in the vicinity of the bioreactor wall. Field Piezometer (F1) was discontinued on 20/4/2019 and redeployed as F3. Periodic excursions in water depth were associated with piezometer purging and water sampling events

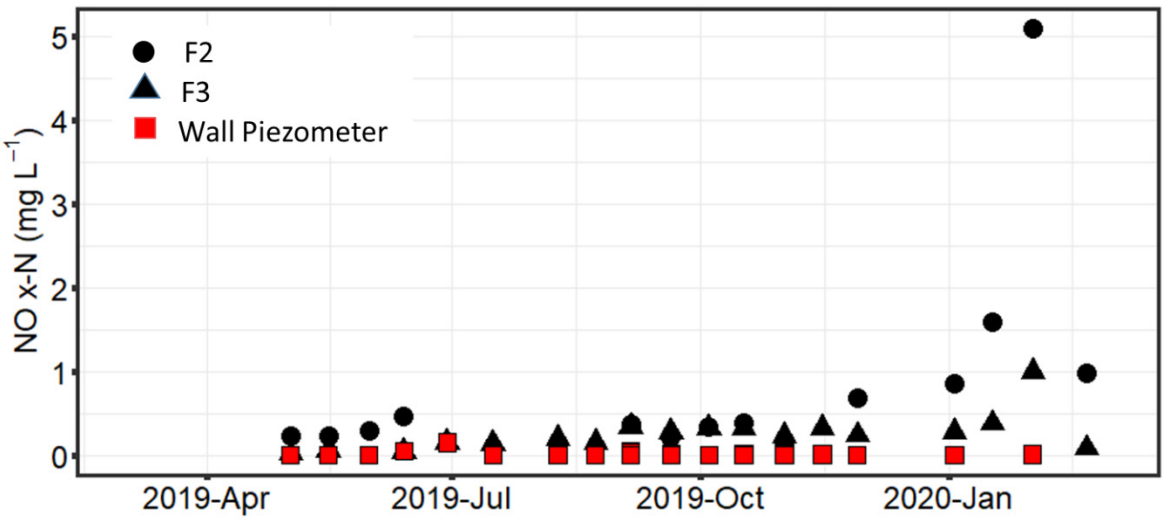

Figure 36: Time series of NO_x-N concentrations in shallow groundwater collected in piezometers across the adjoining paddock (F2, F3) and within the bioreactor wall (W1, W2, W3).

[46] Rainfall data taken from SILO Long Paddock Data Drill for Lat. -17.25° Long 145.95°.

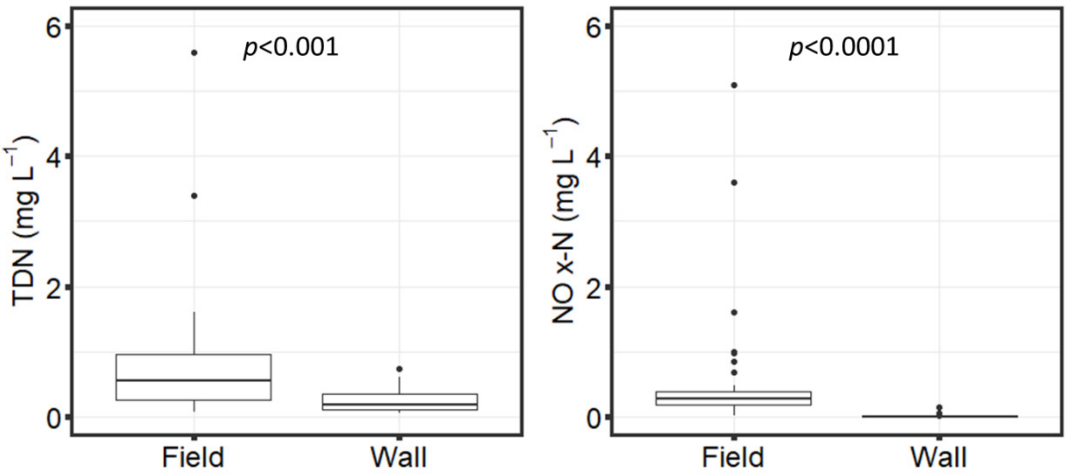

Figure 37: Water quality over 19 sampling events taken both across the paddock (F2 &F3) and within the bioreactor wall (W1,W2, W3). Horizontal line in the box represents the median, bottom and top of the box represent the25th and 75th percentile percentiles, whiskers represent 1.5 x IQR and 'outliers' beyond 1.5 x IQR are given as individual points.

Electromagnetic induction (EMI) survey

A survey of apparent electrical conductivity (EC_a) of the paddock upslope of the bioreactor wall was conducted during both wet season (16/05/2019) and dry season (16/12/2019) conditions to inform our understanding of how groundwater moved in the vicinity of the bioreactor wall.

The EMI surveys were undertaken by David Morrison and Neil Enderlin of DNRME using a DUALEM-21s[47]. The meter has four sensors, referred to as PRP1.0, PRP2.0, HCP1.0 and HCP2.0, arrayed to measure EC_a to nominal depths of 0.5, 1.0, 1.6 and 3.2 m. The instrument has a built-in GPS receiver to record the location of the EC_a readings, and point measurements the four sensors were recorded simultaneously at a frequency of 1 reading per second. The actual or effective depth of readings is reduced by the instrument's height above the ground surface.

At the Farm 3 wall site, the DUALEM 21s meter was mounted on a non-conductive trailer constructed from PVC and towed behind a light all-terrain vehicle (ATV). The transmission (front) end of the instrument was set at a distance of 2.4 m from the ATV to avoid interference from its metallic mass. Towing speed was 10-18 km h^{-1}. Traverses had a spacing of every 3rd plant row. The EMI trailer straddled the plant row with its wheels in the furrows either side. This resulted in the DUALEM instrument recording readings about 10 cm above the ground surface of the plant row. This paddock (~4.3 ha) could be traversed this way as the sugarcane was recently harvested and plant height was quite low.

[47] http://www.dualem.com/products/#2S

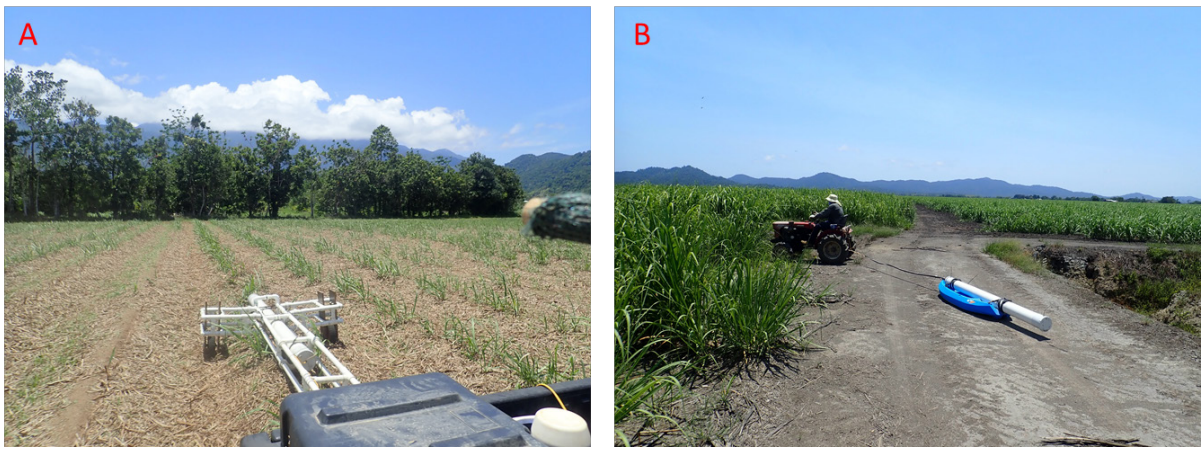

Figure 38: DUALEM 21s being deployed in sugarcane paddocks to measure apparent electrical conductivity (ECa) in sugarcane paddocks in the vicinity of denitrifying bioreactors.

The EC_a measurements and concurrent GPS locations for each data point were logged. Following each survey, the data was cleaned of outliers and extraneous points, such as non-essential measurements taken outside the survey area or random negative readings, to minimise skewing the interpolation of the readings. The cleaned EC_a point data set for each EMI sensor depth was converted into an ESRI shapefile using ArcMap Geographic Information System (GIS) v10.5.1. The points were subsequently interpolated into raster maps for the four depths using the Natural Neighbour approach in ArcMap.

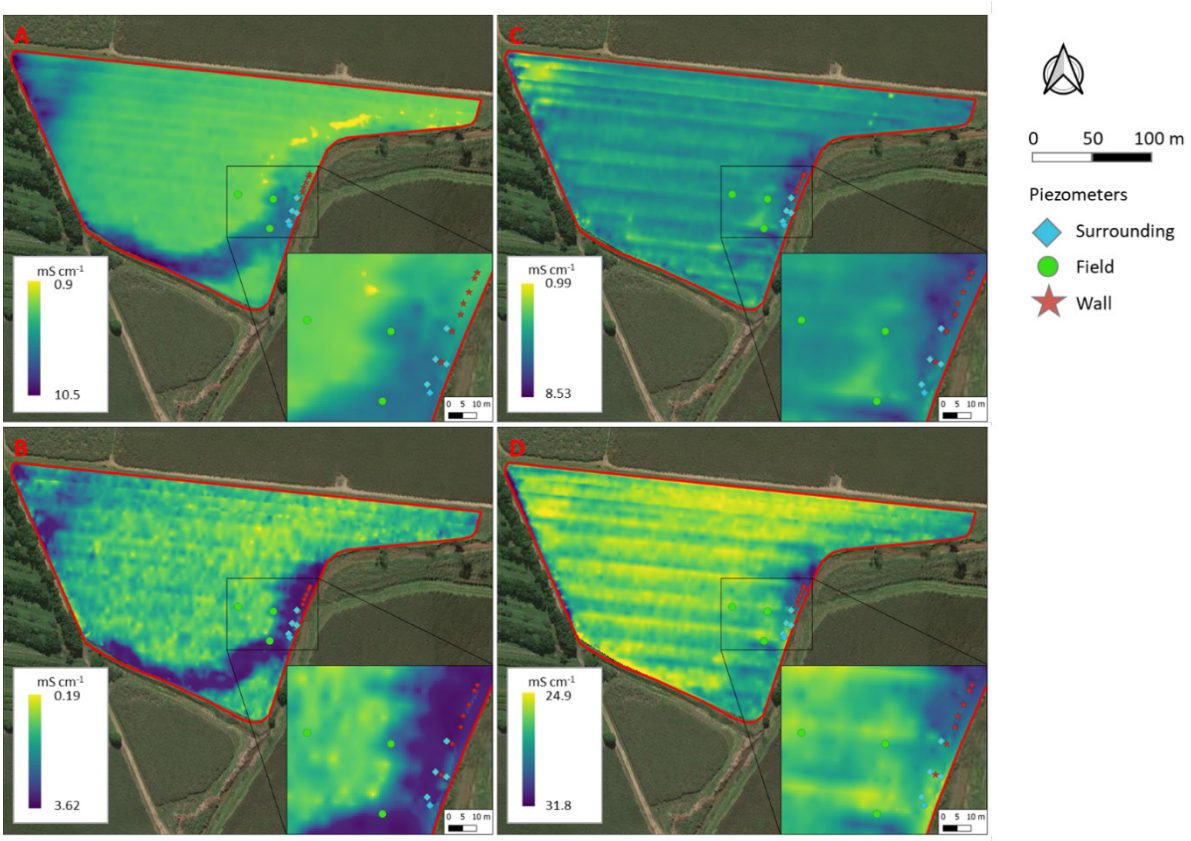

Figure 39. Apparent electrical conductivity of paddock contributing to the bioreactor wall, taken at the end of the dry season, on 16/12/2019. Subplots represent the apparent depths of A) 0.4 m, B) 0.9 m, C) 1.5 m and D) 3.0 m below the surface.

The EMI survey indicated a conductive ribbon of soil, approximately 10 m wide, meandering from the southern to western boundary of the paddock, where it met the bioreactor wall (Figure 39). It increased with depth from south to west, with the top being approximately 0.8 m deep around the vicinity of the wall. Those measurements correspond with the clay layer encountered in the trench dug for the wall (Appendix 2; Figure 33); the clay would have been wetter and, hence, more conductive than the overlying sandy soil during the dry conditions of the survey.

Wall bioreactor – efficacy modelling

Examination of hydraulic gradients (Figure 35) and water quality data (Figure 37) clearly demonstrate that the bioreactor wall as installed was successful in removing nitrogen, and specifically NO_x-N from groundwater that was passing into it. However, it is also apparent that site-specific installation details (e.g. wall dimensions, location, surface topography, soil variability and existing infrastructure such as mole and ag drains) would preclude the extension of a site-specific dynamic ground-water model to predict or assess performance of denitrifying bioreactor walls at other locations across the Wet Tropics.

Therefore, to determine the range of conditions under which a denitrification walls could be a cost-effective remediation strategy. We examined i) the implications of various physical conditions upon the residence time of water passing through a bioreactor wall, allowing us to assess how likely it is for groundwater to be intercepted and treated by an installed wall, ii) the nutrient loss characteristics and wall design characteristics required for the installation of bioreactor walls to provide a viable treatment option.

Consideration of hydraulic residence time within the bioreactor wall is important as denitrification in a bioreactor can be considered a zero-order kinetic reaction (i.e. not dependent upon the concentration of NO_x-N)[48] [49]. Estimates of denitrification rates are variable in the literature dependent upon substrate type and abiotic conditions[50] with rates of 5 g N m^{-3} d^{-1} considered conservative under non-nitrate limited conditions[51]. However, results from our parallel study on denitrification potential of soils and woodchips (Appendix 5) show the woodchips deployed in our trial wall support denitrification rates, under non-nitrate-limited conditions, of 12.11 g N m^3 day^{-1} for a bed of woodchips. Dividing this value by the woodchip bed porosity (i.e. volumetric water content) of 0.53 gives a rate of decline of NO_x-N concentration of 0.95 mg L^{-1} h^{-1}. Given known concentrations of NO_x-N in groundwater we can therefore calculate the residence time required to reduce these concentrations to zero, and compare this requirement with our model outputs.

[48] Halaburka, B.J., G.H. LeFevre and R.G. Luthy. 2017. Evaluation of Mechanistic Models for Nitrate Removal in Woodchip Bioreactors. Environmental Science & Technology. 51:5156-5164.
[49] Conclusions from Department of Industry - Innovations Connections Project: *Improving the quality of water for release from land-based aquaculture in northern Australia*. In collaboration with Mainstream Aquaculture.
[50] Addy, K., A.J. Gold, L.E. Christianson, M.B. David, L.A. Schipper and N.A. Ratigan. 2016. Denitrifying Bioreactors for Nitrate Removal: A Meta-Analysis. Journal of Environmental Quality. 45:873-881.
[51] Prof. Louis Schipper, personal communication

Modelling framework

This modelling framework (Figure 40, and associated R script "Bioreactor Wall Modelling") is based upon the installation of a wall of a given length (W_l), thickness (W_t) and height (W_h), approximately perpendicular to the prevailing groundwater flow from a paddock of a given area and shape. It is envisaged that walls would be placed adjacent to drains or at the downslope edge of paddocks without drains on their edges (Figure 40, Figure 42). The ratio of contributing area to wall length varies according to the area and shape of the contributing area and length of the wall.

Groundwater discharge

Within our steady-state model it is assumed that lateral groundwater discharge (Q) follows Darcy's Law (Equ. 2) for saturated flow through an unconfined homogeneous aquifer. Given the very high porosity (ϕ) and hydraulic conductivity of woodchips, ~0.53 and 0.02 m s^{-1} respectively[52], (which are far greater than all feasible surrounding soils- Table 8) it can be assumed that groundwater discharge through the wall (Q) is determined by the hydraulic conductivity of the surrounding soil (K_s) rather than the hydraulic conductivity of the wall. Q can be calculated by

$$Q = K_s \times A \times \frac{dy}{dx} \qquad \text{Equ. 2}$$

where A is the cross-section area of the wall face ($W_l \times W_h$) and dy/dx is the hydraulic head or gradient.

[52] Conclusions from Department of Industry - Innovations Connections Project: *Improving the quality of water for release from land-based aquaculture in northern Australia*. In collaboration with Mainstream Aquaculture.

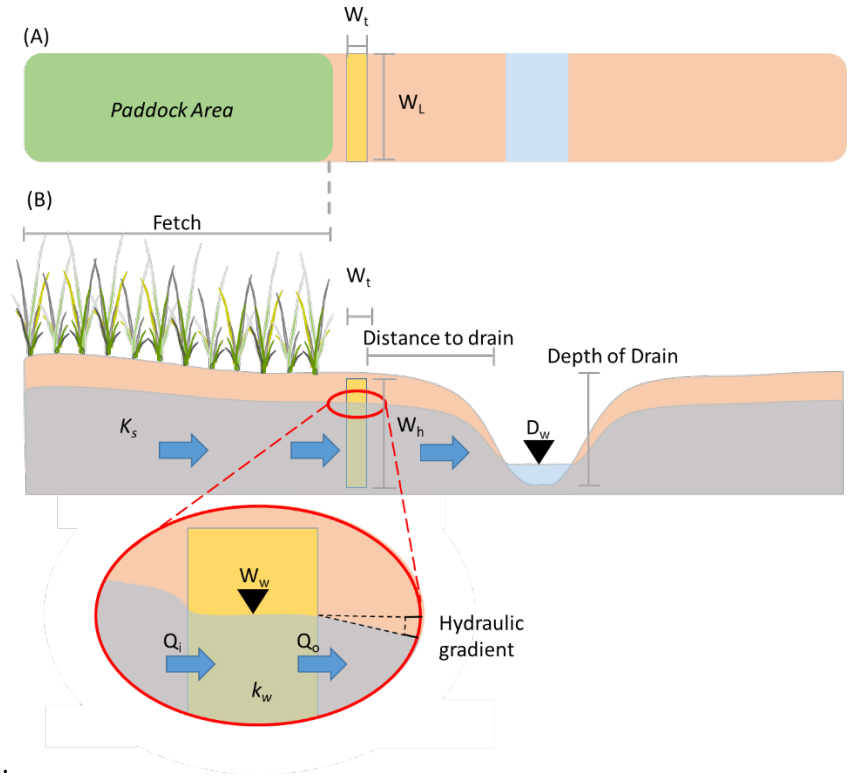

Figure 40: Plan (A) and side (B) view of the 2D non-dynamic model used to test the limitations and controls of bioreactor wall efficacy.

Residence time inside bioreactor wall

The velocity (V, m s^{-1}) of water flowing through the woodchips of the bioreactor wall in our steady state model is given by Equ. 3, wherein ϕ is the porosity of woodchips (taken as 0.53).

$$V = \frac{Q}{A \times \phi} \qquad \text{Equ. 3}$$

It follows that the hydraulic residence time (HRT) of water passing through the woodchips of a bioreactor wall is given by Equ. 4.

$$HRT = \frac{W_t}{V} \qquad \text{Equ. 4}$$

Model Parameterization

Soil saturated hydraulic conductivity

Estimates of saturated hydraulic conductivity (K_s) can be taken from standard soil lookup tables (Table 8) or measured in-situ. For example, we measured K_s upslope of the wall using slug tests in piezometers F2 and F3 (Figure 34). Slug tests were carried out by removing water during a piezometer purge event, recording the water table recovery using a pressure transducer (Hobo, U20L-02, Onset, Bourne, MA, USA) and calculating K_s using the Bouwer-

Rice approach[53]. The calculations were made using the spreadsheet described by Halford and Kuniansky (2002).[54] The mean value of K_s was found to be 6.6 x 10^{-7} m s^{-1}.

To include the range of soil hydraulic properties in which denitrifying bioreactor walls may be deployed we tested the impact of K_s between 0.0001 m s^{-1} and 0.00000001 m s^{-1}, representative of conditions from a loose sand/gravel to a heavy clay.

Table 8: Representative values for saturated hydraulic conductivity of soils (K_s)[55]

Texture	K_s (m s^{-1})
Clay	10^{-11} - 10^{-8}
Silt	10^{-9} - 10^{-5}
Sand (fine or silty)	10^{-7} - 10^{-5}
Sand (coarse)	10^{-5} - 10^{-3}
Gravel	10^{-3} - 10^{0}

Hydraulic gradient

The hydraulic gradient that drives groundwater movement through soil is temporally highly variable, indeed at our site this gradient ranged from ~0 during the dry season to a maximum of 0.02 m m^{-1} (i.e. 2%) during the wet season. However, in other installations, with walls closer to free-moving drain lines, this gradient could vary more, especially during large rainfall events. Therefore for the purposes of the trial, we tested how a large variation in hydraulic gradient impacted hydraulic residence time in a wall of a given thickness. Specifically, hydraulic gradient was varied from a minimum of 1% (representing a typical paddock surface topography) to a maximum of 100% (e.g. $\frac{(W_w - D_w)}{Distance\ to\ drain} = 1$, see Figure 40), which may occur transiently if the soil profile and bioreactor wall are saturated but the receiving drain is empty.

Groundwater NO_x-N concentrations

In addition to the trial measurements of groundwater NO_x-N concentration under sugarcane ratoon (Figure 37), several other studies have examined nitrogen concentrations in shallow groundwater under sugarcane to identify nitrogen loss pathways. Prove et al. (1997)[56] reported concentrations in deep drainage ranging from 0 to 3.2 mg N L^{-1}, and Armour et al. (2013)[57] reported mean NO_x-N concentrations in deep drainage (1-m depth under sugarcane) of 3.7, 0.5 and 4.7 mg L^{-1} in 2007/2008, 2008/2009 and 2009/2010, respectively. In this study we found a maximum concentration of 5.1 mg N L^{-1} in the groundwater

[53] Bouwer, H. and Rice, R.C., 1976. A slug test for determining hydraulic conductivity of unconfined aquifers with completely or partially penetrating wells, Water Resources Research 12(3) 423–428.
[54] Halford, K.J. and E.L. Kuniansky. 2002. Documentation of Spreadsheets for the Analysis of Aquifer-Test and Slug-Test Data, U.S. Geological Survey.
[55] Sanders, L.L. 1998. 'A Manual of Field Hydrology' (Prentice Hall Inc., New Jersey)
[56] Prove, B. G., et al. (1997). DAQ3S Nutrient Balances and Transport from Agricultural and Rainforest Lands, QDNR/QDPI.
[57] Armour, J. D., et al. (2013). "Nitrogen leaching from the root zone of sugarcane and bananas in the humid tropics of Australia." Agriculture Ecosystems & Environment 180: 68-78.

upslope of the wall. These values allow us to assume that the maximum concentrations of NO_x-N entering a bioreactor wall would be below 6 mg L^{-1}.

Model output: hydraulic residence time

Our model output (see associated executable R code) show that in a wall of between 0.3 and 0.6 m the residence time of water traversing the wall is very high across most soil types (Figure 41). Indeed with our installation dimensions (W_t = 0.65 m) and measured soil hydraulic conductivity (6.6 x 10^{-7} m s^{-1}) we find that hydraulic residence time is likely in excess of 500 h (20 days) for the hydraulic gradients observed at the site

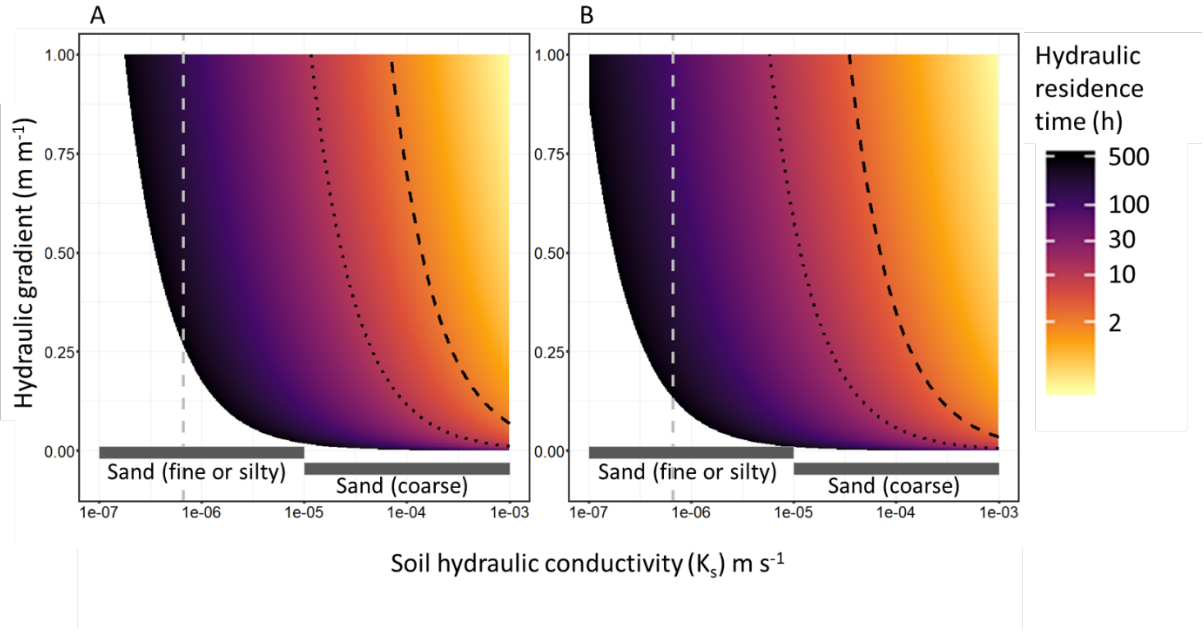

Figure 41: Modelled hydraulic residence time of groundwater traversing a bioreactor wall under conditions of variable soil hydraulic conductivity and hydraulic gradient, with a wall thickness of 0.6 m (A) and 0.3m (B). Grey bars represent range of hydraulic conductivity typical of noted soil textures, while grey dashed line represents measured site hydraulic conductivity. Black lines represent isolines for hydraulic residence time required to treat groundwater at 1 mg L^{-1} (dashed) and 6 mg L^{-1} (dotted) respectively. Note HRT >501 removed for clarity

Using a conservative estimate of potential denitrification efficiency of woodchip bioreactors of 10 g N m^{-3} day^{-1} (or a decline of 0.79 mg N L^{-1} h^{-1} assuming a porosity of 0.53), complete removal of the NO_x- N could be achieved with a residence time of 1.27 h if water entered at 1 mg L^{-1}, or 7.63 h if it entered at 6 mg L^{-1} (the highest concentration observed). As can be seen (Figure 41) it is only in soils with very high hydraulic conductivity (i.e. coarse sand) and under significant hydraulic head (i.e. >0.1 m m^{-1}) that hydraulic residence time within a wall of even 0.3 m thickness would fall below this threshold.

It is therefore apparent that bioreactor walls are likely to treat any groundwater that is intercepted. However, the amount of nitrogen removed will be small in most cases, because of the slow movement of groundwater, assuming a homogenous conductivity. The exception to this would be in soils with a uniformly high hydraulic conductivity (i.e. coarse sands), where extensive wall deployment may prove effective or in areas of highly conductive soil that occur in a heterogeneous landscape, e.g. paleochannels. In this case

walls could be installed to intercept areas of high groundwater flow, however, the identification of such features requires specialized equipment and techniques (e.g. EMI).

Wall bioreactor - cost effectiveness modelling

Having established that bioreactor walls effectively remove NO_x-N from water traversing them, we determined the nitrate loss rates and physical arrangement of paddocks and walls required for bioreactor walls to be a viable cost-effective treatment option. This included using estimates of nitrate loss rates from the literature, a range of interception ratios (paddock area: wall length) and three estimates of installation costs.

Installation costs for modelling

Three cost scenarios were tested, two including constructions costs only, specifically the use of conventional excavator or use of a rapid trenching tool, and then a conventional construction method but including project management costs as estimated above.

The conventional excavator method is based upon a standard 0.45-m wide, 2-m deep trench filled with 1.5 m woodchips and a 0.5-m soil cap. It assumes excavator hire at \$115 h^{-1} with an estimate of being able to completely install 32 m per day. As wall width is unlikely to be a limitation (Figure 41), future construction may be simplified with the use of commercial trenching tools (e.g. Ditchwitch ST120H Attachment[58]). With the use of this, or similar tools, wall depth could be extended to 2.4 m with a width of between 0.15 to 0.6 m. We therefore modelled a wall 0.3 m wide and 2 m deep, with a 0.5-m soil cap and estimated that, although requiring specialized equipment (higher rental cost), faster construction would result in a cheaper construction cost per m. The use of a 0.3m wall width for cost modelling purposes was greater than that believed to be necessary given consideration of hydrologic residence times (Figure 41). However, concerns have been raised on the practical constraints of packing walls any narrower than this, given the risk of woodchips 'bridging' during filling[59].

It should be noted that for all scenarios costs associated with site survey and assessment are not included and may represent a significant additional cost. With estimates of EMI soil mapping and site assessment ranging from \$77 to \$168 ha^{-1}, dependent on paddock arrangement and job size[60].

[58] https://www.ditchwitch.com/trenchers/ride-on/rt125
[59] Richard Gloyne, DrainTech https://www.draintech.net.au/ Personal communication.
[60] David Morrison, DNRME. Personal communication

Table 9: Construction costs for Bioreactor wall per linear meter. Estimates derived for a) build using trenching machine to dig 0.3m wide, 2-m deep wall (including 0.5-m soil cap), b) conventional build using standard excavator to dig 0.45-m wide 2-m deep wall (including 0.5-m soil cap) and c) conventional build plus project management costs

Item	Units m^{-1}	Cost ($ $unit^{-1}$)	Cost ($ m^{-1})
Construction only: Trenching construction			
Woodchip supply (local softwood)	0.45 m^3	22.00	9.90
Trencher hire	0.1 h	200.00	20.00
			Total: $30
Construction only: Conventional construction			
Woodchip supply (local softwood)	0.675 m^3	22.00	14.85
Excavator hire	0.25 h	115.00	28.75
			Total: $44
Conventional Construction + Project management			
Woodchip supply (local softwood)	0.675 m^3	22.00	14.85
Excavator hire	0.25 h	115.00	28.75
Project management	0.75 h	120.00	90.00
			Total: $133

Paddock area-to-wall-length ratio

The ratio of paddock area to effective bioreactor wall length is infinitely variable depending upon paddock geometry, paddock area and groundwater flow. Here we consider it ranging from a small number e.g. 35 m^2 m^{-1}, in which a 2-ha square paddock is surrounded by a wall on all sides, to a large number e.g. 5,333 m^2 m^{-1} in which a 15 m wall is able to intercept groundwater draining from an 8-ha paddock (Figure 42). It should be noted that in these scenarios of high loss rate and highly localized wall placement to intercept preferential flow (i.e. narrow paleochannel draining a large area) wall construction may need to be modified to increase the total volume of engaged wood chips.

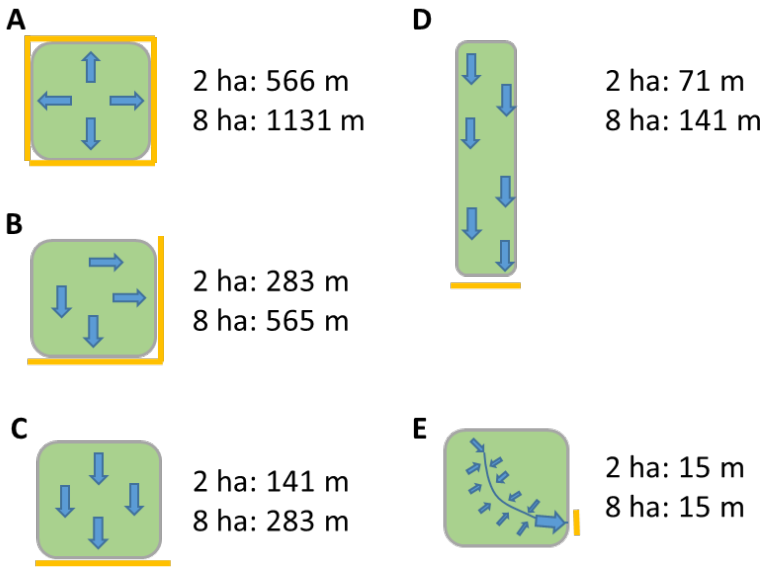

Figure 42: Example ratios of cropping area to bioreactor wall length (orange line) for several combinations of wall length and cropping area size and shape, including A) a ringed paddock, B- D various combinations to catch prevailing groundwater flows and E) where a paddock is drained by a preferential subsurface flow path resulting in a nitrogen loss pathway that can be identified and intercepted by a short wall. Note that the ratio of paddock area to wall changes with the size of the cropping area considered.

Estimates of NO_x-N load flowing through bioreactor wall

Estimates of N losses to groundwater are varied and have been shown to be impacted by edaphic and hydrological conditions as well management practice. Reghenzani et al. (1996)[61] and Prove et al. (1997)[62], working in South Johnstone, showed rates of between 7 and 56 kg N ha^{-1} yr^{-1} while Bohl et al. (2000)[63] estimated average losses of 17 kg N ha^{-1} yr^{-1}, although losses of up to 70 kg N ha^{-1} yr^{-1} were observed in sandy river bank soils of the Herbert region. More locally, Armour et al. (2013)[64] estimated N losses to groundwater from sugarcane as 9.2, 1.0 and 7.1 kg ha^{-1} yr^{-1} over three successive years. Once it has reached the groundwater in deep drainage (recharge) this nitrogen diffuses downwards and moves laterally with the groundwater, towards discharge zones. A portion of this lateral movement will be through the top few metres of soil where it can be intercepted with a bioreactor wall. We call this the 'shallow lateral loss rate'. In cases where permeability reduces with depth (e.g. clay aquitard at depth), the proportion moving laterally near the ground surface is higher. We therefore modelled a range of shallow lateral loss rates of NO_x-N, from 0 to 20 kg N $ha^{-1}yr^{-1}$, which covers the range likely, given the deep drainage losses mentioned above.

[61] Reghenzani, J. R., Armour, J. D., Prove, B. G., Moody, P. W., McShane, T. J. 1996. Nitrogen balances for sugarcane plant and first ratoon crops in the wet tropics. *Sugarcane: Research towards efficient and sustainable production*. (Eds. Wilson JR, Hogarth DM, Campbell JA and Garside AL). CSIRO Division of Tropical Crops and Pastures, Brisbane. 1996. pp. 275-277 275-277.

[62] Prove, B. G., et al. (1997). DAQ3S Nutrient Balances and Transport from Agricultural and Rainforest Lands, QDNR/QDPI.

[63] Bohl, H. P., et al. (2000). "Nitrogen losses via subsurface flow from sugar cane on floodplain soils in the Australian wet tropics." Proc. Aust. Soc. Sugar Cane Technol. 22: 302-307.

[64] Armour, J. D., et al. (2013). "Nitrogen leaching from the root zone of sugarcane and bananas in the humid tropics of Australia." Agriculture Ecosystems & Environment 180: 68-78.

Cost of nitrogen removal

Using the information above, we then modelled the cost of N removal by the equation 5

$$Cost\ per\ kg\ N\ removed = \frac{Installation\ cost\ per\ unit\ wall\ length \times Wall\ length}{Paddock\ area \times Shallow\ lateral\ loss\ rate \times Wall\ life\ span} \quad \text{Equ. 5}$$

across a range of N loss rates and wall lengths, considering i) two paddock areas (2ha and 8ha, representative of many sugarcane paddocks in the region) and ii) the three construction cost scenarios as described above. We also assumed a working lifespan of 10 years. It should be noted that these costings do not include the need for site surveying, which may be considerable in cases where small targeted walls are being considered. It also needs to be kept in mind that site-specific characteristics of the contributing area, such as surface topography, soil variability and the presence of mole or ag drains, will have a large effect on performance and hence cost-effectiveness. However, the model presented here can be used to estimate the cost effectiveness of bioreactor walls in a broad range of likely conditions.

Model outputs (Figure 43, Table 10: Threshold N-loss rates to achieve stated cost effectiveness of nitrogen removal rate. Given design criteria (e.g. paddock area, wall length) and cost estimates of construction assuming 10 yr working lifespan.), show the dynamic interplay of construction cost, paddock area, N-loss rate and wall length upon cost effectiveness of bioreactor walls. Taking a 8 ha contributing area, and wall of 283 m (e.g. one side of a square paddock) and construction cost of $30 per m, the cost per kg of N removed drops below 100, 50 and 25 $ kg^{-1} if average shallow lateral loss rates that can be intercepted by bioreactor walls rise above 1.1, 2.1, and 4.2 kg ha^{-1} yr^{-1} respectively. Similarly, taking a 2-ha contributing area and assuming a bioreactor wall of 141 m (e.g. one side of a square paddock of 2 ha) and construction cost of $133 per m wall we see the cost per kg of N removed drop below 100, 50 and 25 $ kg^{-1} only if average N loss rates from the paddock rise above 9.6, 19.2, and 38.3 kg ha^{-1} yr^{-1} respectively. This range in model outputs highlights the importance of accurately understanding water movement and nitrogen loss pathways in the landscape and targeting wall placement to limit construction costs while maximising interception of groundwater flow.

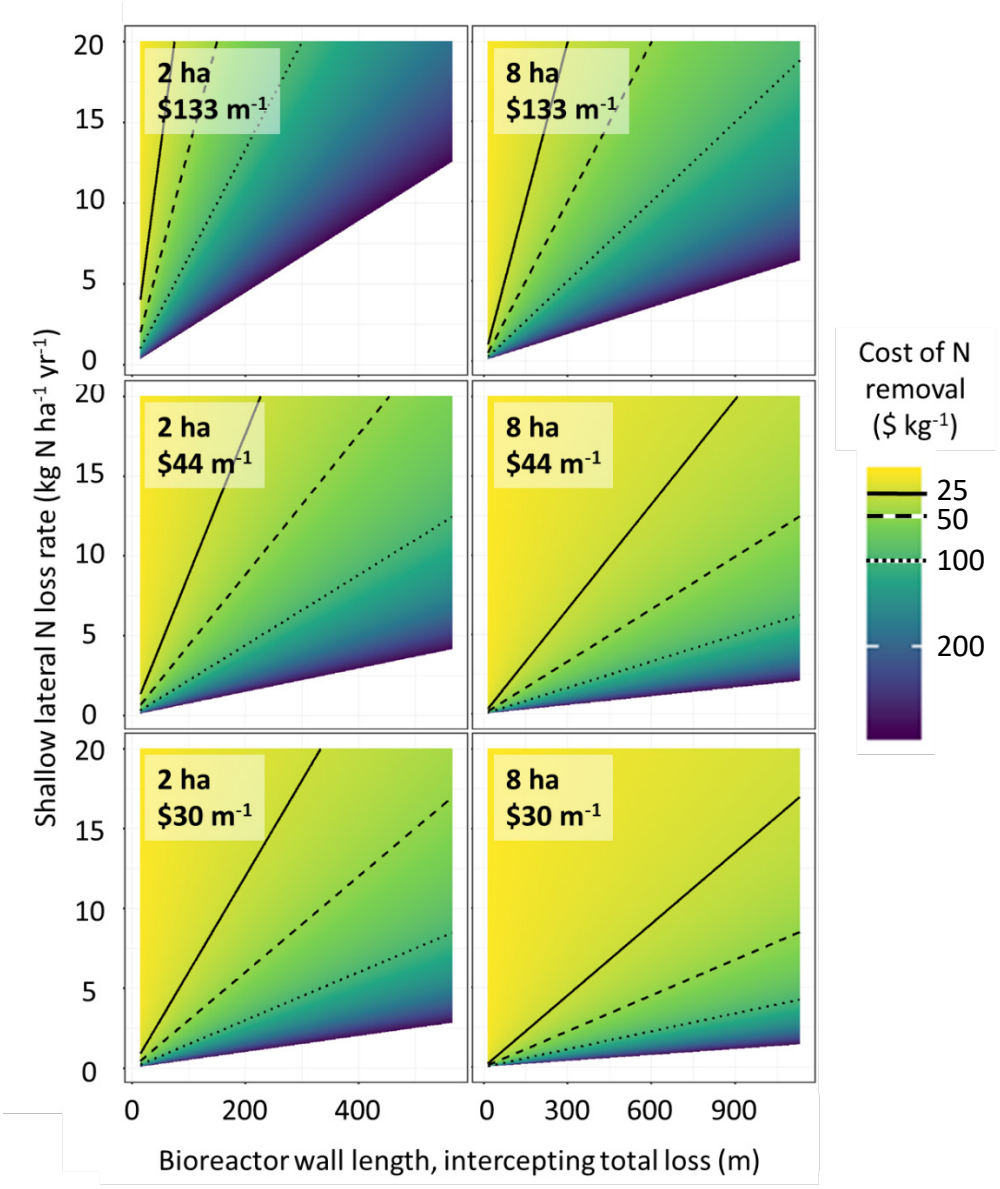

Figure 43: Modelled cost of N removal by bioreactor walls in relation to shallow lateral N loss rate and the length of bioreactor wall required to intercept the loss (see Figure 42), assuming paddock area of 2 ha or 8 ha, a construction cost of $30, $44 or $133 per linear metre, and a 10-year working lifespan.

Table 10: Threshold N-loss rates to achieve stated cost effectiveness of nitrogen removal rate. Given design criteria (e.g. paddock area, wall length) and cost estimates of construction assuming 10-year working lifespan.

Paddock Area (ha)	Installation cost ($ m^{-1})	Wall length (m)	N loss rate (kg ha^{-1} yr^{-1}) required to achieve		
			$100 kg^{-1}	$50 kg^{-1}	$25 kg^{-1}
2	133	144	9.6	19.2	38.3
2	44	144	3.2	6.3	12.7
2	30	144	2.2	4.3	8.6
8	133	283	4.7	9.4	18.8
8	44	283	1.6	3.1	6.2
8	30	283	1.1	2.1	4.2

Wall bioreactors – considerations for future design

It is apparent from hydrologic modelling (Figure 41) that bioreactor walls of even modest width are more than sufficient to achieve a hydraulic residence time required to remove all the NO_x-N from groundwater flowing through them. It is also clear from scenario modelling (Figure 43) that bioreactor walls with realistic design configurations can achieve meaningful and cost-effective remediation rates. However, and as experienced in our study, the locating of bioreactor walls to successfully achieve substantial nitrogen removal requires a detailed a-priori understanding of hydraulic conductivity and groundwater movement across the landscape.

We envisage two distinct conditions under which the deployment of bioreactor walls would prove to be a useful management tool i) in soils with high homogeneous hydraulic conductivity, K_s ii) in heterogeneous soils in which regions of high K_s or preferential flow exist. In both cases there would need to be a significant loss of nitrogen in lateral groundwater movement amenable to interception by a bioreactor wall as constructed. This is likely the greatest unknown as regards nitrogen loss to groundwater in agricultural systems – whether it passes through shallow groundwater to local discharge points i.e. drains that could be intercepted by bioreactor walls, or not.

It may not be sufficient to know a site's soil texture to predict K_s. For example, Smettem and Bristow (1999)[65] examined the relationship between hydraulic conductivity and clay content in sugarcane growing topsoils of QLD and found a poor relationship between clay content and K_s due to the large component of flow through macropores, which varied irrespective of texture. There may therefore be a need for in-field validation to determine site K_s to judge if extensive bioreactor walls are viable.

Similarly, while the identification of preferential flow paths may be helped by local knowledge it is clear that detailed subsurface surveying with techniques such as EMI are required to truly understand groundwater movement at the paddock scale. With the recent development of Reef Protection Regulations methodology[66] including the requirement for "Farm Nitrogen and Phosphorus Budgets" based upon fine-scale soil mapping there is the possibility that the high-resolution data capture required for effective bioreactor wall deployment may become common industry practice.

It is also worth noting that in many regions of the lowland wet tropics, and especially the BSDA, poor draining soils with generally low K_s are common, and artificial drainage (e.g. unlined 'mole' and interceptor drains) is routinely installed to accelerate removal of excess water from the root zone. This provides an opportunity to develop 'hybrid walls' in which woodchips may be used in place of gravel in interceptor drains (Figure 44). This highly novel design of bioreactor can be considered a means of increasing paddock K_s (Figure 41) and as such there is a need to consider if hydraulic residence time is sufficient to remove NO_x-N

[65] Smettem, K.R.J. and K.L. Bristow. 1999. Obtaining soil hydraulic properties for water balance and leaching models from survey data. 2. Hydraulic conductivity. Australian Journal of Agricultural Research. 50:1259-1262.
[66] Queensland Reef Water Quality: Farming in Reef catchments- Prescribed methodology for sugarcane cultivation. 2019. Office of the Great Barrier Reef, Environmental Policy and Programs, Department of Environment and Science.

observed. There are also issues surrounding long-term viability of woodchips as a permeable backfill in interceptor drains that would require testing.

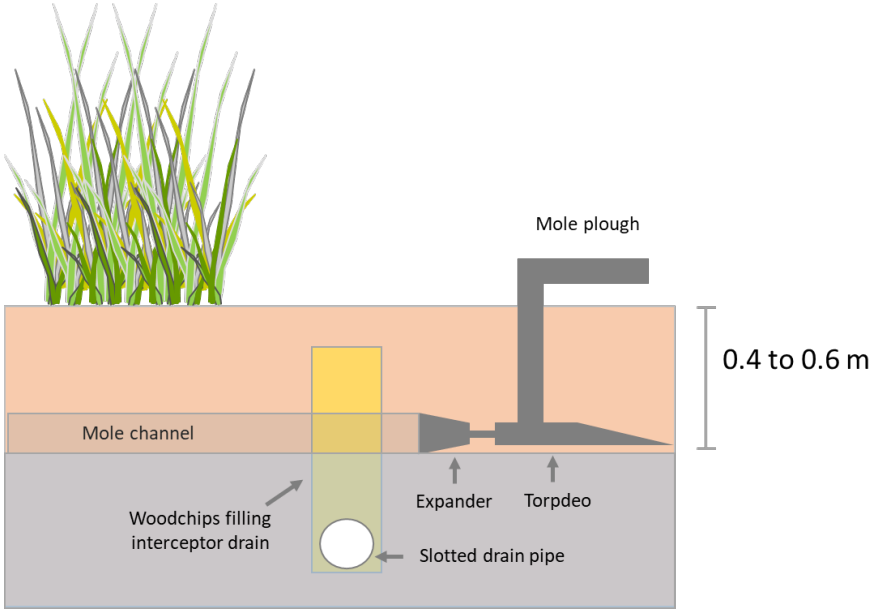

Figure 44: Schematic for 'hybrid bioreactor wall' in which woodchips are used surrounding interceptor drains in areas of mole drainage

Potential Uptake of Bioreactors

Background

The trial encompassed a socio-economic objective to identify factors that might influence broader uptake of bioreactors by sugarcane farmers of the BSDA and Russell River catchment following the trial. This was on the basis that the degree of landholder adoption could be impacted by level of understanding, motivation and/or capacity to adopt bioreactor technology[67,68]. In particular, the study sought to identify (a) the main barriers and opportunities for farmers to adopt bioreactor technology, and (b) how bioreactor technology may be most effectively encouraged.

Insights into these questions were captured using face-to-face interviews with eight end user sugarcane farmers with the assumption that bioreactors technology would be at least partially effective.

Interview method

The interviews were conducted using the 'general inductive method'[69] as part of a Masters research thesis, undertaken through the College of Science & Engineering, James Cook University.[70] Interviews involved a series of pre-determined questions to guide conversations and give coverage of the following themes:

- Level of motivation that might influence adoption of bioreactors, such as values and attitudes
- Level of capacity that might either encourage or discourage adoption of bioreactors, such as finances, availability of time or perceived risks, and
- Level of social connection with farming peers that might influence their decision to include bioreactor technology as standard or best practice.

As most farmers had limited or no experience with bioreactors, farmers were provided with a written overview of bioreactor technology, including their potential to reduce nitrogen. They were also asked about 'good farming practice' to gauge level of knowledge on management of nutrient inputs and outputs to contextualise perspectives.

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[67] Macgregor, C. (2009). Challenges of Applying sustainability. *Presentation to the School of Economic Science, Justice and Equity Conference*, 23rd – 30th August 2009, Oxerhof, Daventer, Netherlands.

[68] Pannell, D. J., Marshall, G. R., Barr, N., Curtis, A., Vanclay, F. & Wilkinson, R. (2006). Understanding and promoting adoption of conservation practices by rural landholders. *Australian Journal of Experimental Agriculture, 46*(11), 1407-1424. doi:10.1071/EA05037

[69] Thomas, D. R. (2006). A general inductive approach for analyzing qualitative evaluation data. *American Journal of Evaluation, 27*(2), 237-246.

[70] Jai Kaartinen-Price, 2019, 'Barriers and opportunities for denitrification bioreactor adoption by cane farmers in the Wet Tropics, Australia'. Minor Masters Project, James Cook UNiversity

The following results summaries reflect farmer information needs, concerns and implementation considerations based on their knowledge of the catchment. More complete details of the interview methods, data analysis and results are in Appendix 6.

Results

Overall farmers' views about bioreactors was positive, especially if the trials demonstrated their effectiveness within the climatic and soil constraints of the Russell River catchment. The best circumstances for farmer adoption involve:

- Conclusive evidence that the Russell River catchment has a nitrogen pollution problem and particularly as it relates to individual farm runoff
- Promotion of environmental benefits to improve uptake
- Guidelines on a standard design and installation procedure, and
- Financial assistance for implementation.

Proof of effectiveness

To have confidence in the technology, farmers require results on the local performance of bioreactors. This is based on uncertainty about their effectiveness in the climatic conditions of the BSDA and Russell River catchment. It is accompanied by the need for more information as to what bioreactor technology entails, need to consider placement for maximum effect within the landscape and need for guidelines for a standard installation design. Views as to the effectiveness of bioreactors centered on capacity of bioreactors to perform:

- During high rainfall events, i.e. likelihood of flow bypassing bioreactors during first-flush events following fertilisation
- In high drain flows with low concentrations of nitrogen
- In soils that have poor groundwater penetration (e.g. clays) and within the context of bioreactors being put forward as a groundwater interception technology, and
- In soils that have high carbon content and inherent denitrification potential, such as peats.

Continued testing of bioreactors 'in the field' is necessary to determine their effectiveness across annual variation and, therefore, potential for widespread farm-based adoption.

Acceptance of a nitrogen pollution issue

Most farmers did not accept that a nitrogen pollution problem exists. Farmers expressed a desire for quantification of nitrogen level export from their farms to support individual ownership of the water quality issue. This included conclusive data that confirms:

- Nitrogen pollution more generally is a water quality issue. The view stems from the fact that nitrogen pollution is not observable, compared with sedimentation/siltation of the waterways.

- Sources of nitrogen across different land uses within the catchment, citing natural and urban areas. Some farmers believe they are identified as a convenient 'scapegoat' for something that they considered should be a shared responsibility.

Nevertheless, farmers expressed a desire to be responsible for off-site problems emerging from their farms. This underpins their willingness to work with Government(s) on nitrogen pollution, compared with the current sense of 'working under Government(s)'. This was accompanied by dislike for increasing industry regulation that was also linked to concern about being 'squeezed out of the industry'. For some, this included regulation of bioreactors, should implementation costs be imposed on farmers.

Placement within the landscape

Farmers freely shared their knowledge of the catchment to help inform appropriate placement of bioreactors. The views expressed indicated that careful consideration needs to be given to:

- Placement in the lower part of the catchment where drains are subject to tidal influence
- Potential to disturb acid-sulfate soils and/or highly acidic soils during construction
- Placement in the larger main drains being problematic.

Potential to adopt bioreactor technology

There was a general positive agri-environmental orientation among the interviewed farmers, with most being open to the idea of implementing bioreactors subject to the following financial and machinery constraints:

- Several farmers were aware that an in-kind percentage contribution could be used to attract Government(s) funding. These farmers were prepared to contribute through supply of machinery and time.
- Most farmers expressed limited financial capacity and the need for funding support to cover the cost of additional expenses. Farmers better able to absorb installation costs indicated a potential to do so should bioreactors prove effective.

It should be noted from the trial, however, that most farmers of the catchment would not have the heavy machinery needed if a similar in-drain bioreactor design were used. Further, interviews did not consider scale and this might affect farmers' ability or preparedness to contribute. That is, there would be a point at which total cost would likely be prohibitive for individual farmers where water quality is addressed through multiple bioreactors, rather than the notion of a single bioreactor.

To accelerate adoption, the views indicated a need to counter wider negative attitudes from farmers less willing to adopt new technologies. That is, a program of broader roll-out should publicise the environmental benefits of bioreactor technology, in terms of better information of nitrogen pollution being an issue in the Russell River catchment and the role of bioreactors in addressing the issue.

Timeframe for take-up

This study found a notable level of pride with the interviewed farmers in regard to their farm management and the local environment. This and other positive attitudes suggest farmers within the Russell River catchment are quite open and willing to 'do their part' for the environment and the GBR. Taking a bottom-up cooperative approach to encouraging adoption of bioreactors is likely to encourage deliberative constructive dialogue between landholders, environmental managers and policymakers.

However, this study also confirmed what other social science studies have found on the speed of adoption of new technologies into farm management practice.[71] [72] These indicate up to seven years to adopt from the first trials and up to eight years for environmental-based technologies.

[71] Rogers, E. M. (2004). A Prospective and Retrospective Look at the Diffusion Model. *Journal of Health Communication, 9*(sup1), 13-19. doi:10.1080/10810730490271449

[72] Macgregor, C. J., & Warren, C. R. (2016). Evaluating the Impacts of Nitrate Vulnerable Zones on the Environment and Farmers' Practices: A Scottish Case Study. *Scottish Geographical Journal, 132*(1), 1-20. doi:10.1080/14702541.2015.1034760

Conclusions and Recommendations

Although the bioreactors built and evaluated during this work significantly reduced the nitrate concentration of water moving through them, paddock-scale beds removed only a small proportion and amount of the nitrogen moving down the drain. This limited effectiveness was due to the hydrology of the Wet Tropics and issues surrounding 'first-flush' nitrogen movement from sugarcane paddocks.

Any changes to design of the beds to improve their treatment of first-flush water, apart from making them much larger, would result in a trade-off between hydraulic residence time and the proportion of drain discharge treated. The greater the flow intercepted, the shorter the hydraulic residence time (for a given volume of wood chips) resulting in a decline in the reduction in NO_x-N concentrations in intercepted water. The counterpoint to this is that under highly variable NO_x-N concentrations (as seen in drain systems of the BSDA) the effective sizing of bioreactor beds is problematic given a need to match bed hydraulic residence time to incoming NO_x-N concentrations.

The use of large-scale bioreactor beds lower in the catchment may be more effective because water flow and NO_x-N concentrations are maintained for a higher proportion of the year in larger drains. They could be considered akin to augmented or enhanced landscape wetlands. It would be worthwhile investigating large-scale bioreactors further, particularly the hydraulic constraints that may limit their design and deployment.

Bioreactor walls appear to offer more promise for significantly and cost-effectively improving water quality, in areas where the water table is close to the surface. Bioreactor walls of even modest width are more than sufficient to achieve a hydraulic residence time required to treat remove NO_x-N from groundwater flow (Figure 41). It is also clear from scenario modelling that bioreactor walls with realistic design configurations can achieve meaningful and cost-effective remediation rates (Figure 43). However, unknowns concerning effective wall lifespan, and the identification of the both the location and rate of lateral nitrogen loss rates from paddocks limit our ability to predict true cost-effectiveness of bioreactor walls, and should be a target of any future investigations on their deployment.

Building on the results of this project, it became clear that further research is needed to:

- Determine the change in bioreactor effectiveness over time, including those installed in this project, which are equipped for the purpose and have now been in place for 2 years. This would have two purposes: first, to determine their lifespan and second, to determine their effectiveness under variable prevailing climatic and management conditions.
- Continue water quality and flow monitoring across the BSDA to allow for determination of the nitrogen budget across variable climatic conditions. This would help reconcile the significant difference found between this study and Queensland DNRME (Department of Natural Resources, Mines and Energy)-modelled DIN loads per unit area of sugarcane.

- Evaluate the use of high-resolution soil mapping (e.g. using electromagnetic induction) to help determine suitable locations for bioreactor walls, in addition to informing better in-field nutrient management.
- Evaluate 'controlled drainage' as a way of delaying the first flush of runoff to enhance loss of DIN via denitrification in paddock soils, drainage systems and inline bioreactor beds.
- Determine the effectiveness of the novel bioreactor configurations and designs conceived during this project. For example, 'hybrid walls' and the use of woodchips in interceptor drains should be evaluated. Such novel designs may provide the compromise needed between concentrating NO_x-N flows (an advantage of beds) and higher hydraulic residence time (an advantage of walls) required to achieve substantial reductions in nitrogen load. Initial evaluations have commenced in collaboration with Terrain NRM, but further replicated studies are required to ascertain their true efficacy in nitrogen removal.

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Appendices

Appendix 1: Initial BSDA Study Site Assessments

Initial site visits were conducted by the project team and farmers to assess possible locations for experimental denitrifying bioreactors in the BSDA and to discuss their current farm practice and preferred locations across their property for research activities. This consultation resulted in the pre-selection of a number of possible study sites for the research.

Farm 1

The farmer showed the location of a Sugar Research Australia (SRA) trial of 'next generation fertilisers' (Zone 1 in Figure A1.1) and discussed the potential for tying in with this trial. However this was considered non-ideal given different fertiliser treatments across the paddock. It was decided preferable to have the bioreactors installed in an area receiving 'normal' BMP fertiliser regime. The SRA experimental paddock was noted as a possible additional monitoring site.

An area under normal farm practice and adjacent to the SRA trial (Zone 2 in Figure A1.1) was considered. However, after soil coring and assessment of the topography we decided that the area may not be suitable for standard bioreactor designs. The soils on this property are generally as *Hewitt series* with only a shallow topsoil overlying a distinct clay horizon at ~0.4 m[73].

Assuming low conductivity in the underlying clay this leaves only a limited zone in which the bioreactor wall could be engaged in denitrification. This zone is further reduced when we consider the need for a distinct topsoil cap to allow for headland traffic. Similarly, it is hard to envisage a standard wall design cut into the underlying clay which would enable significant water throughput. The lack of significant elevation change in zone 2 also precludes the use of hybrid bioreactor designs

Further investigations identified a region (Zone 3 in Figure A1.1) in which there are *Hewitt series* soils (sapric peat overlaying clay) with an increase in the depth of peat towards the east. However, there is also sufficient elevation change between the field and its shallow clay horizon and the major drain along the northern edge of the paddocks to allow for the trialling of a modified hybrid bioreactor or in-drain bed system Zone 3 was chosen as the site for the Inline Bed bioreactors and it is shown in more detail in the main body of the report.

[73] Murtha, G. G., et al. (1996). Soils of the Babinda-Cairns Area, North Queensland. Division of Soils Divisional Report No. 123, CSIRO Division of Soils.

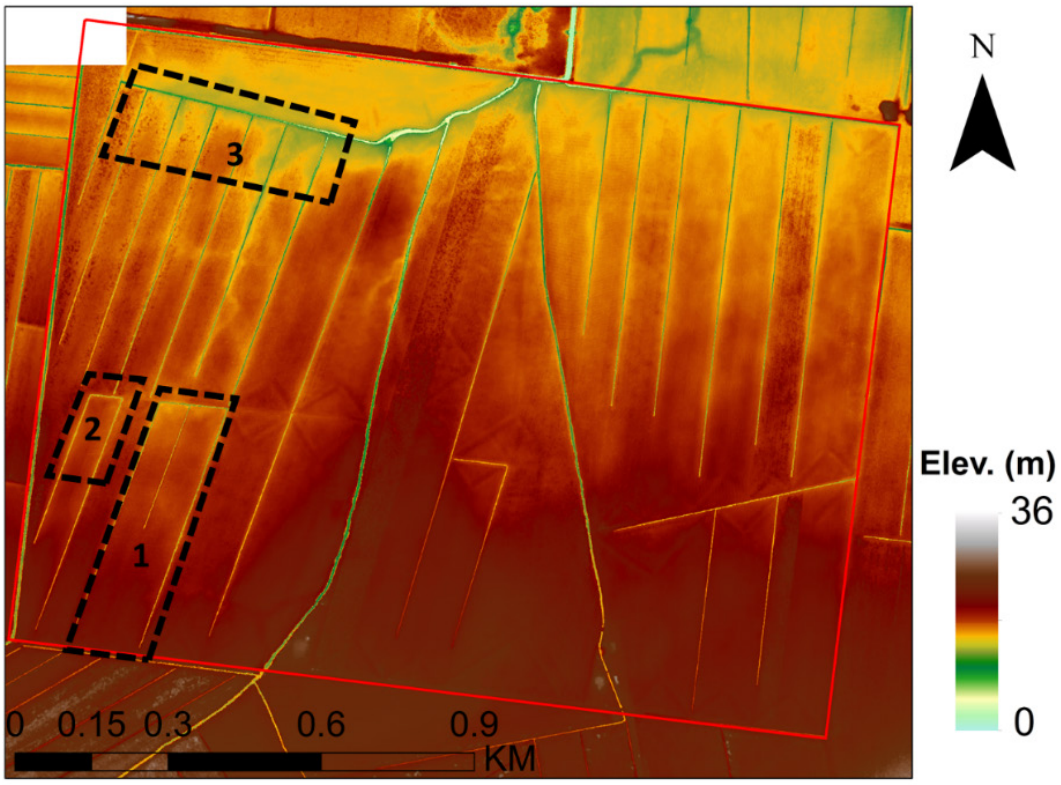

Figure A1.1. Ground surface topography of Farm 1. Digital elevation model derived from 2011 Lidar data collected by Queensland Government. Three areas for possible bioreactor locations delineated

Farm 2

On Farm 2, two potential sites were identified, both in close proximity to the main drain line and the all-weather access route running along the berm wall along the main drain (Figure A1.2)

Zone 1: Although low lying and subject to possible flooding the area receives water from the paddock to its south which represents some of the largest total elevation change on the farm, and indeed across the BSDA. The E-W aligned ditch runs through a 50-cm culvert under the berm wall into the main drain (running N-S in Figure A1.2) and water can be backed up due to tidal fluctuations. However, it initially appeared to be a suitable location for trialling of a bioreactor wall. Piezometers would be required both up and down field to track possible groundwater head reversal. Cane growing in the first 20 m appears stunted due to flooding and the farmer offered space as required to set the wall up. After considering the soil being deep peat, it was decided that a wall of woodchips was unlikely to provide conditions much different from within the peat profile itself.

Zone 2: A location favoured for research work by the farmer was assessed and deemed appropriate for an 'offline' bioreactor bed system in which water is routed from an existing drain line through a sealed bed. The bed would drain directly into the main drain line. This configuration would allow for a suitable elevation fall to maintain the bioreactor under 'flowing' anaerobic conditions while allowing for excess flow during flood events to pass directly down the existing drain line. This site was chosen for installation of the trial 'Offline Bed'.

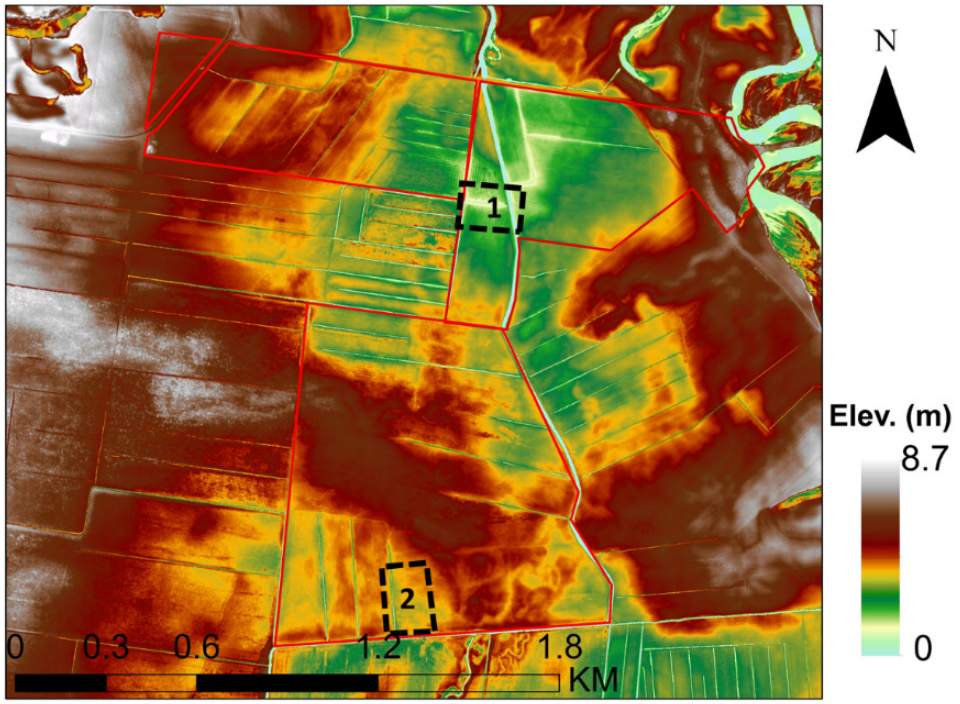

Figure A1.2. Ground surface topography of Farm 2 (outlined in red). Digital elevation model derived from 2011 Lidar data collected by Queensland Government. Two areas for possible bioreactor research identified.

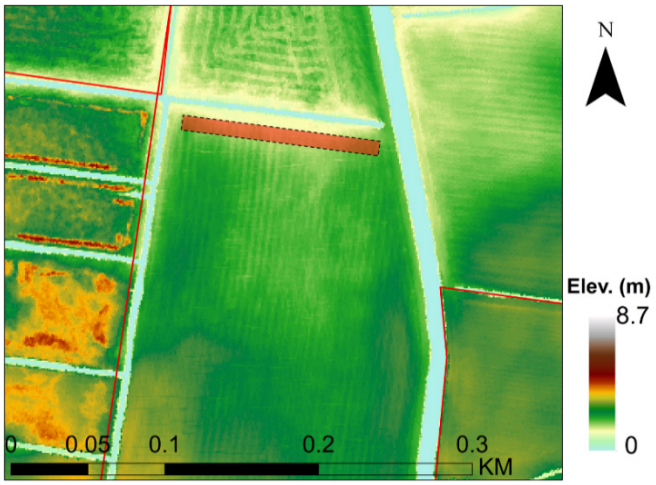

Figure A1.3. Ground surface topography of Farm 2, Zone 1. Potential position of bioreactor wall shown in red with dashed outline. Digital elevation model derived from 2011 Lidar data collected by Queensland Government.

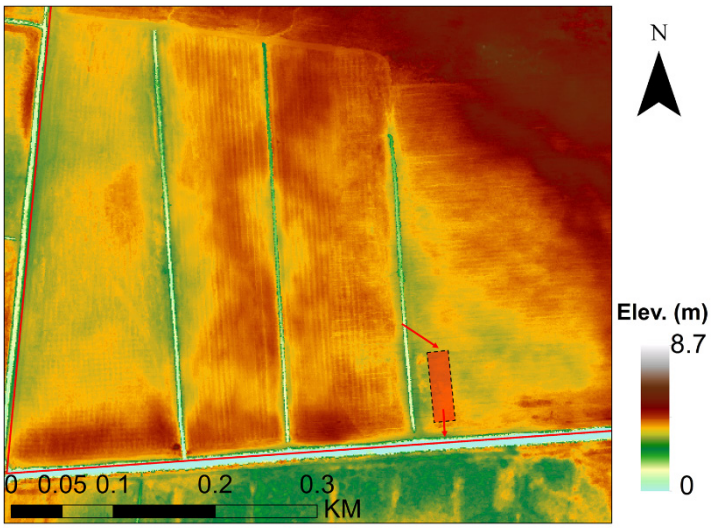

Figure A1.4. Ground surface topography of Farm 2, Zone 2. Indicative position of offline bioreactor bed shown in red with dashed outline. Digital elevation model derived from 2011 Lidar data collected by Queensland Government.

Farm 3

After inspection of numerous sites for a bioreactor wall, Farm 3 was chosen as the most suitable. The site topography is shown in Figure 34 in the main body of this report.

Appendix 2: Detailed Site Surveys

Possible bioreactor installation sites on the three farms were surveyed to facilitate bioreactor design. Detailed topographic surveys were carried out using a RTK GPS (Trimble R8 GNSS). Soil characteristics were examined using cores taken with a truck-mounted Christie hydraulic soil corer.

Soil Survey – Farm1 (Zone 3), site used for trial Inline Beds

This area consists of sapric peat overlying deep clays. In the paddock between the two potential bed sites (Figure A2.1) the depth to clay layer varies from less than 20 cm at the western edge to ~1 m at the eastern edge (Figure A2.2). When sampled, peat in core 1 & 2 was generally dry for the entire profile however water was found to pool into the hole cored into the underlying clay layer. This suggests some degree of lateral flow at the interface of the peat and clay layers. The underlying clay is seemingly deep as demonstrated at the culvert of drain (Figure A2.3).

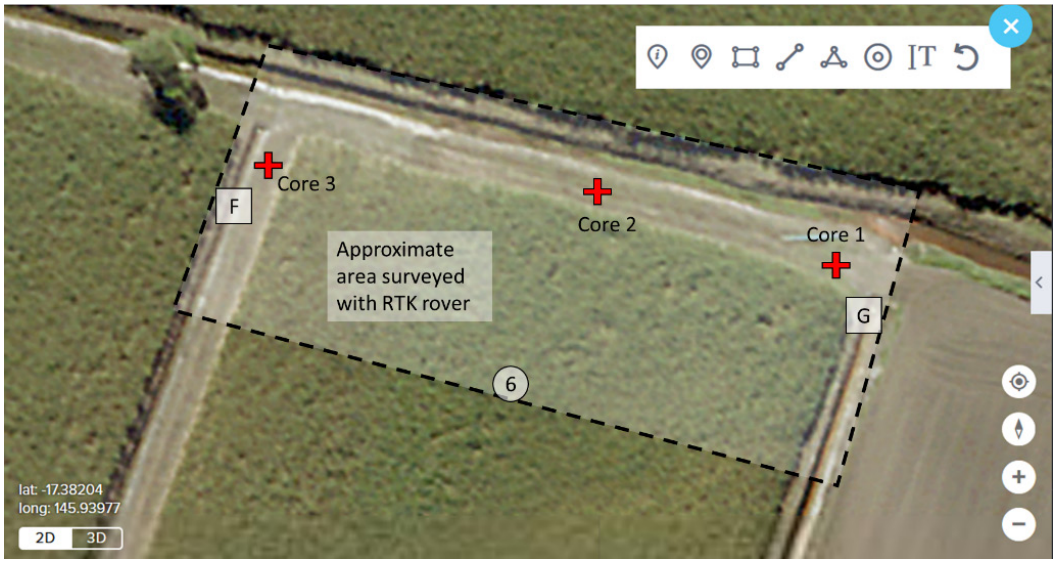

Figure A2.1. Location of soil cores taken at Farm 2, Zone 3.

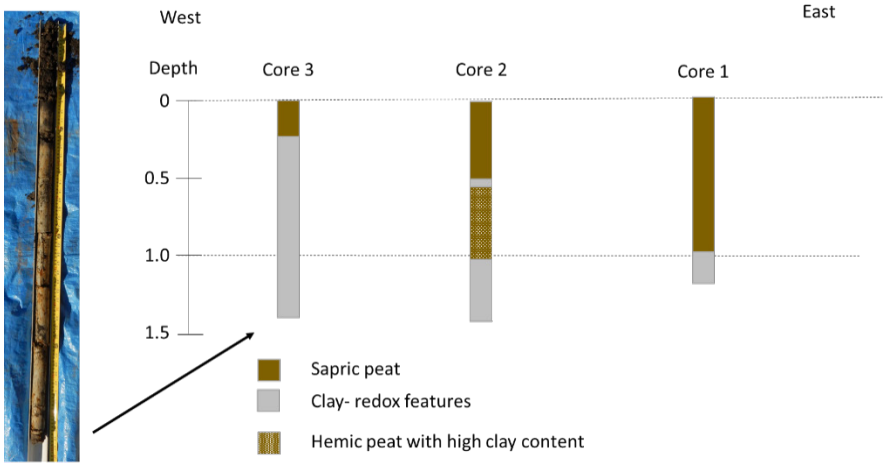

Figure A2.2. Main characteristics of soil cores taken at Farm 1 Zone 3.

Figure A2.3. View looking East at exit of minor drain line into major drain line in Farm 1 Zone 3

Soil survey- Farm 2 (Zone 1)

This site was low-lying (Figure A1.2) and the profile consisted of fibric peats with a lot of wood fragments in the profile and no confining clay layer identified within 2 m (Figure A2.4). The water table was at ~1 m. Doubts raised as to how much benefit there would be to replacing saturated organic profile with fresh wood chips, and the site was rejected as an experimental site.

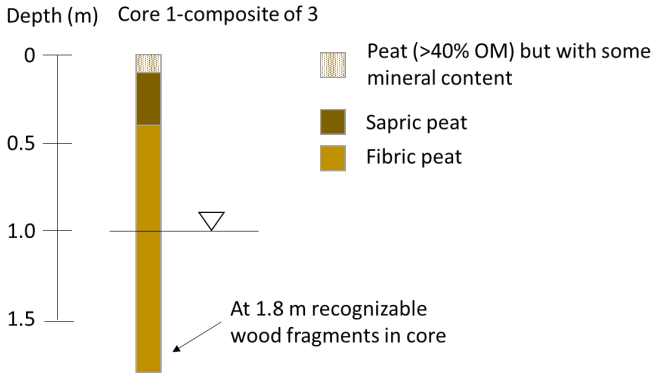

Figure A2.4. Main characteristics of a soil core taken at Farm 2, Zone 1

Soil survey- Farm 2 (Zone 2), site used for Offline Bed

This site appeared to collect a lot of surface water flow, because it is at the bottom of a slope (Figure A1.4), and erosion was observed in the drain line. The soils were peats overlying confining clay (Figures A2.5 and A2.6). The peat was approximately 1 m deep at the southern end of the drain and

became very shallow towards the north. The peat at the southern end has a lot of large wood fragments which may impede any construction efforts. At the north of the drain line river gravel layers were found below the clay layer, suggesting discontinuous and complex deeper alluvial layers from prior streams.

The area was considered suitable for an offline bioreactor, diverting water from the minor drain line through the denitrifying bioreactor before discharge into main drain line. Leaving the existing minor drain culvert for high flow events. Design and construction of the bed would need to consider how to stabilize walls and line the bed within the peat profile.

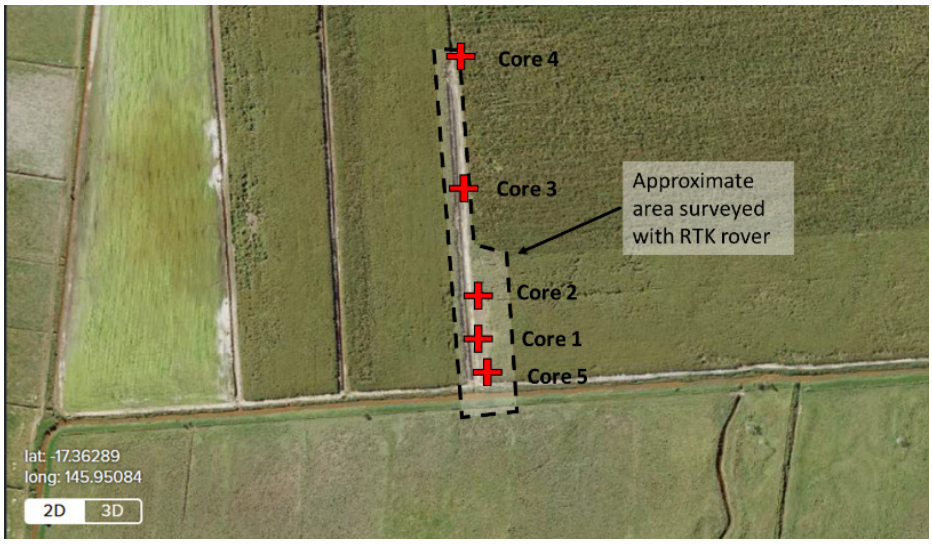

Figure A2.5. Location of topographic survey and soil cores taken at Farm 2 Zone 2.

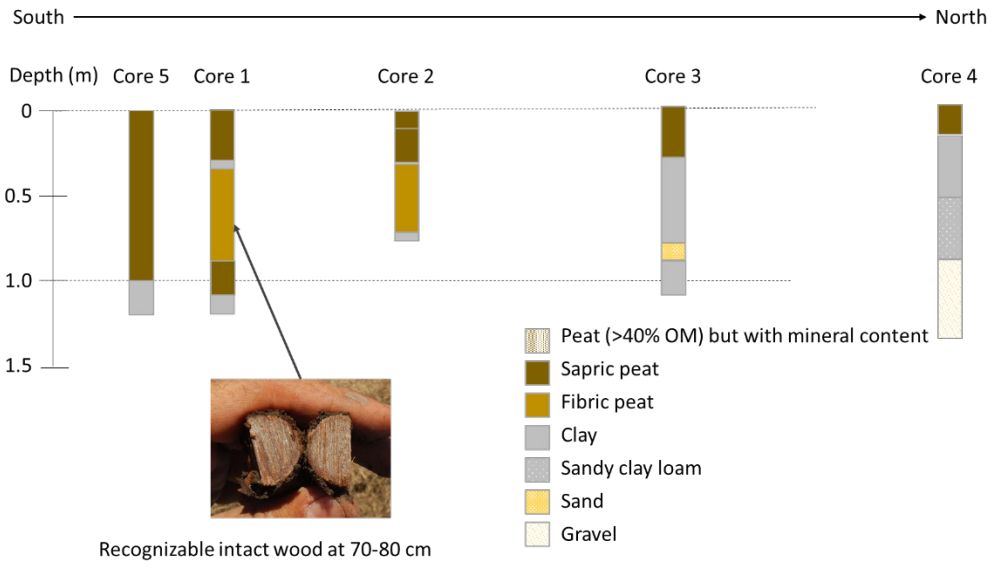

Figure A2.6. General characteristics of soil cores taken at Farm 2, Zone 2.

Soil survey- Farm 3 (used for bioreactor wall)

The regional soil map[74] shows the site to be on the boundary of the Virgil Assocation (upslope) and Coom Association (downslope), which are dominated by soil series of the same names. The Virgil series is a Red Kandosol, a well-drained uniform or gradational textured red massive soil on high terraces. The Coom series is a Redoxic Hydrosol, poorly drained, with uniform or gradational texture and yellow and grey mottled B horizons. These soils do not typically have a deep sandy layer underlain by clay at depth, which is ideal for bioreactor walls. However, a clay layer was found at depth at the site. A map of the paddock and soil core locations (ie. piezometer locations) is given in is given in Figure 34. The soil profiles are described below.

Field Core F1.B

Brown sandy loam at the surface with a sharp transition to a yellow white sand at 400-600mm. Sand particles increase in size with depth, coarse angular grains become frequent with the water table at approximately 1-1.2m depth. Small clay lens at 1.35-1.75 m, no large redox features present. No presences of other redox features in profile.

First bore in field, testing another 4 rows south.

- 0-0.2 m /brown sandy loam
- 0.2-0.4 m light brown sandy loam
- 0.4-0.6 m light brown to yellow sandy loam sand partials 1-2 mm (colour transition)
- 0.6-0.8 m yellow sand, damp, particles 1-2 mm
- 0.8-0.95 m yellow grey sand, <5% clay, particles 1-2mm, some larger particles present 2-5mm (10-15%)
- 0.95-1.15 m wet sand/gravel , yellow grey, large particles 2-5mm (~70%), some larger present.
- 1.15-1.35 m same as above
- 1.35-1.75 m same as above but with thin clay lens at bottom. Clay light yellow brown, no redox features present. Roughly 20-30 mm thick.
- 1.75-1 83 m hole kept collapsing after this point. Same features as from 0.95 m though completely saturated.
- 1.83-1.98 m no sample
- 1.98-2.00 m no sample
- 2.00-2.08 m no sample

Field Core F1

Second core 4 rows from F1. This core was the one used to install piezometer. Soil texture in this core was very similar to the first core though some small redox features were present below 2.2m.

[74] Murtha, G. G., et al. (1996). Soils of the Babinda-Cairns Area, North Queensland. Division of Soils Divisional Report No. 123, CSIRO Division of Soils.

UTM: Zone 55, Northing: 8090457.8, Easting: 386165.3
Elevation: 5.23m AHD

All soil in core was damp

- 0-0.2 m brown /light brown sandy loam (clay?)
- 0.2-0.4 m transitioning brown to light brown, sandy loam particles 1-2mm ~60%
- 0.4-0.6m light brown sandy loam sharp transition to white grey? sand at 600
- 0.6-0.8 m tan? Sand 1-3mm transition to
- 0.8-1.0 m light tan sand 1-3mm size with 10% >5mm
- 1.0-1.2 m same as above, increase in larger particles ~20%, saturated from 1200 down (WT)
- 1.2-2.3 m saturated sand, yellow white, main particle size 1-3mm ~15% larger then 5mm
- 2.3-2.4 m sandy clay, light brown with slight redox features.

Field Core F2

UTM: Zone 55K, Northing: 8090422.6, Easting: 386175.4
Elevation: 4.41m AHD

Brown/red clayey loam transitioning to a grey clayey sand at ~0.7m. Sand particles 1-2 mm with some coarse gravel >4mm.

Clay content increases with depth until 1.7m where it transitions to a red/yellow sandy clay. Red redox features present ~1-3mm long.

0-0.300mm	Brown clayey loam, ribbon 20-30mm
0.3-0.5 m	Brown clayey loam
0.5-0.7 m	Brown light clay, higher amount of clay, ribbon 30-40mm
0.7-1.1 m	Transition to grey clayey sand, Ø 1-3mm
1.1-1.3 m	Wet, grey clayey sandy, Ø1-2mm. larger aggregates 4-5mm. Water Table present.
1.3-1.5 m	Wet sandy clay, light grey
1.5-1.7 m	Yellow sandy clay,
1.7-1.9 m	Yellow sandy clay, red/yellow redox features

Field Core F3

UTM: Zone 55K, Northing: 8090397.4, Easting: 386172.2, 4.57m AHD
Identical profile to F2

Appendix 3: Spatial Variability in Drain [NO_x-N] on Farm 1

Introduction

Due to initial evidence suggesting possible low NO_x-N concentrations at the study site, a program to analyse DIN concentration of drain water was carried out. Water samples were taken from 45 points in drains on both participating farms with the BSDA on 5 December 2017 (Figure A3.1). This included a detailed survey of Farm 1 compromising of 35 sample points, with the objective to understand the dynamics of DIN loss at the paddock scale (Figure A3.1). Farm 1 is participating in both Production Unit Yield Potential (PUYP) and Enhanced Efficiency Fertiliser (EEF) trials, which will further provide for an integrated package of data involving best practice fertiliser application regimes over time.

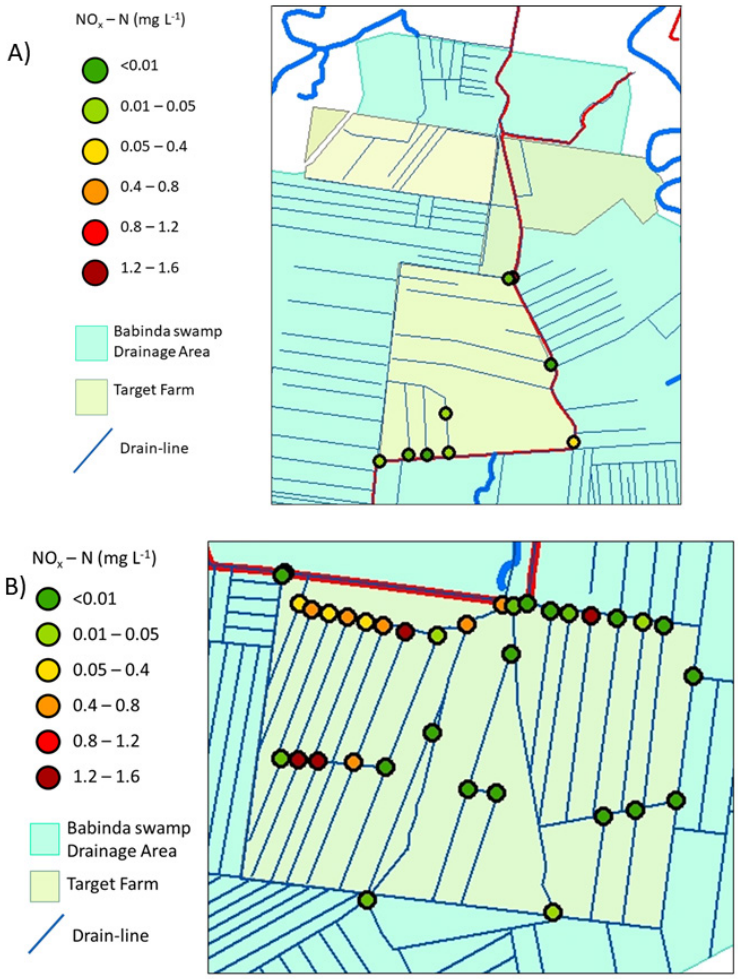

Figure A3.1. Sampling locations and concentration of NO_x-N in drain water samples collected on 5/12/2017 from Farm1 (B) and Farm 2 (A).

Methods

The spatial analysis only included drains on farm 1 that had defined catchment within the farm and had 4 or less paddocks within the farm boundaries. Drains that had outside influences or drains that did not have a well-defined catchment area were excluded.

The spatial dataset of NO_x-N concentrations was coupled with cane harvest dates, fertiliser application date, and fertiliser application rates across the farm, Table A3.1. It was expected

that NO_x-N values would decrease with increases in rainfall and time since fertiliser application.

We assessed the observed NO_x-N drain concentrations against average time since fertiliser application and cumulative rainfall total since fertiliser application. With input parameters for each drain calculated as an area-weighted average of contributing paddocks. Date of fertiliser application was assumed to be 6 weeks after the sugar cane was harvested, based on best available records kept by the farmer.

Table A3.1. Data used in linear regression analysis.

Sample_point	T_fert (days)	R_fert (mm)	NO_x-N mg/l	Paddock count
2	31.5	626.8	0.46	4
3	54.4	852.9	0.29	2
4	67.9	898.8	0.52	2
5	75.6	929.7	0.33	2
6	58.0	805.8	0.53	2
7	58.0	829.9	1.6	4
8	37.5	677.7	0.03	4
9	11.0	230.0	0.48	3
14	9.2	175.0	0.01	2
15	7.4	175.0	1.5	2
17	34.1	842.1	0.02	3
18	40.4	626.8	0.005	2
24	35.1	626.8	1.4	2
25	46.5	642.3	1.3	2
26	74.6	929.7	0.65	2
28	36.1	655.2	0.005	2
29	6.7	175.0	0.005	2
32	25.4	1005.8	0.005	4

Results and conclusions

Analysis of all grab samples showed concentrations of NO_x-N across all drains to be generally low, especially in the context of bioreactor deployment (Addy et al 2006). Samples from across the BSDA had an average NO_x-N concentration of 0.28 ± 0.45 mg NO_x-N L^{-1} (Figure A3.2), samples from within Farm 1 had an average NO_x-N concentrations of 0.48 ± 0.11 mg NO_x-N L^{-1}. Inexplicably, substantial variation was observed for adjacent minor drains between parallel paddocks, ranging from <0.01 mg (i.e. NO_x-N not detectable) to the highest reading of 1.6 mg NO_x-N L^{-1}.

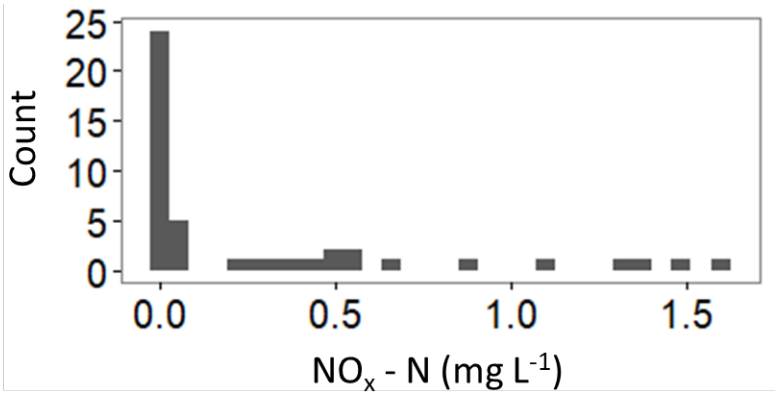

Figure A3.2. Dissolved NO_x-N concentrations of water samples collected on 5/12/2017 from drains across the BSDA (n=45).

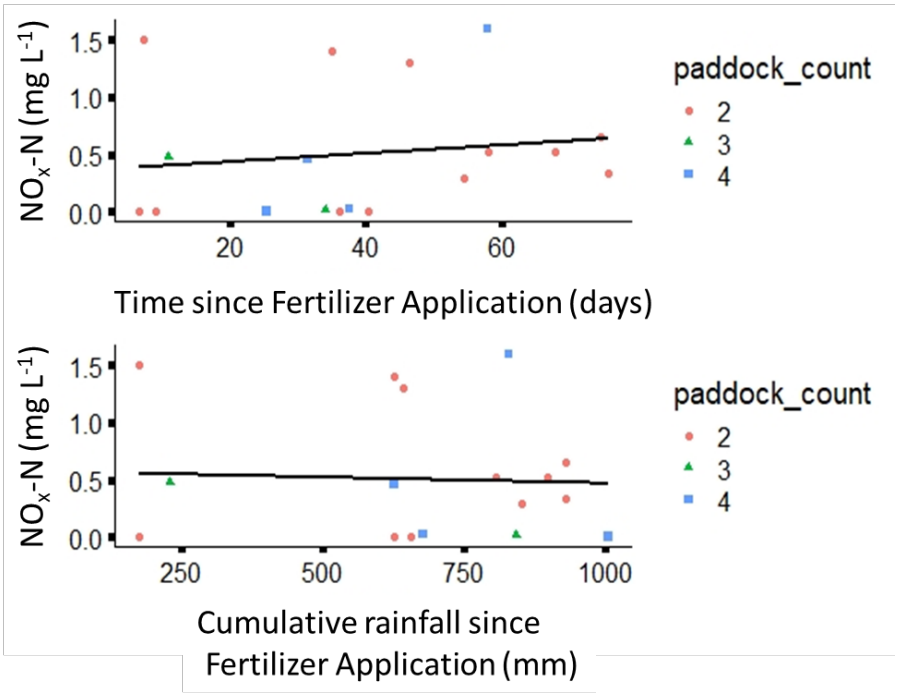

Figure A3.3. Linear regression analysis of T_fert and R_Fert parameters (n=18). Paddock count was added to visualise the difference between paddocks. ***T_fert****: p>0.005, df=16, adj-R^2= -0.04076, slope coefficient 0.0036±0.365, R^2=0.021;* ***R_fert****: p>0.005, df=16, adj-R^2= -0.060 slope coefficient -0.00010± 0.575.*

Linear regression across the spatial-data subset (Figure A3.3) showed that NO_x-N values showed, if anything, a weak positive (but non-significant) response to time since fertiliser application, opposite to our initial assumptions (p=0.57, df=16, adj-R^2= -0.04076, slope coefficient 0.0036 ± 0.365, R^2=0.021). While NO_x-N showed a negative (non-significant) response to cumulative increases in rainfall (p =0.84, df=16, adj-R^2= -0.060 slope coefficient -0.00010± 0.575).

The results of our analysis show no real discernible effect of time since fertiliser application or cumulative rainfall. However, it was clear that precise records on exact fertiliser application dates in relation to rainfall events would be needed to better address possible reasons for the observed variation in observed NO_x-N.

Appendix 4: NO_x-N Removal and Pollution Swapping in Denitrifying Bioreactors: Effects of Residence Time and Desiccation in a Laboratory Column Study

This is a paraphrased version of Laura Donovan's Honours thesis, College of Science and Engineering, James Cook University, June 2019.

Abstract

Denitrifying bioreactors can be deployed to reduce the concentration of nitrate N in water leaving farms, but have not yet been trialled in tropical conditions. Tropical conditions pose several challenges to a bioreactor's efficacy: low influent NO_x-N concentrations, short residence times and highly variable water table depths that may leave the woodchips unsaturated at certain times of the year. The aim of this study was to determine the effect of residence times (3h and 6h) and desiccation intervals (0, 3 and 13 days, in the 6-hr hydraulic residence time treatment only) on NO_x-N removal and pollutant generation with low influent NOx-N concentration in laboratory-scale bioreactors over 66 days. Nitrogen removal efficiency was >90% in the 6-h residence time treatments, irrespective of drying, and 51% with 3-h residence time. However, nitrogen removal was ~5 g N m^{-3} day^{-1} in all treatments, with no significant effect of treatment, i.e. halving the residence time approximately halved the efficiency, but approximately twice as much nitrogen was introduced, so the amount removed was the same. Desiccation increased the generation of N_2O, CH_4 and CO_2. Little reduction of sulfate (SO_4^{2-}) was observed. This study emphasised the importance of monitoring both gas-phase and dissolved gases, particularly in designs such as this where aerobic and anaerobic processes can occur simultaneously.

Introduction

Denitrifying bioreactors can be deployed to reduce the concentration of nitrate N in water leaving farms (Blowes et al., 1994). NO_x-N is considered a major pollutant in the humid wet tropics of north-eastern Australia (Bartley et al., 2017). Denitrification in bioreactors requires low concentrations of dissolved oxygen (DO), high availability of carbon, and adequate hydraulic residence time, nitrate concentration and water temperature. Environmental conditions in the Wet Tropics may limit bioreactor effectiveness and increase the risk of pollutants being generated.

Pollutants are more likely to form under conditions that deviate from the optimum for denitrification**.** Sub-optimal conditions include residence times of 3h or less, influent NO_x-N concentrations below 3mg L^{-1} and DO concentrations above 2mg L^{-1} (Addy et al., 2016; Ashok and Hait, 2015). The potential pollutants generated include nitrous oxide (N_2O), nitrite (NO_2^-) and ammonium (NH_4^+), carbon dioxide (CO_2) and methane (CH_4), hydrogen sulphide (H_2S) and methyl-mercury (MeHg) (Easton et al., 2015; Addy et al., 2016). However, it should be kept in mind that the greenhouse gases mentioned, especially CO_2 and N_2O, will eventually be generated from the nitrate and woodchips whether or not they meet in bioreactors.

Hydraulic residence time may be limited in the tropics due to high flow rates. Hydraulic residence time is generally considered more important than influent nitrate concentration in terms of its effect on denitrification. Denitrification rates are controlled by Michaelis-Menten kinetics – these describe N removal as following zero-order kinetics when the concentration of influent NO_3-N is greater than the half-saturation constant (K_m), and first-order when influent NO_3-N is lower than K_m (Ghane et al., 2015). Most bioreactors receive NO_3-N concentrations greater than the K_m of denitrifying bacteria, thus operationally they are zero-order (Barton et al., 1999; Schipper et al., 2010). Therefore

hydraulic residence time governs the reaction rate rather than influent NO_3-N concentration (Halaburka et al., 2017; Warneke et al., 2011; Schmidt and Clark, 2013). Optimal residence times are usually defined as those between 6 and 8 hours in length (Addy et al., 2016).

The low DO concentrations necessary for denitrification are achieved primarily through saturation, but highly variable hydrology and water table depth in the tropics (Desper et al., 2015) may make it difficult to maintain saturation. Fluctuating water table depth may lead to partial desiccation of the woodchip volume. The effects of desiccation on bioreactor performance and pollutant generation are not well studied. Research to date has shown that dry periods have both increased (Christianson et al., 2017; Hua et al., 2016; Maxwell et all., 2019; Woli et al., 2010) or decreased nitrate removal efficiency (Weigelhofer and Hein, 2015; Lee et al., 2013). The varied effects may be due to the antagonistic processes of 1) increased DO (denitrifiers prefer to utilise oxygen) and, 2) increased DOC availability (from aerobic breakdown) that is more available to denitrifiers (Weigelhofer and Hein, 2015; Maxwell et al., 2019).

The aim of this thesis was to determine the effect of hydraulic residence time (3- and 6-hour) and dessication (3 days and 13 days, both with a 6-hour residence time) on NO_x-N removal and pollutant generation in denitrifying bioreactors subject to 'approximate' field conditions in the humid wet tropics of northeast Australia. These conditions included low influent NO_x-N concentration (3 mg L^{-1}) and high ambient temperature (~25 °C).

Materials and methods

The experimental set up involved twelve woodchip-filled bioreactor microcosms (PVC columns) subjected to four treatments, 3 columns each (Table A4.1). The 6-h hydraulic residence time was chosen as 'optimal', as indicated by previous studies (Hoover et al., 2016; Greenan et al., 2009). The 3-h hydraulic residence time was chosen to represent high flow conditions likely to be encountered in the field under high flows. The two drying intervals (3-day and 13-day) were selected to encompass a range of dry intervals tested in previous studies (4 days and 3 weeks, respectively) (Christianson et al., 2017; Weigelhofer and Hein, 2015). Previous studies have shown that nitrate removal rates are similar in microcosm studies to those in field bioreactors (Addy et al., 2016).

Table A4.1. Experimental treatments (each replicated in 3 columns).

Acronym	Description
3h-W	3-hour hydraulic residence time with continuous throughflow
6h-W	6-hour hydraulic residence time with continuous throughflow
6h-SD	6-hour hydraulic residence time with several short (3-day) dry periods
6h-LD	6-hour hydraulic residence time with several long (13-day) dry periods

Before the nitrate was introduced to the columns they were flushed with tap water from day 1 to 13. Between days 5 and 11 approximately 40 pore volumes were eluted for the 6-h columns, and 80 for the 3-h columns. Nitrate was added to the influent water from day 14 until the end of the experiment. All treatments were subject to the same influent nitrate concentration and air temperatures (23-25°C), with water temperatures ranging from 23-25.8°C. Desiccation of the columns was achieved by shutting off inflow and allowing the columns to drain for at least 12 hours. All dry periods were separated by a wet period of at least one week.

The columns were 15 cm in diameter, filled with woodchips to a height of 72 cm, with the outlet hole at a height of 77 cm. The woodchips were from the batch used for the bioreactor field trials being run concurrently in the Babinda Swamp Drainage Area (BSDA). The chips were sieved to remove the fine fraction (< 5 mm). The resultant woodchips ranged in size from 1.4-9.0 cm in length,

0.1-2.9cm in breadth and 0.1-0.6cm in thickness; the median dimensions were 3cm x 0.9cm x 0.2cm (5.4cm^3, n=30). Bioreactor volume (chips and pores) was 12.74 L and the mass of chips in each was 5.72 kg. Drainable (effective) porosity was 0.47, lower than the typically cited value of 0.70 (Woli et al., 2010; Schipper et al., 2010; Addy et al., 2016) but similar to the value of 0.50 reported by Hua et al. (2016) and 0.45 reported by Ghane et al. (2014).

The influent water entered the bottom centre of the column, driven by gravity, with a pressure head of 7 cm (Figure A4.1). The average adjusted flow rate for each treatment was 32 mL min^{-1} for the 3h-W treatment and 15−16 mL min^{-1} for the 6h treatments. The composition of the influent water was designed to approximate that of water collected from drains in the BSDA. Mean solute concentrations were 2.79 mg NO_3-N L^{-1}, 0.002 mg NO_2-N L^{-1}, 0.04 mg NH_4-N L^{-1} and 4.91 mg SO_4-S L^{-1}. That composition was achieved by continuously mixing tap water with a concentrated solution of $MgCl_2.6H_2O$, $Ca(NO_3)_2.4H_2O$ and K_2SO_4.

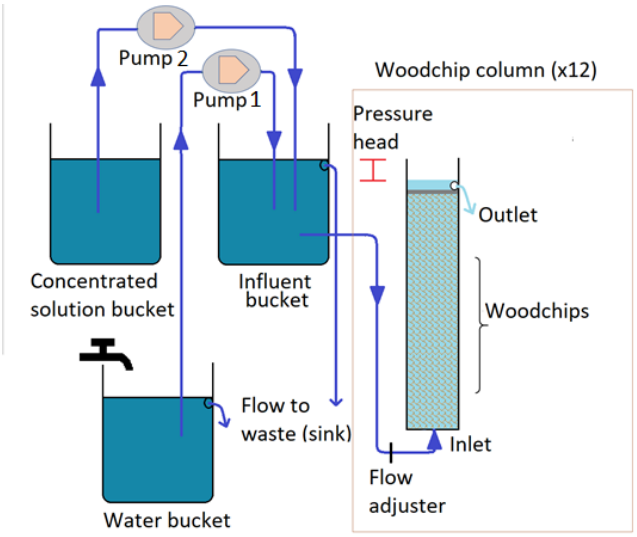

Figure A4.1. Diagram of the experimental apparatus.

Water samples were taken once per week from the effluent of all columns that were flowing (i.e. not desiccated) and from the influent. The samples were analysed for NO_3^-, NO_2^-, NH_4^+ and SO_4^{2-} using standard methods. UV absorbance (254 nm), taken as a proxy for DOC concentration, was measured with a scan probe. Dissolved oxygen concentration (DO) was measured with a ProSens Microx 4 DO probe. pH and electrical conductivity (EC) were measured with Hanna hand-held probes. Phosphorus concentrations of the effluent water was analysed but all measurements were below the detection limit of the instrument.

The mean mass of NO_3-N entering each column over the course of the experiment was 8.39 g in the 3h-W, 2.84 g in the 6h-W, 3.11 g in the 6h-SD, 2.62 g in the 6h-LD. Cumulative amounts of solutes discharged from each column from day 15 to day 81 were calculated by summing the amounts discharged between each sampling event, assuming a linear change in concentration from one sampling event to the next. Dry days were not included in the calculations.

Gas emissions were also measured once per week, but additional measurements were made on dry columns just prior to re-saturation. Gaseous emissions of N_2O, CH_4 and CO_2 were determined by closing the column headspace for a short time (10 minutes) and measuring the increase in gas concentration over time with an Innova Photoacoustic Field Gas Monitor. Caution is advised in

interpreting the gaseous CH_4 emission results as they are likely underestimated because CH_4 was often below the detection limits of the Innova. Volatilisation loss of ammonium was also disregarded as the pH was always <8 (Gibert et al., 2008). The concentration of dissolved N_2O, CH_4 and CO_2 in effluent and influent were measured by headspace gas chromatography. The amounts of N_2O, CO_2 and CH_4 generated were calculated by summing gaseous emissions and emission of dissolved gas in the effluent. A linear change in dissolved concentrations and gaseous emissions between sampling times was assumed.

Removal efficiency was calculated as the cumulative total amount of N lost (presumed denitrified) as a proportion of the total influent N (Rosen and Christianson (2017). Nitrogen removal rates were calculated as the cumulative removal of N (influent-effluent) in grams per m^3 of the entire saturated volume of the column per saturated day, for the entire duration of the 66-day experiment (66 for the 3h-W and 6h-W, 39 for the 6h-SD and 52 for the 6h-LD). Effects of the treatments were analysed by one-way ANOVA.

Results

Nitrate N concentration was reduced in all the treatments (Figure A4.2). The N removal efficiency was strongly influenced by hydraulic residence time but not by desiccation, being >0.90 in all the 6h treatments and 0.51 in the 3h-W treatment (Table A4.2). However the nitrate removal rate was unaffected by the treatments (Table A4.2). The mean effluent NO_3^- concentration was highest in the 3h-W treatment (1.13 mg L^{-1}) and much lower (<0.09 mg L^{-1}) in all the other treatments. Desiccation slightly lowered effluent NO_3-N concentrations relative to the 6h-W treatment (Figure A4.2). The concentration of nitrite in effluent was low (<0.06 mg L^{-1}) and variable, with generally no significant effect of treatment. The concentration of NH_4-N in effluent increased steadily over the initial month and then declined; the effluent had a higher concentration of ammonium than the influent across all treatments for the entire 81-day period (Figure A4.3). Dissolved nitrous oxide concentrations were generally low for all treatments except the 6h-LD (Figure A4.4). The 6h-LD generally spiked in N_2O emissions immediately following re-wetting before declining by the following sampling day, with the pattern strengthening over the course of the experiment (Figure A4.4).

Dissolved oxygen concentration and pH both decreased as water passed through the columns (Figure A4.5). At 26 cm up the column, DO was <1.7 mg L^{-1} in all the 6h treatments and 2.37 mg L^{-1} in the 3h-W treatment. Effluent DO and pH were little affected by treatments except that both tended to be reduced following dessication (Figure A4.5). While effluent pH consistently decreased following drying, it increased to near pre-dry levels by the following sampling event (Figure A4.6).

Mean effluent UV absorbance values in the dry treatments were approximately double those of the continuously flowing treatments (Figure A4.7). However, residence time did not affect mean UV254; the average difference between the 6h-W and 3h-W was not significant. UV absorbance decreased slightly between days 11 and 40 across all treatments. The DOC of the 3h-W was already different from the other treatments at day 11, likely due to the extra flushing that had taken place between day 1 and 11. In both desiccated treatments UV absorbance increased significantly relative to the other two treatments immediately after wetting but the effect had diminished by the following sampling day (one week later) (Figure A4.7).

Sulfate concentration was not reduced in effluent, except immediately following drying periods (Figure A4.8).

Dissolved CO_2 concentrations were lower in the 3h-W treatment than the 6h-W treatment (Figure A4.9). Dissolved carbon dioxide concentration increased initially following drying and decreased to levels similar to the 6h-W treatment by the following sampling day. The effect of drying on carbon

dioxide concentration did not appear to diminish over subsequent drying treatments in either the 6h-SD or 6h-LD.

Dissolved methane concentrations were higher in effluent than influent in all treatments (Figure A4.10). Methane concentrations were lowest in the 3h-W treatment. They increased initially after re-wetting, particularly in the 6h-LD treatment, and decreased to pre-dry levels by the following sampling day. The effect of drying on dissolved methane concentration appeared to diminish after the initial drying treatment for both the 6h-SD and 6h-LD treatments. However, the third dry period in 6h-LD treatment was still resulting in a significant increase in methane concentration, whereas the third and fourth short dry treatments did not elicit a significant response in methane concentration.

Total greenhouse gas emissions (sum of gaseous and dissolved emissions) were strongly affected by dessication but not hydraulic residence time, and CO_2 was the largest contributor (Table A4.3). The 6h-LD emitted significantly more CO_2 and N_2O than the 6h-SD treatment, but CH_4-C emissions did not differ significantly between the two.

Table A4.2. Removal of N from water passing through the columns over the course of the experiment, given as the mean proportion of influent N for each treatment, and the amount of N removed per volume of woodchip-filled column per day. Values are mean ± standard deviation. Means accompanied by the same letter are not significantly different (Tukey's HSD p<0.05). Standard deviations follow the means in parentheses.

Treatment	N removal efficiency (g N removed/g influent N)*	NO_3^- removal rate (g N/m^3/ wet day)*
3h-W	0.514 ± 0.058 [a]	5.20 ± 0.56 [a]
6h-W	0.919 ± 0.034 [b]	4.82 ± 0.30 [a]
6h-SD	0.957 ± 0.004 [b]	4.56 ± 0.05 [a]
6h-LD	0.906 ± 0.023 [b]	4.85 ± 0.40 [a]
P value	< 0.001	0.298
F (3,8)	153.359	1.455
w^2	0.97	0.102

*Calculated only for wet days between days 15 and 81 (3h-W and 6h-W= 66 days total, 6h-SD=52 days and 6h-LD=39 days)

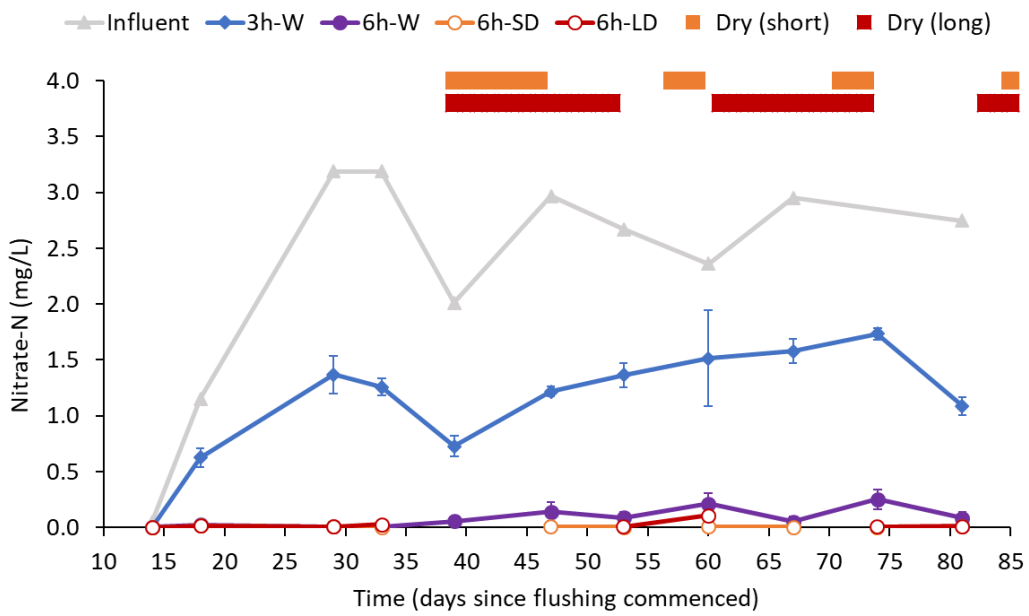

Figure A4.2. Mean (n=3) NO_3-N concentration in the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

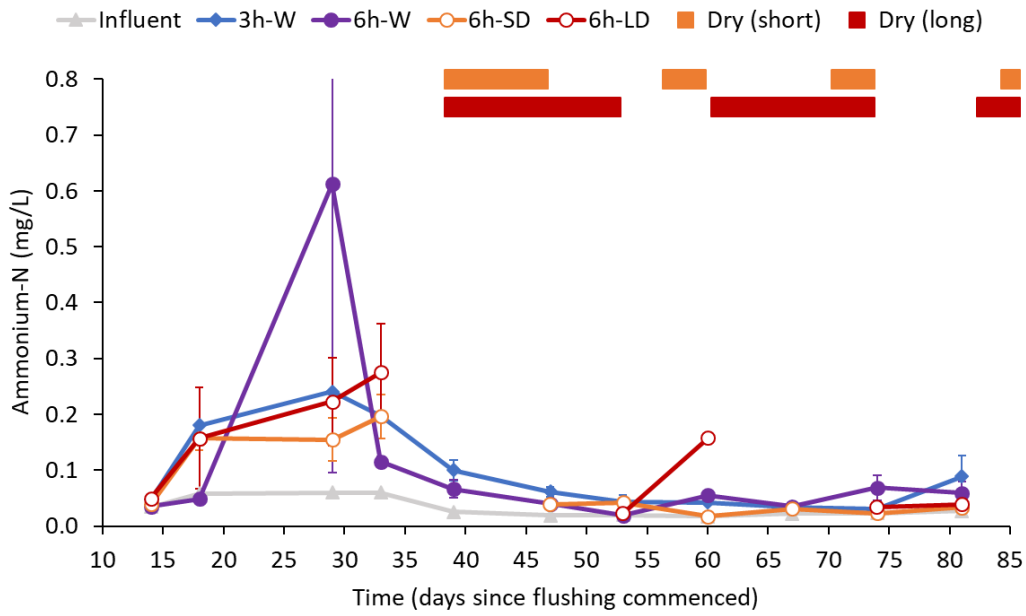

Figure A4.3. Mean (n=3) NH_4-N concentration in the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

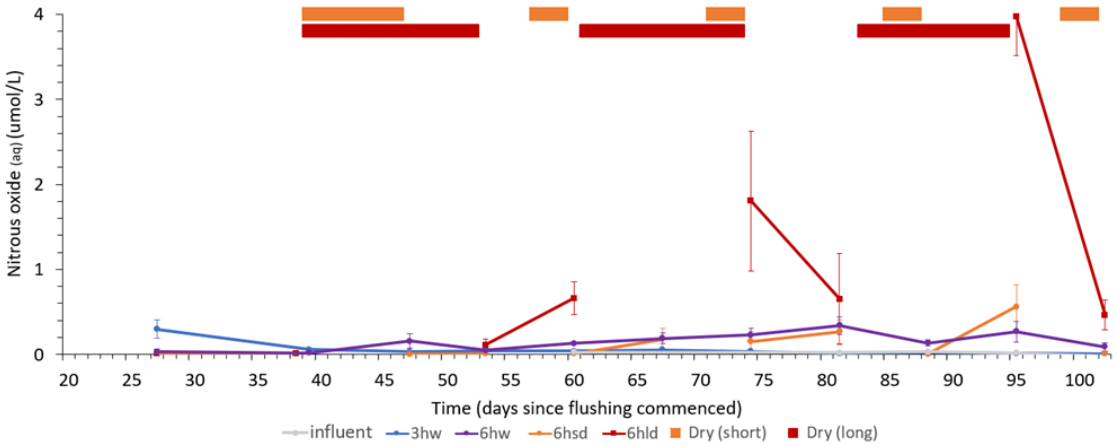

Figure A4.4. Mean (n=3) dissolved N_2O concentration in the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

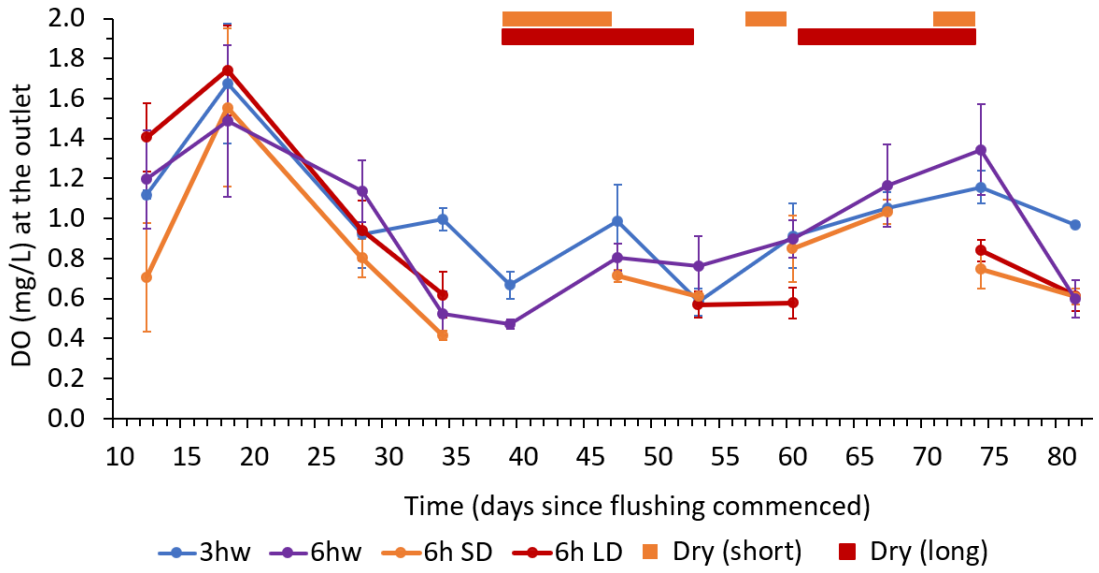

Figure A4.5. Mean (n=3) dissolved oxygen (DO) concentration of the effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

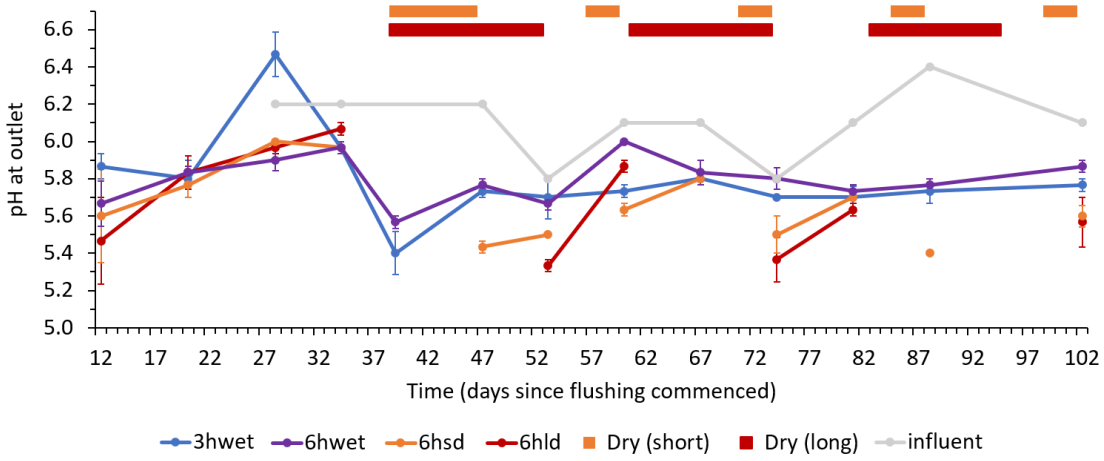

Figure A4.6. Mean (n=3) pH of the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

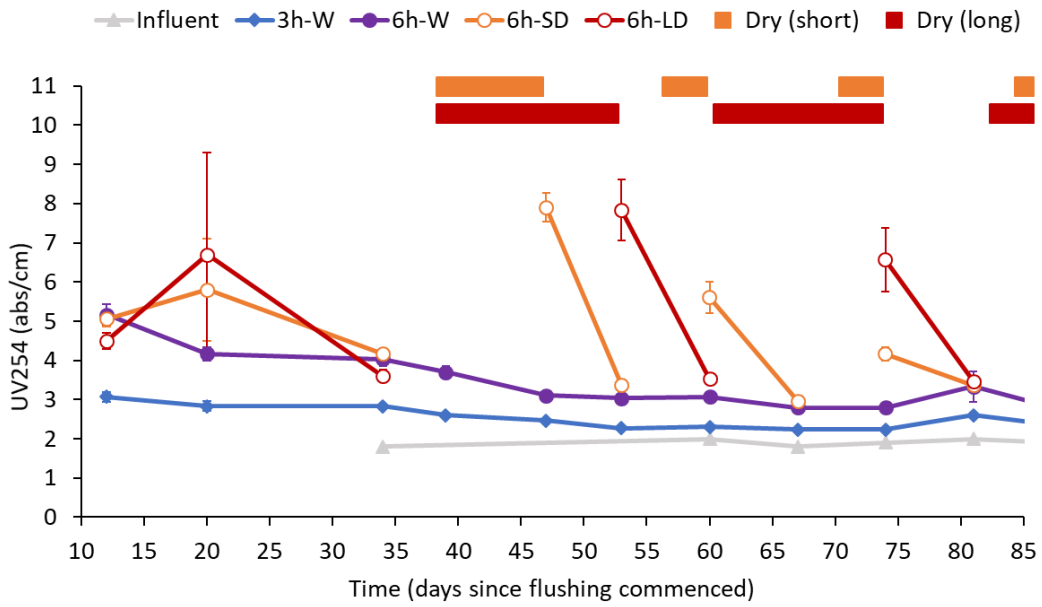

Figure A4.7. Mean (n=3) UV absorbance (UV254) in the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

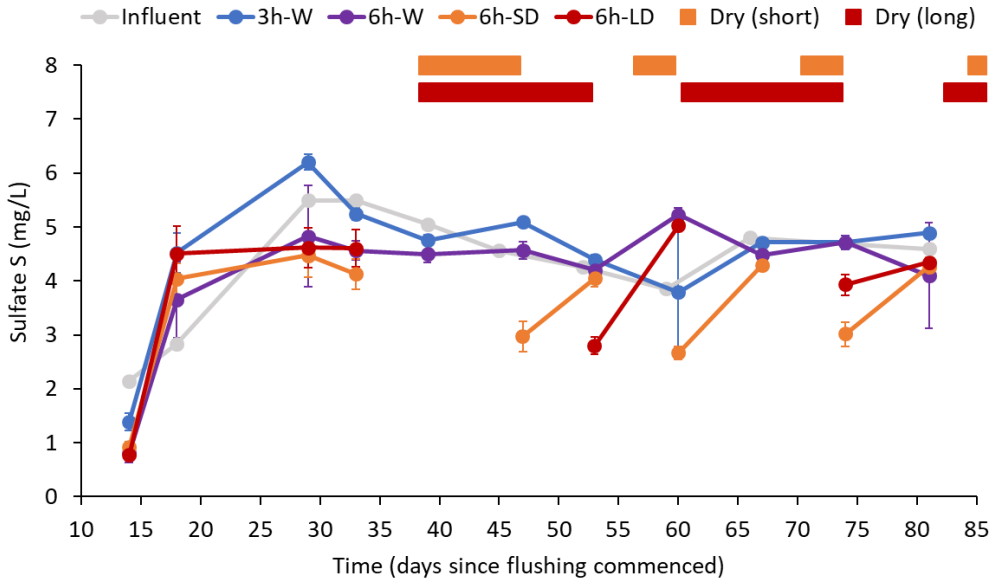

Figure A4.8. Mean (n=3) SO_4^{2-} concentration in the influent and effluent from each treatment over time. Solute dosing commenced on day 14. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars represent dry periods.

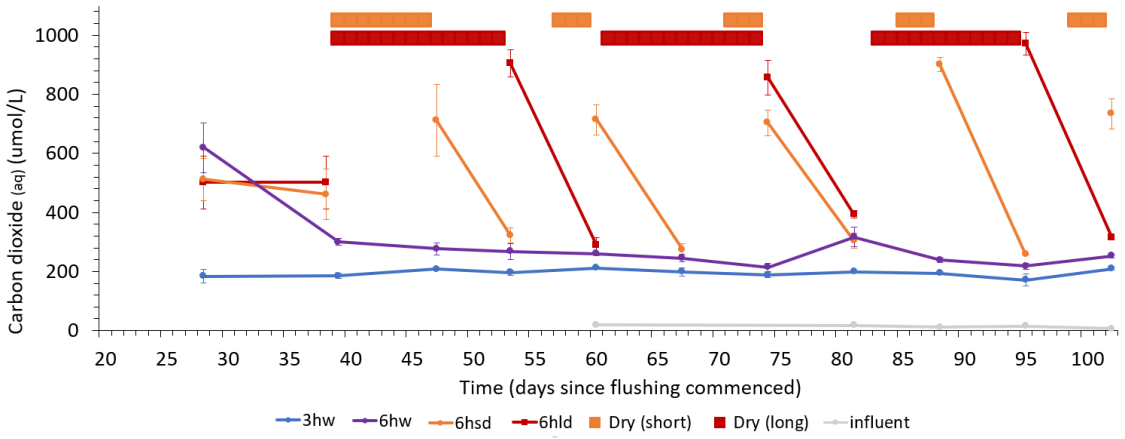

Figure A4.9. Mean (n=3) weekly dissolved CO_2 concentration in the influent and effluent from each treatment over time. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars at the top represent dry periods.

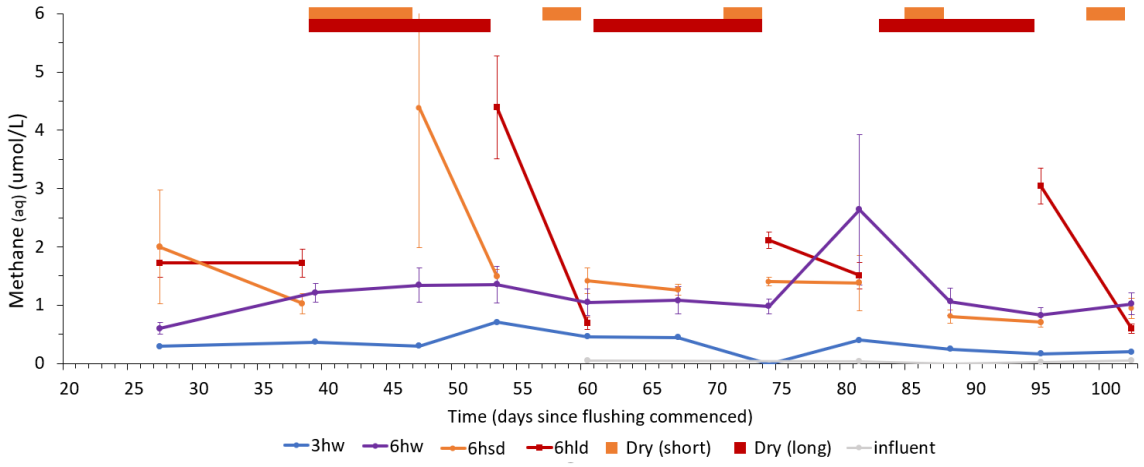

Figure A4.10. Mean (n=3) weekly dissolved CH_4 concentration in the influent and effluent from each treatment over time. Error bars represent standard error of the mean. Treatments that underwent desiccation have gaps in the time series. Horizontal bars at the top represent dry periods.

Table A4.3. Mean emission rates (n=3) of greenhouse gases (dissolved plus gaseous) from each column over the 66-day experimental period ± standard deviation. Percentages indicate the proportion of the total emission that was in the gas phase.

	Gas emission rate (mg/day, gaseous + dissolved)		
	CO_2-C	CH_4-C	N_2O-N
3h-W	92.4 ± 5.2 [a] (47% gas)	0.086 ± 0.043 [a] (37% gas)	0.357 ± 0.093 [ab] (60% gas)
6h-W	86.0 ± 12.0 [a] (42% gas)	0.094 ± 0.032 [a] (18% gas)	0.214 ± 0.054 [a] (66% gas)
6h-SD	314.0 ± 96.0 [b] (79% gas)	0.187 ± 0.143 [ab] (35% gas)	0.530 ± 0.146 [b] (93% gas)
6h-LD	704.3 ± 60.4 [c] (90% gas)	0.436 ± 0.173 [b] (70% gas)	1.575 ± 0.118 [c] (87% gas)
p-value	<0.001	0.019	<0.001

Discussion

Hydraulic residence time

The shorter residence time of 3h in this study yielded a significantly lower removal efficiency (loss in N per unit of influent N) than the 6h residence time, i.e. 51% versus 92% (Table A4.2), as has previously been shown (Gibert et al., 2008; Healy et al., 2012; Coleman 2017). Although efficiency of the 6h-W treatment was slightly less than double that of the 3h-W, the 3h-W was flowing twice as fast, resulting in similar removal rates between the two hydraulic residence times. These results support the notion that differences in nitrogen removal efficiency mostly reflect differences in hydraulic flows, rather than in N removal efficacy (Healy et al., 2012).

Nitrate-N removal rates were ~5 g-N m^3 day^{-1} across all treatments, within the range generally reported for bioreactors. A recent meta-analysis found that the 5th and 95th percentiles for reported nitrate removal rates were 2.9 and 7.3 g-N m^3 day^{-1} (Addy et al., 2016). Although N removal rates might be expected to be influenced by influent NO_x-N, hydraulic residence time and temperature (Schmidt and Clark, 2013; Hoover et al., 2016), in practice, removal rates can be unpredictable and relatively insensitive to these variables. Some previous studies have shown that shorter residence times have higher removal rates (Robertson and Merkley, 2009; Lepine et al., 2016), but others have reported little or no effect of hydraulic residence time on removal rates, similar to our results (Hoover et al., 2016; Hua et al. 2016; Coleman, 2017; Kouanda, 2017).

Zero-order kinetics would result in a halving of nitrate loss with a halving of residence time, which is more-or-less what we found. Therefore the results correspond with Halaburka et al.'s (2017) conclusion, based on laboratory work with a range of influent concentrations and hydraulic residence times, that denitrification in bioreactors is essentially a zero-order reaction. Our results were not strictly zero-order, probably because nitrate concentration became limiting at the higher residence rate, i.e. almost all of it was removed, so further increasing residence time would not have any effect on nitrate removal. Many studies have suggested a transition from first-order to zero-order kinetics at NO_3-N concentrations of approximately 3-10 mg L^{-1} (Nordström and Herbert, 2016; Ghane et al., 2015; Easton et al., 2015; Hua et al., 2016). However, in our study the transition appeared to occur at much lower concentrations.

Dissolved oxygen may have inhibited denitrification in the 3h-W treatment. In most columns DO concentrations remained below 1 mg L^{-1}, significantly less than the 2 mg L^{-1} threshold thought to inhibit denitrification (Ashok and Hait, 2015). However, mean DO in the 3h-W treatment was in the range considered inhibitory at a height of 26 cm (36% of the columns' total length) (Lynn et al., 2015). Inhibition may have occurred either directly via competition between O_2 and NO_3^- as an electron acceptor, or indirectly by stimulating aerobic bacteria that could potentially out-compete denitrifiers for available C (Schipper et al. 2010). The decrease in pH through the columns was presumably due to production of organic acids (Christianson et al., 2017; von Ahnen et al., 2019). Effluent DOC concentration decreased with time in the continuously wet treatments, consistent with a flushing effect (Figure A4.7, Christianson et al., 2017).

The conversion of nitrate into undesirable undesirable products was also influenced by hydraulic residence rate. Production of N_2O was significantly higher in the shorter residence-time treatment (Table A4.3). A significant amount of NH_4-N was produced between 14 and 40 days, with the short hydraulic residence time treatment producing the largest amounts (Figure A4.3). High NH_4-N concentrations at bioreactor start-up have been attributed to high C:N ratios and mineralisation or DNRA (Cameron and Schipper, 2010; Lee et al., 2013). However, it seems plausible that this effect could also be due to NO_x-N loading itself (Grießmeier et al., 2017; Weigelhofer and Hein, 2015). Furthermore, this may explain why some studies have observed higher NH_4-N concentrations at

lower residence times, where NO_x-N removal is generally less complete and C:N ratios would be lower (Nordström and Herbert, 2016). However, regardless of the processes involved, NH_4^+-N concentrations were generally low. The concentrations reported here are similar to those observed by Cameron and Schipper (2010) who noted a decrease from 1 mg L^{-1} to 0.03 mg L^{-1} by month 6 in hardwood chips.

Sulfate reduction was not significantly affected by residence time. The short residence times in this study (3h and 6h) presumably prevented SO_4^{2-} reduction, even in the N-limited 6h-W treatment. The kinetics of SO_4^{2-} reduction are such that it takes approximately twice as long as denitrification given the same availability of C (Zhang et al., 2013).

The N_2O-N produced was a small proportion of unrecovered N, which was presumably mostly lost as N_2. Losses as a proportion of unrecovered N in the continuously flowing treatments were similar to results from previous bioreactor experiments; N_2O-N loss as a proportion of unrecovered N was 0.5% in the 3h-W treatment and a 0.3% in the 6h-W treatment. Elgood et al. (2010) and Moorman et al. (2010) reported values of 0.5% and 0.6%, respectively. These values are less than the amount of N_2O-N estimated to ultimately arise from NO_3^- leaving farms, irrespective of its pathway; the IPCC assume that 0.75% of leached nitrate N is eventually converted to N_2O (Mosier et al., 1998).

Residence time did not significantly affect emission of CO_2 or CH_4 (Table A4.3). Carbon dioxide has a net zero impact in terms of greenhouse gas emissions in the context of woodchip bioreactors as the woodchips will decompose to CO_2 whether or not they were in a bioreactor (Schipper et al., 2010; Schipper and Christianson, 2017).

Desiccation

Neither N removal rate nor N removal efficiency were affected by drying (Table A4.2), possibly due to the N-limited conditions associated with the 6h-hydraulic residence time (Hua et al., 2016). In a study that utilised 3-week long episodes of desiccation, production of NO_3^- was considerable and was associated with a reduction in N removal efficiency (Weigelhofer and Hein, 2015). Desiccation did not lead to increased conversion of NO_3^- to NH_4^+ -N, despite high rates of carbon mineralisation (Figure A4.3). Weigelhofer and Hein (2015) found that desiccation affected NH_4^+ -N release only slightly. As DNRA are obligate anaerobes, they are at a disadvantage relative to denitrifiers in conditions of fluctuating saturation levels (Rivett et al., 2008).

Desiccation of the bioreactors significantly increased N_2O-N emissions. Short-term increases in N_2O have been previously found to occur in soils following drying and rewetting (Davidson et al., 1993; Priemé and Christensen, 2001), but the effects of desiccation on N_2O emission from bioreactors are not well studied. The effect of desiccation on N_2O-N emissions increased with the length of the dry period (Table A4.3). Dry periods of 3-week duration have been found to exert a similar increase in dissolved N_2O-N concentration to the 13-day dry (6h-LD) treatment in this present study (Weigelhofer and Hein, 2015). However, this does not account for the fraction produced as gas, which contributed as much as 93% and 87% of the total emissions for the desiccated treatments in our study (Table A4.3). The availability of oxygen likely played a key role in driving N_2O formation. Sudden decreases in oxygen availability have also been shown to induce larger emissions of N_2O by denitrification in soil (Devêvre and Horwáth, 2000; Priemé and Christensen, 2001). Davidson et al. (2000) found soil water was the most significant control of N_2O emissions whereby at 100% water filled pore space, N_2O emissions from denitrification would be insignificant relative to N_2. The results of this present study fit with this conceptual model as the longer dry period had more time for air to replace water inside the pores between woodchips, resulting in increased N_2O emissions relative to the shorter dry period (Table A4.3).

Changes in oxygen availability and enzymatic activity likely drove the observed increase in N_2O-N following re-wetting over repeated dry/wet cycles. Priemé and Christensen (2001) noted that the enzyme nitrous oxide reductase is less persistent than nitrate reductase. Thus the ability to reduce nitrate is largely conserved over the dry spell but the ability to reduce N_2O to N_2 is quickly lost, resulting in increasing N_2O emissions as the dry period lengthens (an effect that was observed here between the two desiccated treatments) (Table A4.3). This might also explain the pattern of increasing dissolved N_2O concentrations upon re-wetting in the 6h-LD treatment over subsequent dry periods (Figure A4.4). Weigelhofer and Hein (2015) also found spikes in dissolved N_2O following rewetting increased in magnitude over repeated dry/wet cycles.

Desiccation increased mean DOC concentrations. Similarly, Lee et al. (2013) found that the length of the dry period was correlated with DOC leaching intensity. Dry periods likely stimulated aerobic breakdown of the woodchips (Christianson et al., 2017). High DOC leaching following re-wetting occurred in most bioreactor experiments following desiccation (Lee et al., 2013; Abusallout and Hua, 2017; Christianson et al., 2017; Maxwell et al., 2019). There was no relationship between N removal and DOC in these treatments.

Large amounts of carbon dioxide were generated in treatments that underwent desiccation (Table A4.3). There was no evidence of a decrease in C mineralisation over time as a result of drying/re-wetting, as found by Fierer and Schimel (2002). Methane emissions were significantly higher in the 6h-LD relative to the wet treatments, but they were not at environmentally concerning levels (Table A4.3). The maximum dissolved CH_4 concentration 1-day post-saturation in the 13-day dry (6h-LD) of 16 µg CH_4-C L^{-1} was similar to the maximum observed 1-day following a 3-week desiccation in straw reactors (21.9 µg CH_4-C L^{-1}) (Weigelhofer and Hein, 2015). Weigelhofer and Hein (2015) considered these CH_4 emissions to be low.

Lower DO concentrations in the desiccated treatments following re-wetting likely reflected enhanced microbial activity and reducing conditions following rewetting (Song et al., 2010; Robertson et al., 2010; Maxwell et al., 2019). Reductions in DO appear to be rapid in bioreactors, with Maxwell et al (2019) reporting that DO was depleted within one hour of flow resumption. Aerobic decomposition of the woodchips during the dry periods appear to have created a labile pool of microbially available C that denitrifiers utilised upon re-wetting. Conditions one day post-saturation showed strong evidence of nitrate-limitation as oxygen was reduced, SO_4^{2-} was reduced, and pH decreased, in accordance with the findings of others (Lynn et al., 2015; Christianson et al., 2017).

Sulfate reduction following re-wetting provided evidence of increased reducing potentials post-saturation (Figure A4.8). The 6h-SD had a significantly higher SO_4^{2-} reduction efficiency than the 6h-W treatment. Sulfate reduction has been observed to occur in bioreactors under nitrate-limited conditions (Elgood et al., 2010; Schipper et al., 2010; Christianson et al., 2013; Lynn et al., 2015; Lepine et al., 2016; Hua et al., 2016). Sulfate reducers likely inhibited the activity of methanogens, particularly in the 6h-SD treatment, as dissolved CH_4 showed a much more muted response post-wetting when compared to the 6h-LD (Figure A4.10). The gaseous CH_4 time series also displayed a similar trend. Studies have shown that although methanogens can co-exist with sulfate-reducers, they are usually out-competed by them given sufficient SO_4^{2-} availability (Dar et al., 2008; Sela-Adler et al., 2017).

Conclusions

A residence time of 6 hours was sufficient to reduce NOx-N concentrations from 2.8 to <0.5mg L^{-1} (91% removal efficiency) at temperatures between 23-25.8 °C. The removal efficiency of the 3h

hydraulic residence time was approximately half (51%) that of the 6h hydraulic residence time. The rate of N removal in all treatments approximated 5 g m^{-3} day^{-1}, irrespective of hydraulic residence time and desiccation. While N removal rates were likely limited by NOx-N concentration in the 6h treatments, N removal rates in the 3h treatment were probably not. These results indicate that bioreactors operated under tropical conditions can retain good N removal efficiencies and reductions in the concentrations of influent NOx-N even at sub-optimal residence times.

The overall pollutant potential of the bioreactor was lowest with the higher residence time, associated with higher N removal efficiencies and greater conversion of N to N_2, because N_2O was the main potential pollutant. Sulfate reduction was not observed to occur in either of the treatments that was flowing continuously, despite the near-complete removal of influent N in the 6h treatment. Pollution potential was higher in the desiccated treatments than the continuously flowing treatments, especially N_2O emissions. However, the ratio of N_2O to N_2 emissions was lower than expected in the field without bioreactors. Methane emissions andSO_4^{2-} reduction were also increased by desiccation but relatively low.

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Appendix 5: Denitrification Potential of Soils under Sugarcane in the Wet Tropics, and Woodchips

Results of a project carried out by Maureen Masson at JCU in April-August 2019 under the supervision of Paul Nelson and Alex Cheesman, in partial fulfilment of her Masters at AgroSup Dijon.

Introduction

Denitrification occurs in the soil profile, particularly when and where water content is high (>~60% water-filled pore space) and there is abundant organic matter and nitrate. Soils of the Babinda Swamp Drainage Area (BSDA) are predominantly Organosols (peats), so potential denitrification rates may be high. The aims of this work were to a) determine the denitrification potential of soils from Wet Tropics sugarcane growing areas, including the BSDA, b) determine the main factors controlling denitrification potential of these soils, and c) determine the denitrification potential of woodchips.

Methods

Fifteen soil sampling sites were chosen to represent the variety of soils cultivated to sugarcane in the Wet Tropics region (Table A5.1). To choose the sites the area of each soil association under sugarcane was first determined in ArcMap using soil survey data (Cannon et al., 1992; Murtha, 1986; Murtha et al. 1996) and land use data (QLUMP). The soil associations with the largest area of sugarcane were selected, ensuring that a range of parent materials, textures and organic C values (from published values) was covered. Sites were then chosen that were well within a polygon of each chosen map unit and well within a sugarcane paddock. They included the two bioreactor trial sites in the BSDA in this project and the site of a long-term nitrogen management experiment (Masters et al. 2017). At each site a composite soil sample was prepared from 16 samples (4 per row, in 4 rows) taken from the side of the 'hill' at a depth of 0-25 cm. Samples were also taken from 25-50 cm depth at three sites with peaty soils. The soil samples were dried at room temperature, homogenised and sieved. They were then analysed for total C and N content, extractable NH_4^+ and NO_3^- content and pH.

Potential denitrification was measured under anaerobic conditions and unlimited nitrate supply, according to the acetylene block method of Tiedje (1994). That involved measurement of the N_2O generated during a one-hour incubation of a soil slurry (50 g oven-dry equivalent soil with 50 mL of solution added containing 100 mg L^{-1} of N, as KNO_3, with a headspace of 90% N_2 and 10% C_2H_2) in 500-mL jars, incubated at 23°C for one hour in triplicate. Prior to the assay each soil sample was moistened to approximately 60% of water holding capacity for one week and then incubated under assay conditions for 30 minutes. Potential denitrification of a woodchip sample was measured using the same procedure, except that 10 g oven-dry equivalent was used and the pre-incubation was in a saturated state, with added nitrate. The woodchip sample was mixed hardwood (eucalypt) from southeast Queensland, taken from the batch used for the BSDA bioreactor trials.

Results

Mean denitrification potential of the soils ranged from 4.4 to 291.1 µg N kg^{-1} soil h^{-1}. It was positively related to total C content (p=0.051), total N content (p=0.007) and extractable nitrate (p=0.013), but not to pH (Figures A5.1 and A5.2). Denitrification potential of the woodchips was 1,124.1 µg N kg^{-1} h^{-1}. Assuming a bulk density of 1100 kg m^{-3} for the soils and 449 kg m^{-3} for the

woodchips (from Donovan thesis, Appendix 4), the potential denitrification rates were 0.12 to 7.69 g N m^{-3} day^{-1} for the soils and 12.11 g N m^{-3} day^{-1} for the woodchips.

Table A5.1. Soil sampling sites. Soil associations and Orders are from Cannon et al. (1992), Murtha (1986) and Murtha et al. (1996). All samples were taken from 0-25 cm depth only, except those mentioned in the footnotes.

Soil association	Soil Order & sub-Order	Parent material	Coordinates
Innisfail	Brown Dermosol	Alluvium (well drained)	16°52.976'S, 145°41.839'E
Liverpool	Orthic Tenosol	Alluvium (well drained)	16°51.037'S, 145°43.819'E
Mossman	Yellow Dermosol	Alluvium (well drained)	17°07.869'S, 145°50.680'E
Tully	Brown Dermosol	Alluvium (well drained)	17°16.927'S, 145°55.501'E
Virgil	Red Kandosol	Alluvium (well drained)	17°05.122'S, 145°48.614'E
Babinda[1]	Fibric Organosol	Alluvium (poorly drained)	17°22.920'S, 145°56,341'E
Babinda[2]	Fibric Organosol	Alluvium (poorly drained)	17°21.826'S, 145°57.011'E
Bulgun	Grey Dermosol	Alluvium (poorly drained)	17°08.541'S, 145°50.977'E
Bulgun[3]	Redoxic Hydrosol	Alluvium (poorly drained)	17°44.779'S, 146°03.024'E
Timara	Redoxic Hydrosol	Alluvium (poorly drained)	17°09.528'S, 145°53.282'E
Eubenangee	Red Ferrosol	Basaltic	17°26.719'S, 145°57.026'E
Pin Gin	Red Ferrosol	Basaltic	17°04.153'S, 145°46.023'E
Thorpe	Brown Kandosol	Granitic	17°10.666'S, 145°66.286'E
Lugger	Oxyaquic Hydrosol	Granitic	17°08.660'S, 145°50.786'E
Clifton	Yellow Kandosol	Metamorphic	17°02.475'S, 145°46.650'E

[1] Farm 1 bioreactor trial site, samples taken from 0-25 and 25-50 cm depth.
[2] Farm 2 bioreactor trial site, samples taken from 0-25 and 25-50 cm depth.
[3] Nitrogen fertiliser management trial site described by Masters et al. (2017), samples taken from 0-25 and 25-50 cm depth.

Discussion

As expected, the potential denitrification rates of the soils, under anaerobic conditions and with unlimited nitrate supply, was related to the organic matter content of the soils. However, there was considerable unexplained variability, which may be related to the nature of the microbial populations. When considered on a volumetric basis the denitrification potential of the soils ranged from 1 to 64 % of that of the woodchips. Therefore, passage of water through soils with high organic matter content may result in significant denitrification relative to passage through woodchip bioreactors. However, the potential denitrification rates measured here are at the upper end of those possible in soil profiles of the region because they are topsoils; potential denitrification rates would decline with depth in concert with organic matter content.

It is worth noting that the potential denitrification rate of the woodchips, measured under fully anaerobic conditions with ample nitrate (although at slightly lower temperature than in the field), was similar to that measured in bioreactors in the field with much lower nitrate concentrations. This indicates that nitrate concentration is not limiting N removal rate in bioreactors in the field.

Conclusions

Topsoils from the main soil types cultivated to sugarcane in the Wet Tropics, especially the peats of the BSDA, have significant denitrification potential, which is related to their organic matter content. The denitrification rate of woodchips measured under optimal conditions in this work (although at slightly lower temperature than in the field) was similar to that measured in bioreactors in the field.

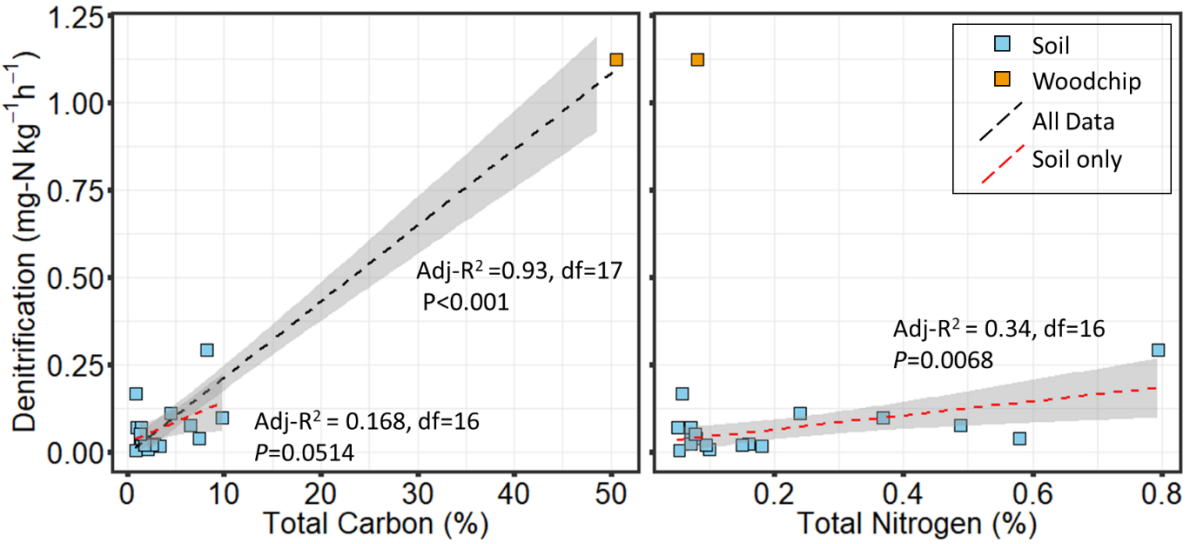

Figure A5.1. Potential denitrification rate of soils and woodchips in relation to total C and N content.

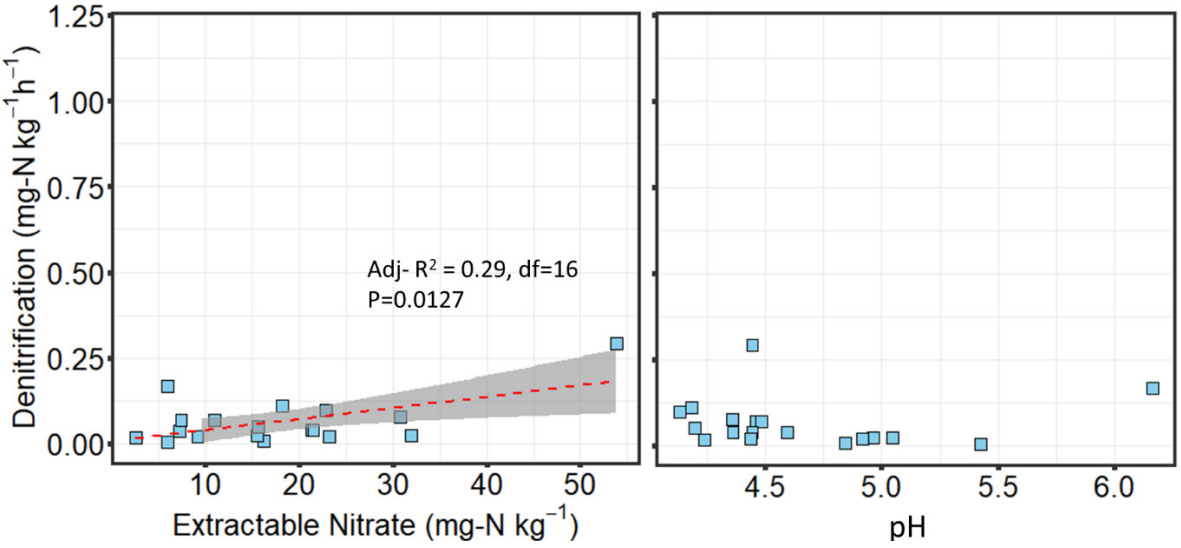

Figure A5.2. Potential denitrification rate of soils in relation to extractable nitrate content and pH.

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Appendix 6: Stakeholder Engagement Component: Potential Interest or Concerns With Denitrifying Bioreactor Technology

This is a paraphrased version of Jai Kaartinen-Price's minor Masters project thesis, 'Barriers and opportunities for denitrification bioreactor adoption by cane farmers in the Wet Tropics, Australia', College of Science and Engineering, James Cook University, June 2019.

Background

Industrial agriculture seeks high yields per unit of land, which implies high inputs of pesticides and fertilisers. However, these chemicals often create negative externalities (e.g. water pollution) to nearby environments the costs of which are typically borne by broader society (Beaudoin et al. 2005).

For crops such as wheat, 30-50% of the nitrogen (N) applied to crops is exported with the crop from the field (Chen et al. 2008; Ladha et al. 2005). Sugar cane production has a typically better N efficiency of approximately 60% (Otto et al. 2016). However, this implies 40% of N is lost from the soil-plant system through various mechanisms e.g. 3-16% of total applied N may be lost as run-off (Thorburn et al. 2011). This run-off inevitably becomes diffuse water pollution.

In the Great Barrier Reef (GBR) lagoon, estimates show an increase of over 5 times pre-European settlement N levels are in the environment (Kroon et al. 2012). The increases in N levels in rivers and coastal systems, when in sufficient quantities, can result in eutrophication, Harmful Algal Bloom (HAB) events, reduced coral fertilisation, increased Crown Of Thorns Starfish (COTS) larval survival and increased algal growth (Bell 1992; Fabricius et al. 2010; Harrison and Ward 2001; Heisler et al. 2008).

The Reef 2050 Long-Term Sustainability Plan (Reef 2050 LTSP) was released in 2015 to address issues surrounding the protection of the GBR resources and values. A revised version – the Water Quality Improvement Plan 2017-2022 (WQIP 2017-2022) – was released in 2018 after unprecedented damage was caused to the reef system over 2016-2017 through bleaching, cyclone damage and COTS damage (GBRMPA 2015). The WQIP 2017-2022 specifies the reduction of N, P, sediment and pesticides in the GBR system. N is one of the targets; specifically, a 20% reduction in anthropogenic end-of-catchment particulate nutrient load and a 60% reduction of anthropogenic end-of-catchment Dissolved Inorganic Nitrogen (DIN) loads by 2025 (AustralianGovernment 2017).

Best Management Practices (BMPs) refers to an industry led accreditation program operating with the Queensland Government (BMP 2019). However, BMPs for sugarcane production, on their own, do not appear likely to reach the targets set by the Australian Government within the allocated time frames (Kroon et al. 2016; Wegscheidl et al. 2015).

Denitrifying Bioreactors (DBs) comprise of a carbon source such as woodchips, placed in a bed or wall to allow affluent water to flow through, and sealed to enable persistent anoxic conditions. Denitrifying bacteria flourish in DBs by converting nitrate (NO_3^-) in the water to

nitrogen gas. A summary paper by Schipper et al. (2010) found that NO_3^- removal generally ranged from 2-22 gN m^{-3} of denitrifying bed per day under 2-20°C, with the bioreactor bed temperature being the strongest attributer to NO_3^- removal rate variation. Considering the success of DBs in other parts of the world there is good reason to assume DBs might prove effective within GBR catchments.

Multiple trials of DBs are underway on farms within the GBR region to evaluate their potential. One trial is taking place on two farms in the Babinda Swap Drainage Area (BSDA) of the Russell catchment, a small sugar cane farming region south of Cairns, Queensland. If DBs in the BSDA prove to be effective in NO_3^- removal then they will be an attractive candidate for land owners and managers to address the nitrogen targets stipulated in the WQIP 2017-2022.

However, for DBs to be regarded as a successful tool for N mitigation in the GBR they must also suit the socio-economic circumstances of farmers in the region i.e. they must suit the needs and wants of land managers and farmers. Further considerations, such as finding suitable land for installation, the funding of installation and their maintenance, communicating the technology to farmers, and getting Natural Resource Managers and farmers to support the technology, are all important.

Considerable research has been carried into the adoption and agricultural innovations and technologies i.e. it is generally well understood (Knowler and Bradshaw 2007; Pannell et al. 2006). For example, Cohen and Levinthal (1990) noted that adoption may emerge when subjects (in this case, farmers) are able "to recognise the value of new, external information, assimilate it, and apply it to commercial ends". Micheels and Nolan (2016) also found social capital and absorptive capacity strongly associated with innovation adoption levels on farms. In short, factors influencing adoption of agricultural conservation innovations have been associated with the following:

- *adoption as a process of learning;*
- *characteristics of potential adopters; and*
- *characteristics of the conservation practice*.

Understanding these factors as they relate to DBs, and potentially additional emergent factors, will give insight into the prospects of DBs being adopted in the Wet Tropics region of Far North Queensland (FNQ).

Socio-economic research questions:

The following research questions were developed to support the social science component of this project:

- What are the barriers and opportunities for farmers in the BSDA and Russell catchment adopting bioreactors?
- How can the adoption of bioreactors on sugar cane farms in the BSDA and Russell catchment be encouraged?

Research approach/methods:

The overall approach to data collection and analysis was qualitative and inductive treating the BSDA as a case study in agricultural technology adoption. Case studies are especially

suited to addressing 'how/why' questions about a contemporary events as opposed to a traditional experiments (Yin 2003). Face to face interviews allow close, investigative research with farmers using the 'general inductive method' (GDM), described by Thomas (2006) as, "a systematic procedure for analysing qualitative data in which the analysis is likely to be guided by specific evaluation objectives". GDM also accommodates unanticipated deviations in the assumed issues. Ethics approval from JCU's Ethics Committee to conduct interviews was granted on Monday the 13th April, 2019 (ethics approval number H7776).

The central themes and approach to data collection are presented in Figure A6.1.

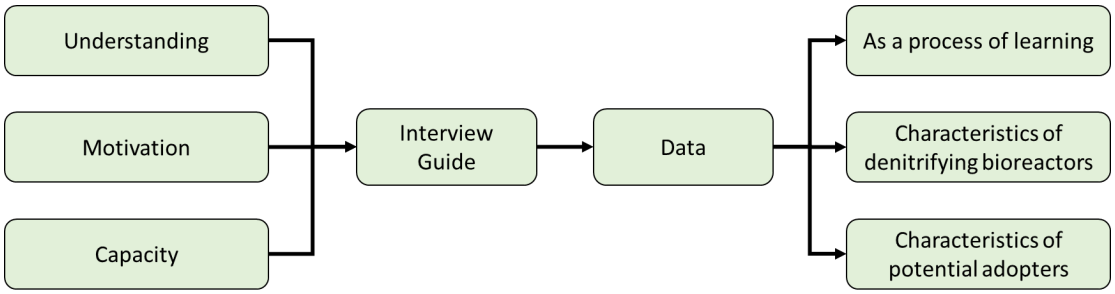

Figure A6.1: Schematic framework guiding data collection (from Macgregor, 2009 and Pannell et al., 2006)

Eight farmers residing in the BSDA and greater Russell catchment were identified for interview based on their local connection and reputation with cane growers and the industry. Collectively, these farmers represented approximately 59% and 15% of farmland for the BSDA and Russell catchments respectively (this estimation was based on farm sizes reported by the farmers and catchment data courtesy of Jaragun Natural Resource Management Pty Ltd). Interviews were audio recorded on a Samsung S9 mobile phone.

While there is no expectation that the findings here are statistically representative, these results should provide a useful insight into the issues and themes relevant to both farmers and other land or industry managers associated with the BSDA, the Russell catchment, and the Wet Tropics of FNQ.

An interview guide was used to ensure assumed central themes were discussed during interviews:

- Perspectives on what is entailed in 'good farming practice'
- Knowledge of nutrient management and current innovations in nutrient input (BMPs) as well as innovations in nutrient output
- Knowledge and level of *Understanding* regarding bioreactors
- Level of *Motivation* regarding bioreactors
- Level of *Capacity* regarding bioreactors
- Level of social connection with farming peers

It is notable that three of the points above were centred on the topics of *understanding*, *motivation* and *capacity* (Figure A6.1). These were based on the sustainable natural resource management framework developed by Macgregor (2009) to guide socio-economic investigations in natural resource management (NERM) situations. In this case study, *understanding* relates to the bioreactor technology itself and its potential to reduce N pollution, *motivation* is concerned about personal factors that drive adoption e.g. farmers' values, attitudes, social standing etc., while *capacity* is concerned with factors that can either encourage/support or stand in the way (barriers) of adoption e.g. financial circumstances, human resources etc.

Following interviews, the audio files were transcribed in verbatim and imported into NVivo 12 software (QSR-International 2018). Transcripts were examined to identify themes relevant to the research questions and themes, the objective being, to build knowledge from the bottom up in order to generate insights inductively (Ormston et al. 2014).

Main findings:

Results are presented in three sub-sections according to those identified in Figure A6.1, i.e. *As a learning process; Characteristics of bioreactors;* and *Characteristics of potential adopters*.

As a learning process:

Most farmers expressed uncertainty towards how effective bioreactors will be in the local context of the BSDA and Russell catchment. A 'lack of understanding of bioreactors' and a 'desire for local results to encourage adoption' were noted in seven out of the eight farmers interviewed. It was also noted that 'locating reactors for maximum effect' and 'creating a standard practice for installation' would be important.

Farmers indicated that the process of learning needs to go beyond those that relate to bioreactor technology i.e. it needs to extend to the broader understanding of both farmers' and land managers' roles in reducing diffuse N pollution. For example, a barrier to understanding the N problem was confusion over issues emerging from sedimentation/siltation and N pollution. This is likely linked to the 'lack of a perceived problem' with N pollution from farms (see below), which was not expressed as a defensive belief but rather, as a conviction based on farmers' observations.

It was also suggested by some informants that urban rivers and the upper catchment forests were also sources of N pollution i.e. cane farmers are not solely responsible. A desire for conclusive evidence of the distribution of N from such sources was expressed.

Characteristics of bioreactors:

Concerns with bioreactor performance were related to the 'efficiency of bioreactors' in removing high concentrations of N. Farmers suggested that both groundwater discharges and first flushes after fertilisation could be the best times for bioreactors to have a positive effect in the study area.

As far as siting bioreactors are concerned, farmers felt there are opportunities to install them in feeding waterways [i.e. those that go into the BSDA] as well as placing them in smaller farms and/or drains.

The opportunity for potential N trading was raised. This briefly came up when discussing carbon trading and while the idea was supported it was also recognised that it may be difficult to quantify. No conclusions could be made about this issue.

The main concerns about bioreactor performance were those associated with their ability to treat water with diluted/low [levels of] N. Most concerns here were based on the high rainfall and therefore high quantity of water that often comes from cane fields in the region. Some farmers also expressed concern over the [perceived] low levels of fertilisers [N] applied to cane in the region. This perhaps implies many farmers do not believe N applications are high enough to be of concern.

The primary barrier around installation of bioreactors was the need for informed advice during installation [location, type, design, etc.]. There was also concern over the potential to disturb acid-sulfate soils during installation, which farmers indicated, should be seriously considered when selecting sites. Tidal influences on drains in the lower part of the catchment were also identified as a potential challenge.

There were also concerns over the inability to place bioreactors in the larger main drains and their likely effectiveness in local soil types. The second point here was based around the lack of groundwater penetration in heavy clay soil, and the heavy carbon in other areas that is believed to limit the release of N.

Characteristics of potential adopters:

The most common positive attitude [mentioned a total of 17 times] was the desire to be 'responsible for problems on my farm' and the desire to 'know how much N is coming off my farm'. There is also a fairly strong desire to work with the government on N pollution [much preferred over 'working under the government']. On this point, the most common negative attitude [mentioned 11 times] was a dislike for increasing government or industry regulation. Some expressed concern over being 'squeezed out of the industry' by increasing regulations. The more cynical informants expressed frustration at 'having the finger pointed at cane growers' and a few felt increasing environmental regulations are about 'securing the green vote'. In general though, most interviewed are open to the idea of trialing bioreactors and there is a general positive agri-environmental orientation among those interviewed.

The ability to contribute 'in kind' for installation of bioreactors (an opportunity) was a common theme of the farmers interviewed with most indicating they have access to earth moving equipment etc.; however, one said he lacked such equipment.

Another common barrier mentioned during interviews was the limited financial capacity farmers have for 'extra expenses' such as bioreactors even though they acknowledged there might be opportunities to obtain financial assistance. Despite this, from the responses on this issue, it appears some farmers may be willing to absorb the costs of installation if bioreactors are proven effective.

Finally, the farmers interviewed speculated about the attitudes of the wider farming population towards bioreactors. Many assume there will be negative views, which it was speculated, may emerge because many desire the familiar and/or they dislike change (i.e. they prefer the way things have always been done).

Implications:

The main discussion points emerging from the farmers' interviews are presented under three sub-headings: *Farmers' knowledge and learning of bioreactors*; *Considerations for bioreactor design and installation*; and, *Likely adopters*.

Farmers' knowledge and understanding of bioreactors:

Bioreactor adoption as a process of learning is still early in its development. Some farmers were quite unfamiliar with the technology and farmers who were familiar had a strong desire for more local results in bioreactor performance and efficiency. By building the understanding of bioreactors and providing easily interpretable results, farmers should be able to better evaluate the technology and make more informed decisions about adoption.

Ensuring that the goals of target farmers are aligned with the benefits provided by bioreactors will be an important part of the adoption process for bioreactors. Farmers must also understand and believe that there is a problem and that they may be a part of its solution, if they are to feel compelled to act.

The lack of a perceived problem was present in interviews with six of the eight farmers in this study. Providing farmers with a robust quantification on the amount of N coming from individual farms and comparing these levels from those of adjacent rainforest, and any nearby urban rivers, seems important to addressing farmers concerns. At the least, quantifying N export from such sources will clarify farmers' contributions to N pollution relative to natural systems and states. Similarly, the mix up of different water quality parameters adds to the confusion. Clarifying the separate issues of sedimentation and N pollution should help overcome this barrier.

Finally, farmers displayed a fairly significant conservationist identity. The process of learning about bioreactors and N pollution in the Russell catchment activate and encourage the conservationist identity within at least some farmers, which should encourage or accelerate adoption. This perhaps implies further effort and resources should be given to publicising bioreactor technology.

Considerations for bioreactor design and installation:

It is assumed by farmers (and evidence supports this) that high levels of N are often found in surface water associated with first flush events and in groundwater/deep drainage water. Farmers in the Russell catchment believe rain events (August-February) and subsurface drains should be targets for bioreactor technology. Future bioreactor evaluations in the catchment should therefore seek to capture N in first flush rain events, especially after fertilising. Continued testing of bioreactors 'in the field' is necessary to determine their potential effectiveness and therefore their potential for widespread adoption.

Despite the relative ease of installing wall type bioreactors, there is evidence from research that groundwater bypasses wall type bioreactors. This also relates to soil types, which was also identified as an issue by farmers in this study. Again, more research is required to determine optimal designs for differing circumstances.

Farmers' knowledge of the presence or absence of acid-sulfate soil on their farms should absolutely be considered prior to installation as well as consideration of the influence of tidal movement on drain flow. Installation of bioreactors in feeding drains/waterways, which carry lower volumes of water, may help ensure a higher portion of total water passes through bioreactors.

Likely adopters:

It should be acknowledged that there is currently no economic benefit for farmers to adopt bioreactor technology. It is also important to ensure, as far as may be possible, that the conservation goals imposed on farmers do not interfere with the successful economics of their business otherwise many farmers/farm families may not be able to adapt to increasing environmental regulation and performance. Neglecting this fundamental issue will seriously impact on willingness to adopt. For example, some notable differences in the business circumstances of farmers were observed in this study. Farmers of larger farms appeared to be most able and willing to absorb costs associated with adoption of bioreactors and they generally spoke favourably about the trials. However, farmers of smaller farms, and particularly farms with more pressing financial circumstances (generally associated with lower cane production i.e. less tonnage per hectare) were more concerned about the prospect of being forced to adopt conservation practices, which they perceived may result in some being forced out of the industry.

Despite the limits on financial capacity, there was also optimism among half the farmers studied in the BSDA for 'in kind' contributions. Most farmers said they would be happy to help out with labour and equipment if they were acknowledged for it. Government subsidies covering 50-75% of installation costs coupled with farmers 'in kind' contributions were a common suggestion. This is perhaps a message for NRM regional groups (e.g. Terrain NRM), relevant State government agencies and/or industry organisations that may be able to campaign for increased funding in support of cane farmers.

Positive attitudes presented largely came from the desire to be responsible for off-site problems emerging from their farms. Farmers evidently attach some pride to the management of their farm and the local environment, wanting to be seen 'doing their part' for the environment and GBR. These positive attitudes suggest there is a strong opportunity for farmers within the Russell catchment, and perhaps wider afield, to strengthen already present conservationist identities. Fear of regulation may be an additional motivating factor here. Further engagement with farmers through a bottom-up style approach to the adoption of bioreactors, may maintain a constructive dialogue between landholders and environmental managers and policy-makers while also encouraging bioreactor adoption. In short, the themes of: desire to 'know how much is coming off my farm', desire to be seen as 'doing my part', and the dislike for 'increasing regulation', all appear to provide an opportunity to strengthen the conservation identity of farmers.

Lastly, consistent with what has been previously found with adoption of new technologies, most farmers in the BSDA are keen to familiarise themselves with bioreactor technology so they may make informed decisions about adoption (Pannell et al. (2006). Previous agri-technology adoption studies suggest that farmers may take up to seven years to adopt from

the first trials (e.g. (Rogers 2004), or similarly, up to eight years for stronger eco-friendly ethic and style of farm management to emerge Macgregor and Warren (2016). Natural resource managers, and those promoting bioreactors, should be realistic about the time it may take for widespread adoption.

Conclusion from farmer interviews:

Agri-technological innovations for reducing DIN loads, such as bioreactors, in the GBR should be carefully evaluated for their performance in proposed localities. Such evaluations should consider the geophysical environment (climate, soils hydrology etc.) but also the local socio-economic contexts and circumstances. The latter is essential if there is to be any prospect of widespread adoption because addressing local barriers and taking advantage of the opportunities that may facilitate adoption are essential.

Three basic socio-economic themes emerged during the interviews with a small group of eight farmers in the BSDA of FNQ regarding adoption of bioreactors: *Farmers' knowledge and learning of bioreactors*; *Considerations for bioreactors' design and installation*; and, *Likely adopters*. By giving due consideration to these three areas, natural resource managers and others concerned with management of diffuse agricultural pollution, especially N, may facilitate the adoption process by addressing: information gathering and dissemination, bioreactor construction and evaluation, and providing motivation and support to target farmers throughout the adoption process.

Since the physical, social and economic context for cane farmers within the Russell catchment is similar to that of farmers in the wider Wet Tropics region it may be reasonably assumed that the adoption themes that emerged in the BSDA may be useful in guiding/encouraging bioreactor adoption and perhaps also other 'progressive' agri-technologies within other catchments of the Wet Tropics. At a minimum, the methods employed in this study many be replicated for investigating the adoption potential of other technologies and/or management practices.

There are still many questions remaining about the effectiveness of bioreactors in differing bio-physical situations. Future research should aim to provide a robust understanding of the capacity of bioreactors to treat N in both groundwater flows and runoff in the Wet Tropics, as well gaining a robust, local understanding of N sources and dynamics which may be translated into information that farmers are able to understand and relate to. In developing these two areas researchers and natural resource managers should be able to overcome the barriers and capitalise on the opportunities for the adoption of bioreactors on cane farms within the Russell catchment and potentially the wider Wet Tropics.

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