Marine Pollution Bulletin xxx (xxxx) 112373

Contents lists available at ScienceDirect

Marine Pollution Bulletin

journal homepage: www.elsevier.com/locate/marpolbul

Land use conversion to improve water quality in high DIN risk, low-lying sugarcane areas of the Great Barrier Reef catchments

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ARTICLE INFO

Keywords:
Constructed wetlands
Ecosystem services
Land transition
Nutrients
Water quality

ABSTRACT

Eutrophication of coastal and nearshore receiving environments downstream of intensive agricultural production areas is a global issue. The Reef 2050 Water Quality Improvement Plan (2017–2022) sets ambitious targets for reducing pollutant loads entering the Great Barrier Reef from contributing agricultural catchments. At a regional scale, the Wet Tropics end-of-catchment target load reduction for dissolved inorganic nitrogen (DIN) is 60% from the 2012–2013 anthropogenic load level. However, not even with the combined efforts of the Reef Regulations (December 2019) mandate and adoption of best practice nutrient management on farm, is it likely that these DIN targets will be reached. Thus, there is a need for innovative and cost-effective approaches to deliver further water quality improvement. Transitioning low-lying, marginal sugarcane land to alternative land uses that require lower or no nitrogen inputs, but still provide farmers with income streams, is a potentially attractive solution. In this study, a multi-criteria analysis was conducted to identify sites suitable for such alternative land uses. The cost-effectiveness of DIN reductions from these land use changes were calculated, accounting for reductions in annuity gross margins and land conversion cost. In certain locations (where conversion costs are low and DIN reductions are high) treatment wetlands and no-input cattle grazing offer cost-effective DIN reduction in the range of 20–26$/kg DIN. This compares favourably with existing agricultural extension-based approaches (c. $50/kg DIN reduction). Ecosystem service wetlands (i.e., wetland restoration for fish production) – again when appropriately situated – offer the prospect of even more cost-effective performance (11–14 $/kg DIN reduction). These results, in conjunction with best management practices, support the premise that alternative land uses are cost-effective options for improving water quality in certain areas of low-lying, low productivity sugarcane land. On-going investments by government in addition to private market funding mechanisms could be appropriate for supporting such land use transitions. These approaches need to be tested and refined via targeted pilot projects, as part of a whole-of-landscape approach to achieve broader reef water quality targets.

1. Introduction

Over a quarter of the world's reefs are under threat from pollutants such as nutrients, sediments and pesticides from catchment run-off associated with agricultural, urban and industrial land uses (Burke et al., 2011). These impacts are compounded by vegetation clearing and wetland reclamation connected to catchment development, reducing the processing and buffering capacity of natural ecosystems (Flanagan and Richardson, 2010; Verhoeven et al., 2006). Each year over 185 million tonnes of fertiliser are applied worldwide to agricultural crops (FAO, 2019) and some of this is lost as run-off leading to eutrophication in waterways and coasts (Boesch and Brinsfield, 2000). The extent of agriculture and demand for fertiliser (FAO, 2019) is projected to increase to support the food requirements of an increasing global population (FAO, 2004). To balance food and fibre demands with adverse agricultural pollutant impacts, countries around the world have imple-

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https://doi.org/10.1016/j.marpolbul.2021.112373
Received 7 November 2020; Received in revised form 1 April 2021; Accepted 7 April 2021
0025-326/© 2021

mented catchment management programs to reduce pollutants entering aquatic ecosystems (Kroon et al., 2012). These have included the regulation of nitrogen fertiliser use on crops (Kronvang et al., 2008), soil and water conservation (Chu et al., 2009), reduced stocking density (Kronvang et al., 2008), on-farm treatment systems (i.e., treatment wetlands and bioreactors) (Christianson et al., 2013) and conversion of high pollutant risk agricultural land to alternative production systems or natural ecosystems (Frisvold, 2004; Tian et al., 2010). Such responses are necessary under the United Nations declaration of ecosystem restoration in order to halt and reverse the decline of the world's ecosystems (Waltham et al., 2020).

The World Heritage-listed Great Barrier Reef in northern Australia faces the same challenges, with increased nutrient, sediment and pesticide loads from broad-scale land uses threatening ecosystem health and resilience (Brodie and Pearson, 2016; Brodie and Waterhouse, 2012; Waltham et al., 2019; Waterhouse et al., 2012). This iconic natural asset holds significant cultural and spiritual values, is a biodiversity hotspot, and has an economic, social and icon asset value of $56 billion (Deloitte Access Economics, 2017). It contributes $6.4 billion per year to the Australian economy and supports an estimated 64,000 jobs through tourism, recreation, fisheries and scientific activities (Deloitte Access Economics, 2017). Declining reef health will potentially have a significant impact on the environmental value and economic return generated by the GBR as an asset. Stoeckl et al. (2014) conclude that changes in the environment of the GBR would have a major impact on national and regional economies, through impacts to the tourism industry. Addressing key threats to the health and resilience of the GBR, including land-derived pollutants, is therefore a priority to maintain the economic, environmental and social values of the unique GBR ecosystem (Adame et al., 2019b; DEHP, 2016; Waltham et al., 2019).

Dissolved inorganic nitrogen (DIN), fine sediments and pesticides are the main pollutants of concern entering the GBR and agricultural land uses are the dominant contributor (Brodie and Waterhouse, 2012; Brodie et al., 2016; Waterhouse et al., 2016; Waterhouse et al., 2012). Fertilised crops, predominantly sugarcane, cover extensive areas of the coastal floodplains along the GBR coast and there is a strong relationship between the area of nitrogen-fertilised agricultural land and average DIN concentration in high-flow events (Brodie and Pearson, 2016; Brodie and Waterhouse, 2012). Therefore, fertiliser residues from agricultural land are considered a key contributor to DIN entering the GBR lagoon (Bartley et al., 2017; Terrain NRM, 2015) and sugarcane contributes up to 80% of the total DIN load (from anthropogenic sources) in some catchments (Hateley et al., 2014). DIN is a major threat to coral reef ecosystems in the GBR as high nutrient concentrations have been linked to outbreaks of corallivorous Crown of Thorns starfish (COTS). There is evidence to suggest that nitrogen runoff reaching the COTS 'initiation zone' (approximately between Lizard Island and Cairns) contributes to outbreaks, as higher nutrient levels sustain higher levels of phytoplankton and increase survival rates of the COTS larvae (Fabricius et al., 2010). It is estimated that the rivers in the Wet Tropics region contribute around 80% of the DIN load to the COTS initiation zone (Terrain.NRM 2015).

The Australian and Queensland Government's Reef 2050 Long-Term Sustainability Plan (2050 Plan) and associated Reef 2050 Water Quality Improvement Plan (Reef WQIP) 2017–2022 (State of Queensland, 2018) set out ambitious targets and actions required to reduce threats to the health and resilience of the GBR, with a major focus on reducing pollutants from broad-scale land use, notably agriculture (State of Queensland, 2018). A range of voluntary and regulatory policy instruments have been implemented by Government, agricultural industry bodies and natural resource management organisations to promote adoption of best management practices on farms in the GBR catchment. These have led to a reduction in DIN loads entering the GBR (Eberhard et al., 2017), however, the load reductions achieved fall well short of the DIN reduction targets set out in the Reef WQIP (Great Barrier Reef Marine Park Authority, 2019). The Scientific Consensus Statement 2017 (Waterhouse et al., 2017) recommends that different management options, including wetland and floodplain restoration, engineered treatment systems and changes in land use to less intensive options require investigation, to complement on-farm practice change.

There is a need to identify options to reduce DIN loads in the GBR that are cost-effective and also provide longer-term viability for landholders. This paper explores such options for land use transition to reduce DIN losses, targeting low-lying, high DIN risk sugarcane areas in the priority Wet Tropics catchments. It identifies potentially suitable land uses that pose a lower DIN risk while also potentially generating on-going income for landholders. This is followed by a cost-effectiveness analysis of the identified land use transition options for reducing DIN loads to the GBR.

2. Management of water quality in the wet tropics region

2.1. Study area

The Wet Tropics region is located in the northern region of the GBR catchment (Fig. 1) and has been highly modified since European settlement, with extensive vegetation clearing and hydrological modifications to facilitate agricultural production. Intensive agriculture dominates the coastal floodplain and grazing is the major land use in the drier, western parts of the region (Bainbridge et al., 2009; Canning and Waltham, 2021). Sugarcane production is the dominant intensive agricultural land use on the coastal floodplain and accounts for approximately 80% of the anthropogenic loads of DIN, and over 95% of the photosystem II (PSII) herbicide load to the GBR lagoon in the Wet Tropics region (Terrain NRM, 2015).

The Wet Tropics Water Quality Improvement Plan (WTWQIP) identifies that adoption of current best management practices in sugarcane will only go part of the way towards meeting the DIN water quality targets - it predicts 19% DIN load reduction for full adoption of B-class management practices (best practice) and 30% DIN load reduction, for full adoption of A-class practices (innovative/aspirational practices) (Hateley et al., 2014). It further explains that restoration of ecological functions in coastal ecosystems and floodplains is needed alongside improved agricultural management practice, although the water quality benefits of such actions are yet to be quantified. Similarly, the Alluvium (2016) report that estimated the costs of meeting the Reef WQIP water quality targets found that the target for DIN reduction in the Wet Tropics region could not be met under the then-current policy landscape and concluded alternative policy responses would be needed. The WTWQIP did not assess alternative policy options such as land use change or land buy-back options in its cost-effectiveness analysis. This was because insufficient information was available to make appropriate cost estimates (Alluvium, 2016), and what was perceived to be the high socio-political risk associated with promoting conversion of the remaining sugarcane land under D-class management (i.e., poor management of cane farming with high risk of pollutant loss) to alternative land uses, including biodiversity conservation or grazing with no fertiliser application. The modelling showed minimal DIN reduction from this land use change, due to the small area of D-class management remaining in the Wet Tropics region (Alluvium, 2016). This suggests that limiting land use change solely to land under D-class management severely constrained the opportunity for this policy option to substantially reduce DIN loads.

2.2. Transitioning low lying, high DIN risk, sugarcane land

Transitioning high DIN risk sugarcane land to an alternative land use that has a lower nitrogen input requirement or increased ability to process excess nitrogen is outlined as an alternative policy response in the Reef 2050 Plan targets (State of Queensland, 2018). There are several potential reasons why land use conversion may be an economically

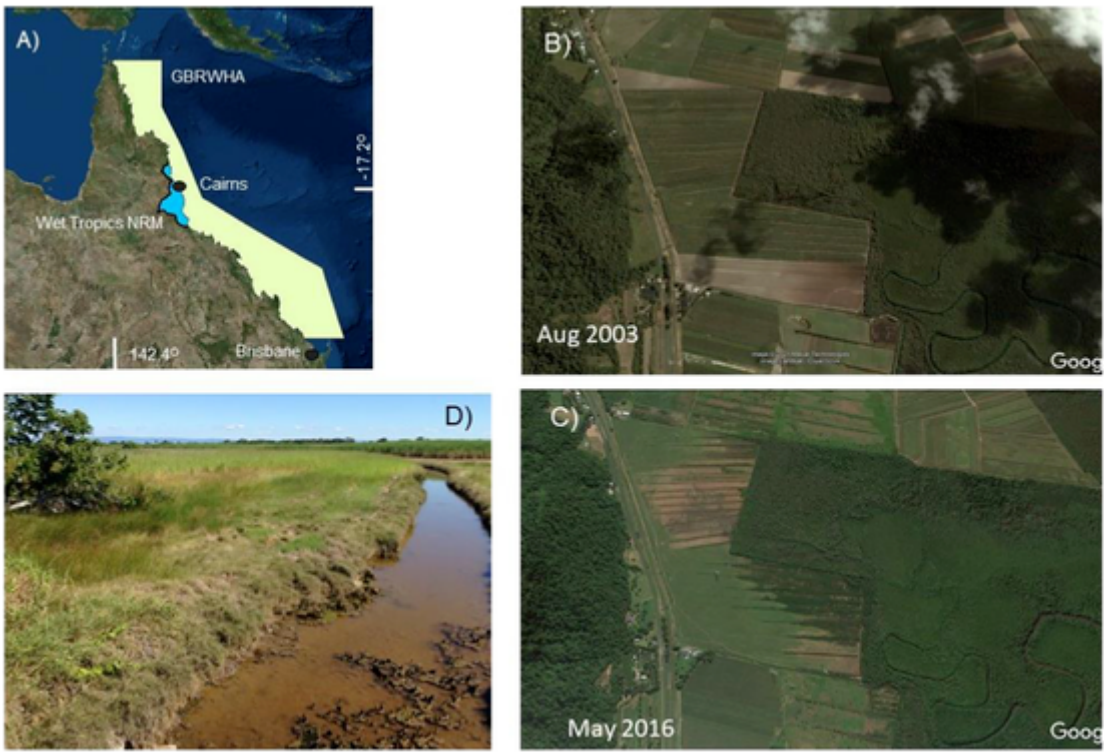


Fig. 1. A) Wet Tropics region and the Great Barrier Reef World Heritage Area; B and C) low-lying cane property illustrating encroaching mangrove and melaleuca forest over time; and D) example of a sugar cane drain used to remove wet season rainfall away from cane paddocks, where saltmarsh and mangroves are slowly returning.

and environmentally rational response, particularly in low-lying, high DIN risk, sugarcane production areas (Wallace et al., 2020). These areas are noted (but not in every case) for being unreliable from a production perspective and provide consistently difficult drainage and management challenges for landholders. Low-lying sugarcane land generally has lower productivity yield and requires greater inputs of fertiliser, and pesticide, to meet desired yield targets (Roebeling et al., 2007). It is also difficult to establish and harvest crops in these areas, high rainfall and inundation could potentially result in total crop loss, and they are likely to be significant sources of high nitrogen loss (Roebeling et al., 2007).

2.3. Land use change initiatives to mitigate agricultural pollutants

There are direct benefits to water quality from taking cropping land out of production and ceasing fertiliser and pesticide applications. Constructed or restored wetlands have also been shown to reduce agri-pollutant loads (Flanagan and Richardson, 2010; Griffiths and Mitsch, 2020; Mitsch et al., 2001; Verhoeven et al., 2006), however, monitoring to date has not identified pollutant reduction in constructed or restored wetlands in the Wet Tropics and Burdekin regions (Adame et al., 2019a).

2.4. Frameworks for prioritising water quality and ecosystem health initiatives

Spatial prioritisation is widely used to inform investment in water quality initiatives and is recognised as a way to accelerate progress towards reef water quality targets, e.g., Water Quality Improvement Plans, Water Quality Relative Risk Assessment. Spatial assessment and prioritisation frameworks can be used to assess multiple ecosystem services and the trade-offs between services, to support decision-making. When ecosystem services are considered, retention or restoration of natural ecosystems are often shown to be economically viable from a societal perspective due to the range of services and benefits achievable, i.e., carbon sequestration, amenity and recreational value, and fish production (Waltham et al., 2020). Ecosystem service valuation can help support a holistic approach to coastal management, although it needs to be considered site-by-site, and within the appropriate social context. Assessing multiple ecosystem services enhances the capacity to achieve co-benefits, and there are a range of models and frameworks available (e.g., InVEST, https://naturalcapitalproject.stanford.edu/software/invest).

The study aims to identify alternative land use options for low-lying, high DIN risk, low productivity/profitability, sugarcane land that would achieve the goal of reducing DIN loads to the GBR lagoon and also benefit farmers economically. Land use options were identified via a process of discussions with stakeholders and a review of available literature. Under advice provided by stakeholders and experts an initial set of alternative land uses were identified: aquaculture, alternative crops (e.g., fruit trees, rice, and hemp), forestry (e.g., pine and mangroves for harvesting), wetlands for fish production or water treatment, and also leaving land to regenerate under its own processes.

A desktop review of land use options was undertaken to identify climatic, market feasibility and nitrogen application requirements. Nitrogen application requirement is treated as a proxy for DIN loss risk. An important prerequisite was that nitrogen requirements had to be lower than the generic recommendations for sugarcane in the Wet Tropics as outlined by (Sugar Research Australia, 2013): 120-150 kg N/ha for fallow plant; 160-200 kg N/ha for replant and ratoons. Those land use options requiring less than 150 kg N/ha, tolerant of a humid coastal wet tropical climate, and potentially feasible from a market perspective, were shortlisted for consideration in the Multi-Criteria Analysis (MCA), and subsequent cost-effectiveness analysis as prospective land use options for low-lying sugarcane land transition in the Wet Tropics (Table 1). Note – not all low-lying land that were mapped here are marginal cane, each mapped site requires further on-ground inspection and consideration of constraints (e.g. soils, hydrology).

Table 1
Ranking of data sources in the MCA for either restore wetland ecosystem services or treatment system.

Criteria	Ranking for restoring ecosystem services	Ranking for treatment systems	Rationale for ranking and assumptions
DIN generation in the sub-catchment	Relatively low DIN (<13 kg/ha/yr) is ranked highest, moderate DIN (>13 - <21 kg/ha/yr) is ranked lower.	High DIN (> 21 kg/ha/yr) is ranked highest, moderate DIN (>13 - <21 kg/ha/yr) is ranked lower.	In sub-catchments with high DIN generation, treatment systems are preferable to help capture and treat pollutants. Restoring ecosystem services should be limited to sub-catchments with low to moderate DIN to minimise eutrophication of restored ecosystems.
Proximity to wetlands, including those of high ecological significance (HES)	Adjacent to HES estuarine or freshwater wetland is ranked higher than non-HES wetlands.	Adjacent to HES wetland has the highest ranking. Within 50 m of estuarine or freshwater wetland has a lower ranking.	To protect and enhance the values of HES wetlands, restoring ecosystem services and treatment systems adjacent to HES wetlands are ranked higher than non-HES wetlands. Estuarine wetlands are ranked higher than freshwater due to greater potential connectivity to the GBR. The model prioritises treatment systems located within 50 m of a wetland, so that runoff is treated just prior to entering a wetland.
Proximity to protected areas, for management purposes	Adjacent to declared Fish Habitat Area is ranked highest. Adjacent to natural land or protected estates (e.g. national park) is ranked lower.	Not applicable	Land adjacent to a declared Fish Habitat Area or protected area is a priority for the restoring ecosystem services option, because the land could potentially be added onto the existing management unit. For example, if land was restored to estuarine wetland it could be included in an adjoining declared Fish Habitat Area. This is not applicable to treatment systems, as the intent and management of a treatment system is generally not commensurate with protected estates.
Connectivity to GBR	Very frequently and frequently connected	Very frequently and frequently connected	To improve GBR water quality and ecosystem function, the land should be regularly connected to the GBR.
Historic wetland (i.e., pre-clear)	Mapped as a wetland pre-clearing	Not applicable	Areas that were originally wetlands prior to catchment development are likely to be more suitable for restoring wetland ecosystem services.

Following a review of available data and discussions with State Government experts, a refined list of land use options was established, comprising:

1) Grazing (grass-fed beef fattening – with and without fertiliser applied to pastures); and
2) Wetlands:
   a. Restored natural wetlands to provide multiple ecosystem services and additional, extended, habitat for wildlife (hereafter termed restored ecosystem service wetlands); and
   b. Treatment systems to provide nitrogen runoff treatment before discharge to receiving waters (hereafter referred to as treatment systems).

Data were sourced primarily from State and Federal Government, industry and the local natural resource management group. Only comprehensive data layers for the entire Wet Topics region were used; for example, layers (such as acid sulfate soils) that were not consistently mapped for the study region were excluded because of confounding problems in applying decision rules consistently and fairly. Instead appropriate surrogate data were investigated, for example, in the case of acid sulfate soils, land zone mapping (land zone 1 = tidal flats and beaches; 2 = coastal sand dunes and beach ridges; and 3 = river and creek landforms) proved suitable, as these land zones are likely to have greater connectivity to watercourses and the GBR.

2.5. Multi-criteria analysis

The MCA was completed using GIS processes, where mapping and spatial analysis routines are used to assess specific land for alternate and/or suitable uses that differ from its current use. MCA is a decision-making tool to solve problems by applying large amounts of complex data in a structured way. Combined with GIS, MCA synthesises spatial and non-spatial as well as qualitative and quantitative data, representing criteria known to affect land suitability, to produce a suitability map of potential alternative land uses for the specific area concerned. It allows assessment of multiple, often conflicting, criteria and is suited to informing complex natural resource management decisions because it incorporates ecological, biophysical and social components, and enables stakeholders' judgement to be introduced into the decision system.

The advantage of using GIS-based MCA is that it offers the possibility to aggregate data from various disciplines (e.g., biological, environmental, social, economic factors) obtained from expert knowledge (Chen et al., 2010; Koschke et al., 2012; Malczewski, 2004). This lets managers identify alternative and/or preferred options for land parcels according to diverse suitability criteria. Land use suitability analysis is frequently used in ecological economics due to its powerful and flexible capability for assessing land suitability (McDowell et al., 2018); the technique has been applied in a number of mapping applications such as identifying soil suitability for cotton crops, land suitability for prawn farming, and conservation effects on urban land use planning.

2.6. MCA process to assess suitability and identify locations for land use change

As determined through stakeholder engagement and the review of available data and literature, the GIS-based MCA consisted of the following steps: 1) defining sugarcane areas at risk of DIN loss, that were frequently connected to primary rivers; 2) determining a ranking system of suitability following four different land use scenarios using spatial analysis in ArcGIS Desktop 10.3.1 (ESRI), i.e., rank candidate land parcels to produce land suitability maps and provide summary statistics for each land use scenario; and 3) validating/reviewing the land suitability model using stakeholder and expert knowledge.

The first step of the MCA defined those sugarcane lands that are deemed low-lying and at risk of generating and delivering DIN to the GBR. This dataset is labelled DINrisk. The DINrisk dataset was generated by first defining the low-lying sugarcane areas within the Wet Tropics region and then adding information on connectivity to the coast, DIN generation rate and property boundaries.

Explicit sugarcane production data were not readily available and therefore low-lying, flood prone sugarcane areas were used as a surrogate for generally low productivity sugarcane land (notwithstanding that in some low-lying flood-prone areas, sugarcane yield may be higher than that of neighbouring higher elevation land in dry years). Here the low-lying sugarcane areas within the Wet Tropics region were defined as areas that are (Fig. 2):

1. less than 50 m above sea-level (to exclude sugarcane in elevated areas, such as the Atherton Tablelands and elevated parts of the Johnstone River catchment), but also constrain the study area comprising of Land Zone 1, 2, and 3;
2. subject to flooding, or within coastal or alluvial land zones, and
3. mapped as sugarcane land use in the Queensland Government (2009) spatial data (https://www.qld.gov.au/environment/land/management/mapping/statewide-monitoring/qlump) Note that the 2015 land use in Queensland Government was not available at the time of this research project, and could change slightly the results.

The low-lying sugarcane land parcels identified via this process were then ranked according to their risk of DIN generation and delivery to the GBR. Land parcels at risk of DIN loss to the GBR (DINrisk) are those that are frequently connected to the GBR, as identified via the GBRMPA blue maps (https://wetlandinfo.des.qld.gov.au/wetlands/facts-maps/wetland-background/). Estimates of DIN generation rate (in kg/ha) using Source Catchment model outputs and sugarcane property boundaries were added to the DINrisk dataset.

2.7. Land use transition scenarios

Two land use scenarios (three options in total) were considered in the MCA based on the land use options identified in the review. Specific factors and parcel ranking systems for each land use scenario are described below. Rankings were systematically scored on a continuous scale from the least (i.e., 0 or 1) to the most (i.e., 3 to 6) suitable land use options. Scores were assigned in rank order according to the characteristic factors involved in the parcels' evaluation for each land use scenario without repetition. To verify generated scores, group discussion with stakeholders and experts was carried out to achieve a final score consensus.

2.7.1. Scenario 1: Grazing

The land parcels in the DINrisk dataset were ranked according to their suitability for grazing as defined by expert opinion from Queensland Government, as follows:

- Land within 10 m of a stream or river was considered to be unsuitable for grazing, in order to provide space for a riparian buffer for control of erosion and sediment loss (based on recommended minimum buffer width for bank stabilization; Wegscheidl and Layden, 2011);
- Medium suitability was assigned to land at least 20 ha in size (normal area for establishing beef grazing systems pers. comm. Queensland Government) but not adjacent to current grazing land; and
- Land adjacent to existing grazing land is deemed highly suitable because new grazing land could potentially be added to an existing grazing enterprise, without the need to construct new cattle loading and trucking facilities. It is assumed that existing grazing properties are viable and that adding new grazing land would be possible. In addition, the assumption here is that converting to grazing land not only results in DIN reduction, but also would not contribute to increased risk from sediment erosion losses to receiving waterways.

2.7.2. Scenario 2: Wetlands (ecosystem services or treatment system)

This scenario investigates two wetland use options: restoring wetland ecosystem services or installation of a treatment system to capture and treat runoff. These two options utilise similar datasets, and a suitability ranking system was used to determine the most suitable land parcels for each option.

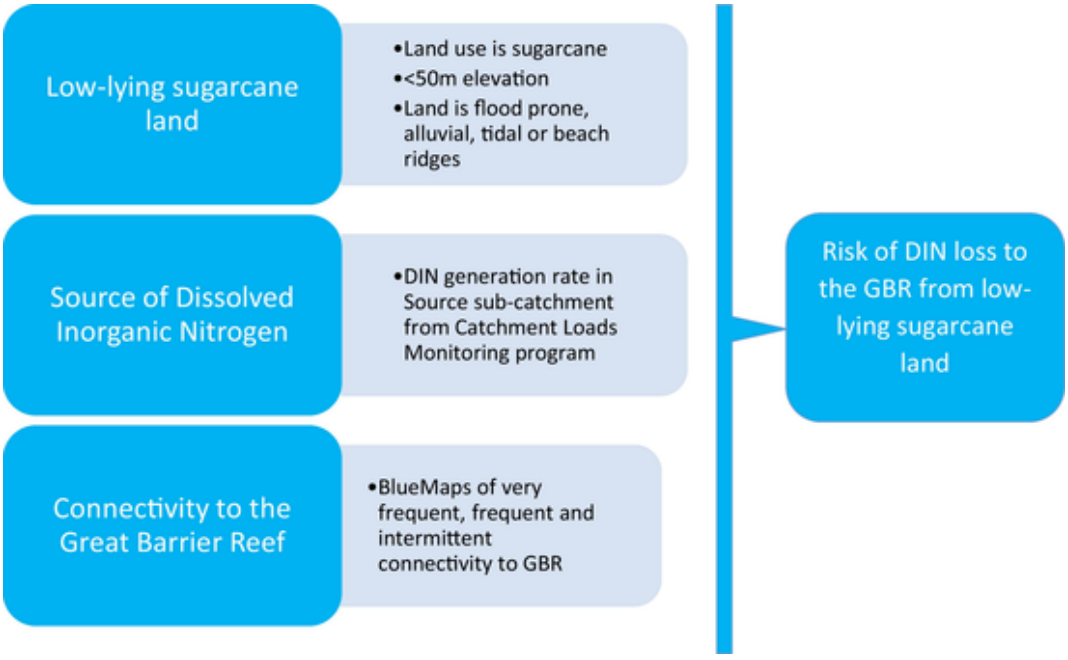


Fig. 2. Spatial data and their application in the framework to define low-lying land with risk of high DIN loss to the GBR.

This system ranked the values or features in each criterion as most suitable, moderately suitable, low suitability or unsuitable (Table 1) and the combination of values or features determined the suitability of a land parcel for either restoring wetland ecosystem services or a treatment system.

A) *Restore wetland ecosystem services*

Land suitable for restoring wetland ecosystem services was defined as:

1. Adjacent to a freshwater or estuarine wetland, with a higher ranking given to those of high ecological significance or land within a 'historic wetland' (i.e., land mapped as being a wetland prior to catchment development/clearing);
2. Adjacent to a declared Fish Habitat Area (FHA) or natural land and protected estates;
3. Frequently or very frequently connected to the GBR; and
4. Having a low to medium DIN generation rate (Fig. 3).

To determine parcels of land that met these criteria, land within the DINrisk dataset was ranked according to this wetland ecosystem restoration definition.

B) *Wetland treatment system*

Land most suitable for a treatment system was defined as:

1. Within 50 m of an existing wetland (i.e., freshwater or estuarine) consistent with buffer zones in the Wetland Management Handbook, with a higher ranking for land adjacent to wetlands of high ecological significance;
2. Frequent or very frequent connectivity to the GBR; and
3. Had a medium to high DIN generation rate prior to conversion (Fig. 4).

2.8. Limitations in the framework data analysis

All spatial data are subject to temporal and spatial uncertainty, which needs to be considered when interpreting results. Data availability and extent were major limiting factors in running the land suitabil-

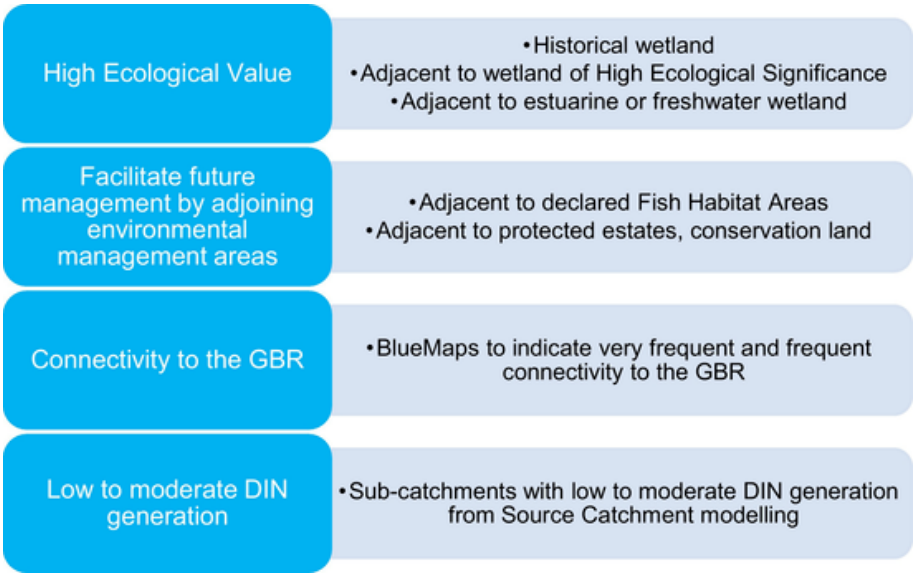


Fig. 3. Spatial data and application in the framework to define land most suitable for restoring wetland ecosystem services.

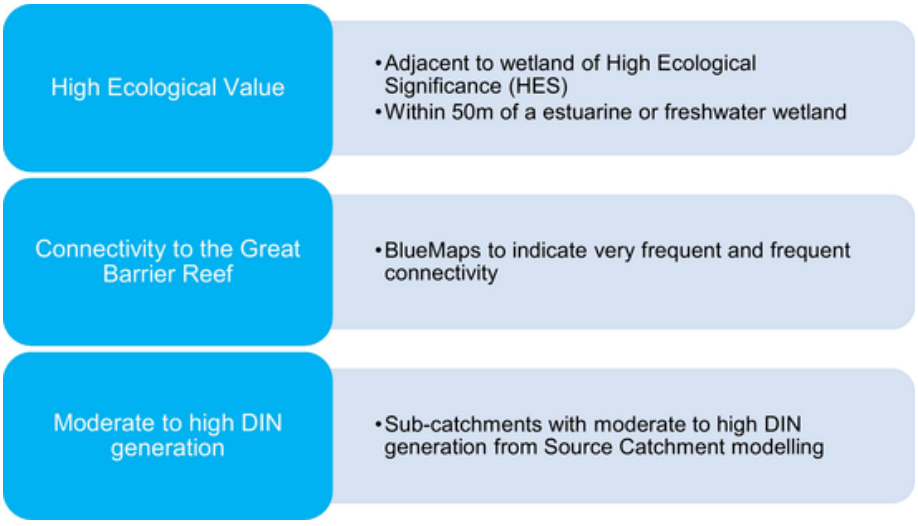


Fig. 4. Spatial data and application in the framework to define areas most suitable for treatment systems.

ity model. The data used were the most up-to-date data available with temporal ranges from 2009 to 2016. LiDAR has the greatest temporal range (2009 to 2015), however, is only used to create the 50 m contour line, having little impact on the resultant rankings. The river dataset (Ordered Drainage) is from 2007, however, the potential for the rivers to have changed significantly is minimal in our view, and not anticipated to have affected the outcome of the analysis (this assumption might not be supported in all rivers in the study area, and would require local data to confirm). Data extent is an important factor in obtaining a comprehensive land suitability model. Data needs to cover the entire area of interest otherwise there is a risk of excluding or including areas incorrectly (for example mapping of acid sulfate soils is limited in the region). Furthermore, due to inconsistencies in data extent and boundaries, a 1 m buffer was used when ranking parcels' suitability in relation to their adjacency/proximity to specific criteria. Finally, some parcels overlapped multiple sub-catchment boundaries, in these cases the parcel was intersected and the portions contained within each sub-catchment were treated as separate entities.

3. Economic analysis for land use transition options

3.1. Economic approach

The cost-effectiveness of each low-DIN-loss alternative land use option (grass-fed beef fattening with and without fertiliser input; restored wetland for supply of ecosystem services; wetland treatment systems) was calculated to compare the cost: outcome performance of the different options from the perspective of private landholders. This cost-effectiveness calculation requires estimation of: i) gross margins (GM) in present value terms from the new land use and from the sugarcane production that it would replace; ii) costs incurred in land conversion and establishing the new land use; and iii) DIN emissions under existing sugarcane production and the new land use. GM was calculated as the difference between production revenues and operating costs, assuming the production process is established. The GM calculation does not include upfront capital costs, taxes, capital depreciation and maintenance costs of non-production system-specific fixed assets. For each land use evaluated, all anticipated future revenues (R_t) and costs (C_t) incurred in year t over a production cycle of duration T years were discounted at real discount rate r to derive a total present value gross margin (TPVGM):

$$TPVGM = \sum_{t=1}^{T} \frac{R_t}{(1+r)^t} - \sum_{t=1}^{T} \frac{C_t}{(1+r)^t} \quad (1)$$

Sugarcane production and alternative land uses have operating cycles which differ in length and follow different cost profiles. A typical sugarcane production cycle in the Wet Tropics is five years in duration, comprising a harvest 15 months after the cane is first planted, three subsequent harvests from ratoon cane at 13 months intervals, and a final 6-month fallow before new sugarcane is planted again. Cattle fattening with fertiliser input to support pasture growth may operate on a four-year cycle consisting of one year of heavier fertiliser application alternating with three years of lighter applications. Restored wetlands and wetland treatment systems have relatively long lifespans of 15 years or more (US Department of Agriculture, 2019), but very different cost outlays, with the latter incurring higher upfront costs.

To provide a consistent basis for comparing the economic performance of land use options with operating cycles of different durations, TPVGM for each land use was converted to an (equivalent) annuity gross margin (AGM):

$$\text{AGM} = \text{TPVGM}\left(\frac{r}{1-(1+r)^{-T}}\right) \quad (2)$$

Having calculated the AGM from existing sugarcane production and the AGM from the new land use, the *reductions* in AGM for each alternative land use (relative to the AGM from sugarcane) were calculated as follows:

$$\text{AGM Reduction} = AGM_{sugarcane} - AGM_{new\ land\ use} \quad (3)$$

From the perspective of a private owner of low-lying sugarcane land, the choice is between continuing with sugarcane production or converting to another land use. In the absence of market-based incentives to pay for delivery of ecosystem services, the AGM from sugarcane production is higher than the AGM of the new land uses considered here. By switching to a new land use, the sugarcane farmer incurs an opportunity cost annually in the form of reduced AGM, in addition to incurring upfront capital costs. Thus, the present value cost (*PVC*) of land use change from the perspective of the sugarcane producer was calculated as:

$$PVCost = \sum_{t=1}^{T_e} \frac{\text{AGM Reduction}}{(1+r)^t} + C_0 \quad (4)$$

where C_0 is the upfront capital cost of land transition and T_e is the time horizon over which the comparison is evaluated. The total present value cost (in absolute $) (Eq. (4)) was converted to an annualised equivalent present value cost (APVC) (in $/year) using the formula shown in Eq. (2). This enables the annualised cost to be used alongside the yearly (assumed constant) DIN reduction from land use change to calculate the cost-effectiveness of the different land use change options (Star et al., 2018)[2]:

$$Cost_effectiveness = \frac{APVC}{Annual\ Reduction\ in\ DIN\ loss} \quad (5)$$

Using an annual discount rate of 5% (after e.g. Roebeling et al., 2007), the cost-effectiveness of land conversion to grazing, restored wetlands and wetland treatment systems was calculated for an evaluation time horizon of 12 years. A production time horizon of 12 years is chosen because it sits between the 10-year timeframe that has frequently been used when assessing the economic performance of management practice changes in sugarcane production in catchments draining to the Great Barrier Reef (e.g. Van Grieken et al., 2010, 2014) and the 15-year lifespan prescribed for wetland creation, wetland enhancement and wetland restoration by the US Department of Agriculture's Natural Resources Conservation Service (USDA, 2019 www.nrcs.usda.gov).

3.2. Annuity gross margins and upfront capital cost of land transition

Estimates of AGM from sugarcane production on four soil classes (S1, S2, S3 and S4 (see Supplementary Material for descriptions of soil characteristics); https://www.qld.gov.au/environment/land/management/soil/soil-data/mapping) in the Tully-Murray catchment in the Wet Tropics are reported in Roebeling et al. (2007). Overlaying the soil map (Roebeling et al., 2007) with the GIS maps of the locations of low-lying, high DIN risk, sugarcane land for the study area indicated that the majority of the low-lying, high DIN risk sugarcane locations are on soil classes S2 and S4. Thus, plot-level estimates of AGMs[3] from (Roebeling et al., 2007) for sugarcane grown on soil class S2 ($1603/ha) and soil class S4 ($1109/ha), both with a fertiliser application rate

[2] Star et al. (2018) use a similar approach to evaluate the cost-effectiveness of interventions to reduce fine sediment, DIN and pesticide loads to the Reef.

[3] The AGMs for sugarcane are expressed in 2016 AUD using the 'all groups' consumer price index for Brisbane Australian Bureau of Statistics, 2016. Consumer Price Index, Australia, December 2016, Australian Government.

of 150kgN/ha (Sugar Research Australia, 2013), are used in this study as the basis for deriving subsequent estimates of the AGM *reductions* following land use change.

Restored wetland would entail annual maintenance cost once construction was complete. Following land use change to restored ecosystem services wetland, from a landholder's perspective, the flow of gross margins from land that previously produced sugarcane ceases and is replaced by wetland maintenance cost. There is no published information on the annual maintenance costs of natural coastal wetlands in GBR catchments, thus a conservative value of $750/ha/year was assumed in this study based on Greening Australia's daily hire rate of $360/person (Bartley et al., 2018) plus travel cost. The assumed value of $750/ha/year is slightly higher than the median annual costs of wetland rehabilitation ($407/ha year^{-1} which equates to around $664/ha/year in 2016[4]) obtained in questionnaire responses from 31 projects in Australia (Streever, 1997).

Upfront costs for wetland restoration can be highly variable depending on restoration strategies, spatial- and time-specific factors. Restoration strategies should be chosen to address the impacts on, or threats to, wetland health (Streever, 1997; Waltham et al., 2019) such that, when restored, wetlands could over time regain their ecological functions and integrity to supply highly-valued regulating, provisioning and cultural ecosystem services (Barbier et al., 2011; Costanza et al., 2014). Wetland restoration may require just a single technique (e.g., minor modification to existing drainage or earthworks to flood previously drained land) or a mixture of techniques and thus be more expensive (e.g., fencing, weed removal, earthworks and revegetation in combination) (Abbott et al., 2020; WetlandCare Australia, 2008). In a global study of the costs and benefits of ecosystem restoration, De Groot et al. (2013) found that the upfront capital cost of restoring inland wetlands (defined as freshwater floodplains, swamps, marshes and peatlands (De Groot et al., 2012) ranged between $7000/ha - $38,000/ha (2016 AUD$).[5] Coastal wetland restoration that requires significant earthworks and vegetation establishment can mirror the process of installing a constructed treatment wetland. The upfront cost of a constructed treatment wetland was estimated by Alluvium (2016) at $275,000/ha. In this study, an upfront conversion cost ranging from $7000/ha to $275,000/ha was used in the cost-effectiveness calculation for land use change to restored coastal wetland.

Information on the maintenance costs of wetland treatment systems in tropical settings is scarce. Taylor (2005) suggested that annual maintenance cost is approximately 2% of construction costs. An example provided in *The Wetland Economic Assessment Tool* (WetlandInfo, 2006) indicates a maintenance cost of around 4% of construction cost (annual operating expenses of $5636 for a 10 ha wetland with a construction cost of $127,059). This study used the average of these two estimates, assuming that annual maintenance cost was 3% of wetland treatment system construction cost.

Estimates of construction costs for treatment wetlands in Australia and New Zealand were collated from publicly available sources (Alluvium, 2016; Faulkner and Bichan, 2015; Taylor, 2005; WetlandInfo, 2014) and all reported cost figures were escalated to 2015/16 AUD.[6] The total construction cost of a treatment wetland installed on a banana farm in the Wet Tropics (WetlandInfo, 2014) and a dairy farm in New Zealand (Faulkner and Bichan, 2015) were reported to be $61,472/ha and $57,951/ha, respectively. However, estimates of installation cost reported by Alluvium (2016) and Taylor (2005) are much higher at $900,000/ha for a small wetland. Thus, construction costs for constructed treatment wetlands were assumed to range between $60,000/ha and $900,000/ha in the cost-effectiveness assessment in this study.

Estimates of AGM for grazing with and without fertiliser input were obtained following a detailed discussion with a Queensland Department of Agriculture and Fisheries (DAF) Senior Beef Extension Officer. Detailed descriptions of how AGMs for grazing were calculated are provided in the Supplementary Material. Annual revenues, operating costs and AGMs from grazing with and without fertiliser input for light feeder cattle on soil classes S2 and S4 (Roebeling et al., 2007) are summarised in Table 2. Major costs of transitioning from sugarcane production to beef production are watering infrastructure, fencing, land preparation, and pasture preparation and establishment. With advice from the DAF Senior Beef Extension Officer, total establishment cost for a 10-ha grazing operation was estimated to be $14,451. An additional cost of $15,000 for constructing a stockyard and loading facilities was added for land blocks which were not adjacent to existing grazing land.

3.3. DIN losses

Transitioning from sugarcane to alternative lower-DIN loss land uses reduces DIN loss. Roebeling et al. (2007) estimated the DIN losses from sugarcane production in the Tully on soil classes S2 and S4 at a nitrogen application rate of 150 kgN/ha to be 31 kgDIN/ha and 50 kgDIN/ha, respectively.

Wetland treatment systems, when installed in the right location, and appropriately designed to remove nutrients under ambient groundwater and surface water flows, are likely to contribute to DIN load reduction at the Reef (Department of Environment and Heritage Protection, 2016). The wetland treatment systems considered in this study are assumed to meet the minimum design specifications detailed in WetlandInfo (https://wetlandinfo.des.qld.gov.au/wetlands/) case studies (e.g., WetlandInfo (2014)). The range of DIN removal rates assumed for constructed treatment wetlands in this study is derived from results reported by Kadlec and Wallace (2008). Of the 72 free-water surface wetlands whose performance is reported by Kadlec and Wallace (2008), 60% delivered average DIN removal rates of greater than 370 kg DIN/ha/year, whilst 70% removed in excess of 220 kg DIN/ha/year. On this basis, the range of DIN removal rates assumed for constructed treatment wetland in this study is 200–400 kg DIN/ha/year.

Under appropriate conditions, natural floodplain wetlands can also denitrify nitrogen runoff from surrounding agricultural lands (Adame et al., 2019a; Kroon et al., 2016). The rate of denitrification is dependent on the input nutrient load, residence time, wetland configuration, vegetation type and chemical properties. Adopting a conservative approach, we used the lower end of the DIN removal rate range for wetland treatment systems quoted by the Queensland Department of Environment and Heritage Protection (200 kg DIN reduction/ha/year) as an upper bound for DIN removal rate by restored coastal wetlands (Department of Environment and Heritage Protection, 2016). This upper bound removal rate is well below the mean potential denitrification rate (438 kg DIN removed/ha/ year) reported by Adame et al. (2019a) in a study across the wet and dry seasons of five floodplain wetland forests dominated by *Melaleuca* spp. in the Wet Tropics. A nominal minimum annual DIN removal rate of 20 kg DIN/ha/year (10% of the maximum rate) is used as a lower bound for the denitrifying performance of restored ecosystem service wetlands.

[4] Escalated using the Consumer Price Index published by the Australian Bureau of Statistics.

[5] De Groot et al. reported costs in 2007 US $. These costs were converted to 2007 Australian $ using the US$: AUD$ exchange rate for June 2007 from the US Federal Reserve (https://fred.stlouisfed.org/series/DEXUSAL) and then escalated to 2015/16 Australian dollars using the Consumer Price Index published by the Australian Bureau of Statistics.

[6] Cost figures that are in Australian dollars were escalated from the original studies to 2015/16 dollars using the Consumer Price Index published by the Australian Bureau of Statistics. If the original reported figures are in New Zealand dollars, costs were escalated to 2015/16 using the New Zealand Reserve Bank Calculator and then converted to Australian dollars at 1 NZD = 0.96 AUD exchange rate.

Table 2
Annuity gross margins (AGM) in $/ha and DIN loss in kgDIN/ha/year from grazing with low fertiliser input (Low input grazing) and without fertiliser input ('No input' grazing) on soil classes S2 and S4 in the Wet Tropics.

Business scenario	Low input grazing		'No input' grazing	
	AGM ($/ha)	DIN loss (kg/ha year^{-1})	AGM ($/ha)	DIN loss (kg/ha year^{-1})
L2: light feeders on soil class S2	352	2.7	307	0
L4: light feeders on soil class S4	254	5	269	0

For beef grazing with fertiliser input, this study used the DIN loss estimates on soil classes S2 and S4 for light feeder cattle provided by Roebeling et al. (2007), as summarised in Table 2. Detailed descriptions of how DIN losses under grazing are calculated are provided in the Supplementary Material.

3.4. Cost-effectiveness results

Results of the economic analysis of land use transition options are presented as reductions in annuity gross margins (calculated via Eq. (3)), present value cost of conversion (Table 3) and cost-effectiveness of DIN reduction (Fig. 5), by soil types. All land use transition options in Table 3 deliver lower AGMs than sugarcane, producing positive results for reductions in AGM (see Supplementary Material for further derivation of reductions in AGMs). Low- and no-input grazing on soil class S4 produced the lowest reductions in AGMs of between $840–$860/ha/year and the lowest present value costs of conversion, between $10,400–$10,500/ha. This is because cattle grazing can still deliver positive AGMs of $254–$352/ha and $273–$311/ha for low-input and no-input grazing, respectively. Consequently, reductions in AGMs from sugarcane are offset somewhat by these positive AGMs from beef production.

Table 3
Reductions in annuity gross margins (AGMs) and present value costs of land conversion. Present value costs of conversion are calculated with a 5% discount rate over a 12-year timeframe for all land uses.

Land use conversion	Soil type	Reduction in AGM ($/ha/year)		Present value cost over 12-year timeframe ($/ha)	
		Min	Max	Min	Max
Ecosystem wetlands[a]	S2	–	2353	27,855	295,855
Ecosystem wetlands[a]	S4	–	1859	23,477	291,477
Treatment wetlands[b]	S2	3403	28,603	90,162	1,153,516
Treatment wetlands[b]	S4	2909	28,109	85,783	1,149,137
Low input grazing[c]	S2	–	1251	12,533	14,033
Low input grazing[c]	S4	–	855	9023	10,523
No input grazing[c]	S2	–	1296	12,932	14,432
No input grazing[c]	S4	–	840	8890	10,390

[a] The minimum and maximum present value costs for restored ecosystem wetlands reflect the range in the construction costs from $7000–$275,000/ha.

[b] The minimum and maximum present value costs for constructed treatment wetlands reflect the range in the construction costs from $60,000 - $900,000/ha. The maximum and minimum values for reduction in AGM are due to using a range of $1800–$27,000/ha/year for treatment wetland maintenance costs. The minimum and maximum present value costs for treatment wetlands reflect the ranges of maintenance cost as well as the construction costs.

[c] The minimum present value cost for grazing corresponds to land that is adjacent to existing grazing land, thus removing the requirement to construct a stock yard for livestock loading/unloading. The maximum present value cost for grazing refers to converted land that is not adjacent to existing grazing land, so additional stock yard construction costs are incurred.

Reductions in AGMs (Table 3) for transitioning to wetlands are higher than for grazing because the reduction comprises the sum of forgone AGM from sugarcane and wetland maintenance cost. In the absence of any market mechanism to generate revenue streams for supplying DIN reductions and other ecosystem services (e.g., carbon sequestration), the wetland options would continue to deliver negative AGMs. The Present Value (PV) costs of conversion (Table 3) include the impact of upfront costs of land conversion and reductions in AGMs (Eq. (4)). Maximum PV costs of land use conversion arise from applying the upper bound estimates of the upfront costs of conversion, whereas the minimum costs arises from the use of lower bound cost estimates in the PV cost calculations. The difference between the minimum and maximum PV costs of conversion is high for wetlands, particularly for treatment wetlands, because of the wide range of upfront construction costs reported in the literature. Conversion to cattle grazing incurs much lower upfront costs relative to wetlands.

As shown in Fig. 5, the most cost-effective DIN reductions are obtained when land use transition incurs minimum (PV) cost but delivers maximum DIN reductions in appropriate locations. Under the assumptions of this assessment, the most cost-effective treatment wetlands and grazing locations deliver DIN reductions in the range of $20–$26/kgDIN. Appropriately situated restored ecosystem wetlands offer better cost-effectiveness at $11–$14/kgDIN. For the grazing options, DIN loss reductions are given by the difference between DIN loss from sugarcane and DIN loss from grazing. For the wetland options, DIN loss reductions come from savings in DIN loss from sugarcane plus active DIN removal by the wetlands themselves. Higher assumed rates of DIN removal enable treatment wetlands to deliver relatively cost-effective DIN reduction, even though the lower bound of upfront costs assumed for treatment wetlands is much higher than that for ecosystem wetlands or grazing. For the cattle grazing options, DIN reductions can be obtained much more cost-effectively by converting sugarcane land on the poorer soil class S4 (Fig. 5).

3.5. Cost-effectiveness discussion

Reducing nitrogen loads to the Reef remains a priority, backed by government policy (Commonwealth of Australia, 2015), water quality improvement plans (State of Queensland, 2018) and catchment-specific load reduction targets (Brodie et al., 2016; State of Queensland, 2018). Recent changes to the regulations governing sugarcane farming in Reef catchments mandate that nitrogen application rates are closely matched to cane requirements, informed by regular soil testing and farm-level nitrogen management plans (State of Queensland, 2019a, 2019c). Compliance with the new regulations should reduce end of catchment nitrogen loads substantially (State of Queensland, 2019a, 2019b). Compliance will be achieved by implementing improvements in nitrogen management practice that were identified in a recent review as being amongst the most cost-effective approaches for nitrogen reduction (Farr et al., 2019). However, even when the 2019 Reef Regulations have been implemented in full, it is likely that further nitrogen reductions will still be required to achieve the catchment-specific load reduction targets for the Wet Tropics in the Water Quality Improvement Plan (Terrain NRM, 2015). Thus, there remains a need for additional cost-effective approaches for nitrogen load reduction in Wet Tropics catchments. The Farr et al. (2019) review reports the cost-effectiveness of extension-based approaches for nitrogen reduction to be around $50/kg DIN. In comparison, the cost-effectiveness estimated here for ecosystem

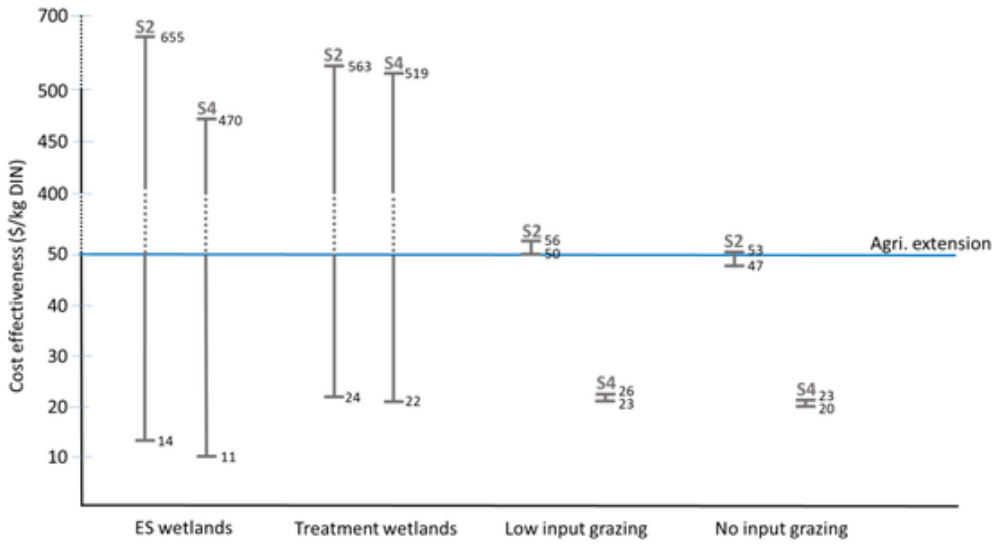


Fig. 5. The range of cost-effectiveness for DIN reductions achieved by the different land conversion options from sugarcane land on soil classes S2 and S4. Cost effectiveness is expressed as the annualised present value cost of land conversion per annual kg DIN reduction delivered (Eq. (5)). (ES wetlands denotes ecosystem service wetlands).

service wetlands, treatment wetlands and no-nitrogen input grazing – when implemented in the right circumstances – appears very attractive (Fig. 5). The wide range of cost-effectiveness estimated in this study for restoring ecosystem service wetlands and constructed treatment wetlands arises from the wide ranges assumed for both construction costs and denitrification rates.

No-input grazing on S4 soils also appears to offer cost-effective nitrogen reduction, compared with the cost-effectiveness of previous extension programs. In addition, the low upfront costs and familiarity with beef grazing as a land use option within the farming industry suggest that barriers to entry may be lower for this option than for wetlands. To avoid increasing pathogen loading in locations with sensitive receiving waters, it may be necessary to install vegetated buffer strips to reduce pathogen runoff (Atwill et al., 2002) as part of the overall land re-configuration.

To facilitate transition from sugarcane to ecosystem service wetlands, constructed treatment wetlands or no-input grazing, a funding stream will be required that (at minimum) is sufficient to cover the costs that landholders incur. Under the proposed Reef Credits initiative, verifiable reductions in nitrogen applications that persist over 10–25 year time frames can attract regular payments. A change to a land use that uses less fertiliser than cane thus presents an option for securing Reef Credit payments, i.e., method for accounting reduction in nutrient run-off through managed fertiliser application (Schultz and Sinclair, 2020). Reef Credit payments could thus be used to recover the modest upfront costs of conversion to no-input grazing and could help absorb the opportunity cost of transitioning to non-agricultural land uses. A Reef Credits methodology for wetland restoration is currently in the final stages of development, so this funding pathway may also become available for the wetland options. Other mechanisms, such as the Queensland Land Restoration Fund's 'carbon with co-benefits' scheme (State of Queensland, 2020), that recognise and reward the bundling benefits supplied by ecosystem service wetlands (e.g., carbon sequestration and storage, habitat enhancement for threatened flora and fauna, water quality improvements for the GBR, and employment opportunities for rural and First Nations communities) could provide stacked payments that are more likely than a single service-focused payment to cover wetland construction and maintenance costs (Canning et al. in review). For all the alternative land use options addressed here, careful consideration must be given to site selection to obtain the most cost-effective outcome, and thus achieve the highest possible improvement in reef water quality under available funding mechanisms.

4. Facilitating on-ground implementation – Next steps towards water quality improvement

4.1. Site-specific options analysis

The mapping and economic assessment identified potential areas of land suitable for transitioning to grazing, restored wetlands and treatment systems, and also indicated that these land use changes can be cost-effective for reducing DIN if the correct site is selected. A site-specific options analysis needs to be undertaken to identify site-specific social, economic and environmental implications of land use transition. The site-specific options analysis consists of two elements: 1) shortlisting sites; and 2) site assessment.

4.1.1. Shortlisting sites

The process for shortlisting sites will depend on the source of funding available to initiate the transition. In some instances, landholders may wish to convert to another land use that offers a similar or greater return or lifestyle choice based on economic information alone. In many cases, government or private investment, including via credit schemes, will be required to provide a financial incentive for land use change. Investors should compare sites, via a detailed site assessment process, to determine those that are most cost-effective for reducing DIN whilst also meeting other investor priorities.

4.1.2. Site assessment

There are existing decision support systems for assessing and comparing sites for management interventions in GBR catchments. These generally focus on one objective, such as water quality (e.g., Reef Trust tender process and Water Quality Grants assessment) or wetlands (e.g., wetland decision support system and Walking the Landscape approach; DEHP, 2016). No single existing Decision Support System (DSS) covers all potential investor priorities, which may include water quality improvement, wetland restoration and/or biodiversity improvement (Canning et al., in review).

Most DSSs provide a framework for assisting in the analysis of site-specific options and do not consider all possible factors and site nuances. For example, the social and economic consequences of land use change may not be adequately considered in existing DSSs, e.g., impacts on mill viability. Therefore, DSSs should only be used to *support* decision making, not to make the final decision. A recommended process for assessing sites, to account appropriately and comprehen-

sively for multiple social, economic and environmental factors, is suggested.

4.1.3. Process for assessing sites

The following steps should be taken to identify the most suitable site/s for land use transition and the most appropriate land uses for those site/s:

1. Identify the management objectives and investor priorities (i.e., water quality improvement, enhancing fish habitat, carbon sequestration);
2. Define the scope, including the geographic area of interest, funding available and area (i.e., hectares) of land seeking transition;
3. Collate published biophysical, social and economic information on the site/s and broader catchment;
4. Elicit local landholder and stakeholder knowledge of the site/s; and
5. Workshop with landholders, investors and stakeholders to discuss available information, suitability of alternative land uses and further implications of land use transition.

5. Conclusions

Here we used a multi-criteria analysis to identify sites suitable for land use transition that would deliver water quality improvements for the GBR lagoon. The cost-effectiveness of DIN reductions from these changes was estimated, accounting for reductions in annuity gross margins and present value land conversion cost. In certain locations (where conversion costs are low and DIN reductions are high) treatment wetlands and no-input cattle grazing appear to offer cost-effective DIN reductions in the range of 20–26$/kg DIN. This compares favourably with the cost-effectiveness reported for existing agricultural extension-based approaches (c. $50/kg DIN reduction). Ecosystem service wetlands (i.e., wetland restoration) – again when appropriately situated – offer the prospect of even more cost-effective performance (11–14 $/kg DIN reduction). These results support the hypothesis that alternative land uses are cost-effective options for improving water quality in certain areas of low-lying, low productivity sugarcane land that are at high risk of DIN loss. On-going investments by the government in addition to private market funding mechanisms on the horizon could be appropriate avenues for this. These approaches need to be tested and refined via targeted pilot projects to improve the knowledge base for the MCA and the economic analysis, further assisting managers to advance towards the reef water quality targets. With full adoption of agronomic best management practices unlikely to meet the ambitious water quality targets in the Wet Tropics region, land use transition should be considered as part of a suite of interventions to improve water quality to the GBR. Funding for these transition initiatives for farmers could be accessed via market mechanisms such water quality improvement, carbon abatement or biodiversity mechanisms that are on the horizon.

Uncited reference

Declaration of competing interest

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

Acknowledgements

This project was funded through the Australian Government's National Environment Science Program Tropical Water Quality Hub (Project 2.1.2). We thank D. M. Audas and P. Groves from Great Barrier Reef Marine Park Authority, M Ronan from the Queensland Wetlands Program from the Department of Environment and Science, Bernie English from the Queensland Department of Agriculture and Fisheries, and E. Nicholas from the Office of Great Barrier Reef in the Queensland State Government, staff from Terrain NRM, cane industry groups and the many local farmers consulted during this project.

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.marpolbul.2021.112373.

References

Abbott, B.N., Wallace, J., Nicholas, D.M., Karim, F., Waltham, N.J., 2020. Bund removal to re-establish tidal flow, remove aquatic weeds and restore coastal wetland services—North Queensland, Australia. PLoS One 15, e0217531. https://doi.org/10.1371/journal.pone.0217531.

Adame, M., Franklin, H., Waltham, N., Rodriguez, S., Kavehei, E., Turschwell, M., Balcombe, S., Kaniewska, P., Burford, M., Ronan, M., 2019a. Nitrogen removal by tropical floodplain wetlands through denitrification. Mar. Freshw. Res. 70, 1513–1521. https://doi.org/10.1071/MF18490.

Adame, M.F., Arthington, A.H., Waltham, N., Hasan, S., Selles, A., Ronan, M., 2019b. Managing threats and restoring wetlands within catchments of the Great Barrier Reef, Australia. Aqua. Conserva.: Mar. Freshwa. Ecosys. 29, 829–839. https://doi.org/10.1002/aqc.3096.

Alluvium, 2016. Costs of Achieving the Water Quality Targets for the Great Barrier Reef. Alluvium Consultants Australia, Department of Environment and Heritage, Brisbane, Queensland, Australia.

Atwill, E.R., Hou, L., Karle, B.M., Harter, T., Tate, K.W., Dahlgren, R.A., 2002. Transport of Cryptosporidium parvum oocysts through vegetated buffer strips and estimated filtration efficiency. Appl. Environ. Microbiol. 68, 5517–5527. https://doi.org/10.1128/AEM.68.11.5517-5527.2002.

Australian Bureau of Statistics, 2016. Consumer Price Index, Australia, December 2016. Australian, Government.

Bainbridge, Z.T., Brodie, J., Faithful, J., Sydes, D.A., E, L.S., 2009. Identifying the land-based sources of suspended sediments, nutrients and pesticides discharged to the Great Barrier Reef from the Tully–Murray Basin, Queensland, Australia. Mar. Freshw. Res. 60, 1081–1090. https://doi.org/10.1071/MF08333.

Barbier, E.B., Hacker, S.D., Kennedy, C., Koch, E.W., Stier, A.C., Silliman, B.R., 2011. The value of estuarine and coastal ecosystem services. Ecol. Monogr. 81, 169–193. https://doi.org/10.1890/10-1510.1.

Bartley, R., Waters, D., Turner, R., Kroon, F., Wilkinson, S., Garzon-Garcia, A., Kuhnert, P., Lewis, S., Smith, R., Bainbridge, Z., 2017. Scientific Consensus Statement 2017: A Synthesis of the Science of Land-based Water Quality Impacts on the Great Barrier Reef, Chapter 2: Sources of Sediment, Nutrients, Pesticides and Other Pollutants to the Great Barrier Reef. State of Queensland, Australia.

Bartley, R., Wilkinson, S.N., Henderson, A., Hawdon, A., 2018. Cost Effectiveness of Gully Remediation in the Burdekin Catchment: Preliminary Insights Based on Measured Data from NESP Monitoring Sites and Progress with New Lidar Methodologies, Report to the National Environmental Science Programme. Reef and Rainforest Research Centre Limited, Cairns, Australia.

Boesch, D., Brinsfield, R., 2000. Coastal eutrophication and agriculture: contributions and solutions, Biological Resource Management Connecting Science and Policy. Springer, pp. 93–115., Heidelberg, Berlin.

Brodie, J., Pearson, R.G., 2016. Ecosystem health of the Great Barrier Reef: Time for effective management action based on evidence. Estuar. Coast. Shelf Sci. 183, 438-451. doi.org/10.1016/j.ecss.2016.05.008.

Brodie, J., Waterhouse, J., 2012. A critical review of environmental management of the 'not so Great' Barrier Reef. Estuar. Coast. Shelf Sci. 104-105, 1-22. doi.org/10.1016/j.ecss.2012.03.012

Brodie, J.E., Lewis, S.E., Collier, C.J., Wooldridge, S., Bainbridge, Z.T., Waterhouse, J., Rasheed, M.A., Honchin, C., Holmes, G., Fabricius, K., 2016. Setting ecologically relevant targets for river pollutant loads to meet marine water quality requirements for the Great Barrier Reef, Australia: A preliminary methodology and analysis. Ocean Coast. Manag. 143, 136-147. doi.org/10.1016/j.ocecoaman.2016.09.028.

Burke, L., Reytar, K., Spalding, M., Perry, A., 2011. Reefs at Risk Revisited: Technical Notes on Modeling Threats to the World's Coral Reefs. World Resources Institute, USA, Washington, DC.

Canning, .D., A., Waltham, N.J., 2021. The influence of survey bias in assessing the ecological impacts of climate change and habitat loss on vertebrate assemblages across the Great Barrier Reef catchment wetlands. Ecol. Evol.. https://doi.org/10.1002/ece3.7412.

Chen, Y., Yu, J., Khan, S., 2010. Spatial sensitivity analysis of multi-criteria weights in GIS-based land suitability evaluation. Environ. Model. Softwa. 25, 1582-1591. doi.org/10.1016/j.envsoft.2010.06.001.

Christianson, L., Christianson, R., Helmers, M., Pederson, C., Bhandari, A., 2013. Modeling and calibration of drainage denitrification bioreactor design criteria. J. Irrig. Drain. Eng. 139, 699–709. https://doi.org/10.1061/(ASCE)IR.1943-4774.0000622.

Chu, Z., Zhai, S., Lu, X., Liu, J., Xu, J., Xu, K., 2009. A quantitative assessment of human impacts on decrease in sediment flux from major Chinese rivers

entering the western Pacific Ocean. Geophy. Res. Let. 36 (19). doi.org/10.1029/2009GL039513
Commonwealth of Australia, 2015. Reef Trust Partnerships for the Reef. Canberra, Commonwealth of Australia.
Costanza, R., De Groot, R., Sutton, P., Van der Ploeg, S., Anderson, S.J., Kubiszewski, I., Farber, S., Turner, R.K., 2014. Changes in the global value of ecosystem services. Global environ. Chan. 26, 152-158. doi.org/10.1016/j.gloenvcha.2014.04.002.
De Groot, R., Brander, L., Van Der Ploeg, S., Costanza, R., Bernard, F., Braat, L., Christie, M., Crossman, N., Ghermandi, A., Hein, L., 2012. Global estimates of the value of ecosystems and their services in monetary units. Ecosyst. Serv. 1, 50-61. doi.org/10.1016/j.ecoser.2012.07.005.
De Groot, R.S., Blignaut, J., Van Der Ploeg, S., Aronson, J., Elmqvist, T., Farley, J., 2013. Benefits of investing in ecosystem restoration. Conserva. Biol. 27, 1286-1293. doi.org/10.1111/cobi.12158.
DEHP, 2016. Wetlands in the Great Barrier Reef Catchment Management Strategy 2016–2021. Department of Environment Heritage and Protection. Brisbane, Australia.
Deloitte Access Economics, 2017. At What Price? The Economic, Social and Icon Value of the Great Barrier Reef. Deloitte Access Economics, Australia.
Department of Environment and Heritage Protection, 2016. Treatment Systems in Coastal Catchments Forum: Summary Report. Queensland Wetlands Program and Office of the Great Barrier Reef. Queensland Government, Australia.
Eberhard, R., Brodie, J., Waterhouse, J., 2017. Managing water quality for the great barrier reef, decision making in water resources policy and management. Elsevier 265–289.
Fabricius, K., Okaji, K., De'Ath, G., 2010. Three lines of evidence to link outbreaks of the crown-of-thorns seastar Acanthaster planci to the release of larval food limitation. Coral Reefs 29, 593–605. https://doi.org/10.1007/s00338-010-0628-z.
FAO, 2004. Fertiliser Requirements in 2015 and 2030 Revisited. Food and Agriculture Organization, Rome.
FAO, 2019. World Fertilizer Trends and Outlook to 2022. Food and Agricultural Organization, Rome.
Farr, M., Mccane, D., Star, M., Moravek, T., Poggio, M., 2019. Cost-effectiveness of Changing Land Management Practices in Sugarcane and Grazing to Obtain Water Quality Improvements in the Great Barrier Reef: Evaluation and Synthesises of Existing Knowledge. Department of Agriculture and Fisheries. Queensland Government, Australia.
Faulkner, J.P.J.S.T., Bichan, A., 2015. Benefits and costs of a constructed wetland on a Wairarapa dairy farm. Journal of New Zealand Grasslands, 173-176. doi.org/10.33584/jnzg.2015.77.474.
Flanagan, N.E., Richardson, C., 2010. A multi-scale approach to prioritise wetland restoration for watershed-level water quality improvement. Wetl. Ecol. Manag. 18, 695–706. https://doi.org/10.1007/s11273-010-9188-9.
Frisvold, G.B., 2004. How federal farm programs affect water use, quality, and allocation among sectors. Water res. Res. 40. doi.org/10.1029/2003WR002753.
Great Barrier Reef Marine Park Authority, 2019. Great Barrier Reef Outlook Report 2019. Great Barrier Reef Marine Park Authority. Townsville, Queensland Australia.
Griffiths, L.N., Mitsch, W.J., 2020. Nutrient retention via sedimentation in a created urban stormwater treatment wetland. Sci. Total Environ. p.138337 doi.org/10.1016/j.scitotenv.2020.138337.
Hateley, L., Ellis, R., Shaw, M., Waters, D., Carroll, C., 2014. Modelling Reductions of Pollutant Loads Due to Improved Management Practices in the Great Barrier Reef Catchments - Wet Tropics NRM Region, Technical Report 3. Cairns, Australia.
Kadlec, R.H., Wallace, S., 2008. Treatment Wetlands. CRC press, New York, USA.
Koschke, L., Fuerst, C., Frank, S., Makeschin, F., 2012. A multi-criteria approach for an integrated land-cover-based assessment of ecosystem services provision to support landscape planning. Ecolog. Indica. 21, 54-66. doi.org/10.1016/j.ecolind.2011.12.010.
Kronvang, B., Andersen, H.E., Børgesen, C., Dalgaard, T., Larsen, S.E., Bøgestrand, J., Blicher-Mathiasen, G., 2008. Effects of policy measures implemented in Denmark on nitrogen pollution of the aquatic environment. Environ. Sci. Pol. 11, 144-152. doi.org/10.1016/j.envsci.2007.10.007.
Kroon, F.J., Kuhnert, P.M., Henderson, B.L., Wilkinson, S.N., Kinsey-Henderson, A., Abbott, B., Brodie, J.E., Turner, R.D., 2012. River loads of suspended solids, nitrogen, phosphorus and herbicides delivered to the Great Barrier Reef lagoon. Mar. Poll. Bull. 65, 167-181. doi.org/10.1016/j.marpolbul.2011.10.018.
Kroon, F.J., Thorburn, P., Schaffelke, B., Whitten, S., 2016. Towards protecting the Great Barrier Reef from land-based pollution. Glob. Chan. Biol. 22, 1985-2002. doi.org/10.1111/gcb.13262.
Malczewski, J., 2004. GIS-based land-use suitability analysis: a critical overview. Progress in Planning 62, 3-65. doi.org/10.1111/gcb.13262.
McDowell, R.W., Snelder, T., Harris, S., Lilburne, L., Larned, S.T., Scarsbrook, M., Curtis, A., Holgate, B., Phillips, J., Taylor, K., 2018. The land use suitability concept: introduction and an application of the concept to inform sustainable productivity within environmental constraints. Ecol. Indicat. 91, 212-219. doi.org/10.1016/j.ecolind.2018.03.067.
Mitsch, W.J., Day, J.W., Gilliam, J.W., Groffman, P.M., Hey, D.L., Randall, G.W., Wang, N., 2001. Reducing Nitrogen Loading to the Gulf of Mexico from the Mississippi River Basin: Strategies to Counter a Persistent Ecological Problem. Bioscience 51, 373. doi.org/10.1641/0006-3568(2001)051[0373:RNLTTG]2.0.CO;2.
Roebeling, P., Webster, A., Biggs, J., Thorburn, P., 2007. Financial-Economic Analysis of Current Best Management Practices for Sugarcane, Horticulture, Grazing and Forestry Industries in the Tully-Murray Catchment. Report to the Marine and Tropical Sciences Research Facility. Reef and Rainforest Research Centre, Cairns, Australia.
Schultz, J., Sinclair, J., 2020. Method for Accounting Deduction in Nutrient Run-off Through Managed Fertiliser Application - Version 1.1. GreenCollar, Sydney, Australia, p. 24.
Star, M., Rolfe, J., McCosker, K., Smith, R., Ellis, R., Waters, D., Waterhouse, J., 2018. Targeting for pollutant reductions in the great barrier reef river catchments. Environ. Sci. Pol. 89, 365-377. doi.org/10.1016/j.envsci.2018.09.005.
State of Queensland, 2018. Reef 2050 Water Quality Improvement Plan. Queensland Government, Brisbane, Australia.
State of Queensland, 2019a. Farming in Reef Catchments: Farm Nitrogen and Phosphorus Budget Guide (Agricultural Environmentally Relevant Activity Standard for Sugarcane Cultivation). Office of the Great Barrier Reef. Environmental Policy and Programs, Department of Environment and Science, Queensland Government, Brisbane.
State of Queensland, 2019b. Great Barrier Reef River Basins End-of-Basin Load Water Quality Objectives. Environmental Policy and Programs Division. Department of Environment and Science, Queensland Government, Australia.
State of Queensland, 2019c. Prescribed Methodology for Sugarcane Cultivation, in: Office of the Great Barrier Reef. Environmental Policy and Programs, Department of Environment and Science, Queensland Government, Brisbane, Australia.
State of Queensland, 2020. Land Restoration Fund co-Benefits Standard. Queensland Government, Australia.
Stoeckl, N., Farr, M., Larson, S., Adams, V.M., Kubiszewski, I., Esparon, M., Costanza, R., 2014. A new approach to the problem of overlapping values: A case study in Australia′ s Great Barrier Reef. Ecosyst. Serv. 10, 61-78. doi.org/10.1016/j.ecoser.2014.09.005.
Streever, W., 1997. Trends in Australian wetland rehabilitation. Wetl. Ecol. Manag. 5, 5–18.
Sugar Research Australia, 2013. Best practice nutrient management - The six easy steps approach, in: Sugar Research Australia (Ed.), Information Sheet IS13016.
Taylor, A., 2005. Structural stormwater quality BMP cost/size relationship information from the literature. Cooperative research Centre for Catchment Hydrology, Melbourne, 53-64, Australia.
Terrain NRM, 2015. Wet Tropics Water Quality Improvement Plan 2015–2020. Terrain NRM, Department of Environment and Heritage, Queensland Government, Australia.
Tian, Y., Huang, Z., Xiao, W., 2010. Reductions in non-point source pollution through different management practices for an agricultural watershed in the Three Gorges Reservoir Area. Journal of Environmental Sciences 22, 184-191. doi.org/10.1016/S1001-0742(09)60091-7.
US Department of Agriculture, 2019. Natural Resource Conservation Service. (USA.).
Van Grieken, M.E., Webster, A.J., Poggio, M., Thorburn, P., Biggs, J., Stokes, C., McDonald, C., 2010. Implementation Costs of Agricultural Management Practices for Water Quality Improvement in the Great Barrier Reef Catchments. CSIRO, Water for a Healthy Country National Research Flagship.
Van Grieken, M., Poggio, M., Smith, M., Taylor, B., Thorburn, P., Biggs, J., Whitten, S., Faure, C., Boullier, A., 2014. Cost-effectiveness of management activities for water quality improvement in sugarcane farming. Report to the Reef Rescue Water Quality Research & Development Program. Reef and Rainforest Research Centre Limited, Cairns (85pp.). ISBN: 978-1-925088-30-4.
Verhoeven, J.T., Arheimer, B., Yin, C., Hefting, M.M., 2006. Regional and global concerns over wetlands and water quality. Trends Ecol. Evol. 21, 96-103. doi.org/10.1016/j.tree.2005.11.015.
Wallace, J., Adame, M.F., Waltham, N.J., 2020. A constructed wetland near Babinda, north Queensland: a case study of potential water quality benefits in an agricultural tropical catchment in: Report to the National Environmental Science Program. Reef and Rainforest Research Centre Limited, p. 76, Australia.
Waltham, N.J., Burrows, D., Wegscheidl, C., Buelow, C., Ronan, M., Connolly, N., Groves, P., Audas, D., Creighton, C., Sheaves, M., 2019. Lost floodplain wetland environments and efforts to restore connectivity, habitat and water quality settings on the great barrier reef. Front. Mar. Sci. 6, 71. doi.org/10.3389/fmars.2019.00071.
Waltham, N.J., Elliott, M., Lee, S.Y., Lovelock, C., Duarte, C.M., Buelow, C., Simenstad, C., Nagelkerken, I., Claassens, L., Wen, C.K., 2020. UN Decade on Ecosystem Restoration 2021–2030—what chance for success in restoring coastal ecosystems? Front. Mar. Sci. 7, 71. doi.org/10.3389/fmars.2020.00071.
Waterhouse, J., Brodie, J., Lewis, S., Mitchell, A., 2012. Quantifying the sources of pollutants in the Great Barrier Reef catchments and the relative risk to reef ecosystems. Mar. Poll. Bull. 65, 394-406. doi.org/10.1016/j.marpolbul.2011.09.031.
Waterhouse, J., Brodie, J., Lewis, S., Audas, D.-M., 2016. Land-sea connectivity, ecohydrology and holistic management of the Great Barrier Reef and its catchments: time for a change. Ecohydrol. & Hydrobiol. 16, 45-57. doi.org/10.1016/j.ecohyd.2015.08.005.
Waterhouse, J., Brodie, J., Tracey, D., Smith, R., VanderGragt, M., Collier, C., Petus, C., Baird, M., Kroon, F., Mann, R., 2017. Chapter 3: the risk from

anthropogenic pollutants to Great Barrier Reef coastal and marine ecosystems. In: Government, Q. (Ed.), 2017 Scientific Consensus Statement: Land Use Impacts on the Great Barrier Reef Water Quality and Ecosystem Condition. Brisbane, Australia.
Wegscheidl, C., Layden, I., 2011. Grazing for Healthy Coastal Wetlands: Guidelines for Managing Coastal Wetlands in Grazing Systems. The State of Queensland (Department of Employment, Economic Development and Innovication), Brisbane, Australia (118p).
WetlandCare Australia, 2008. Wetland Rehabilitation Guidelines for the Great Barrier Reef Catchment. (WetlandCare Australia).
WetlandInfo, 2006. Wetland Economics Assessment Tool: Assessment of Sub-Surface (Reed-bed) Treatment Wetland. Queensland Wetlands Program, Queensland Government, Brisbane, Australia.
WetlandInfo, 2014. Banana Farming for Healthier Wetlands. Queensland Wetlands Program, Queensland Government, Brisbane, Australia.