Synthesis of evidence of the effectiveness of wetlands in water quality improvement, costs and cost-drivers in the Great Barrier Reef catchment area

Jane Waterhouse[1], Bianca Molinari[1], Megan Star[2], Tony Weber[3], Nathan Waltham[4], Katie Motson[4]
[1]C$_2$O Consulting, [2]Star Economics, [3]Alluvium, [4]Centre for Tropical Water & Aquatic Ecosystem Research (TropWATER), James Cook University

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Glossary

Climate change	Climate change refers to long-term shifts in temperatures and weather patterns, mostly driven by human activities (i.e., burning fossil fuels like coal, oil and gas) since the 1800s. Climate change-related potential pressures in the context of this document include: increasing temperature, intensity and frequency of heatwaves, ocean acidification, altered extreme rainfall events (drought / floods), rising sea levels, and frequency and strength of tropical cyclones.
Construction (economics)	In the context of wetlands, construction/capital works in an area that was not a wetland in the recent past and that is isolated from existing wetlands (i.e., not directly adjacent).
Costs (economics)	Expenses incurred across the three phases for wetland restoration, rehabilitation and construction: 1) pre-construction, 2) construction, and 3) post-construction. The pre-construction phase includes conceptualisation, design, planning, landholder engagements, and approvals; the construction phase includes earthworks and planting; and the post-construction phase which captures costs associated with monitoring, maintenance and repair.
Cost-effectiveness (economics)	Cost-effectiveness studies involve the integration of environmental and economic results. Cost-effectiveness may be calculated as the present value of costs (private, public, program, maintenance) of a particular intervention divided by the per unit reduction in pollutant (e.g., cost per kg of DIN abated). Some studies report separately on economic and environmental results to give an indication of the cost effectiveness of each management practice change (e.g., if the study includes some changes that improve profit and others that decrease profit). The economic methodologies associated with cost-effectiveness studies can differ between studies.
Effectiveness / Efficacy / Efficiency	Mass removal (including processing and retention) of nutrients (nitrogen, phosphorus), fine sediments or pesticides from the water body per unit area. Efficiency is often expressed as percentage of the input load of nutrients, sediments or pesticides removed (including processing and retention). For natural/near-natural wetlands where it is difficult to identify inlet and outlet points, efficiency will also consider measurements of denitrification and other nitrogen processes such as anammox and dissimilatory nitrate reduction to ammonium.
Natural/near-natural wetlands	<u>For this review</u>, natural and near-natural wetlands refer to lacustrine, palustrine and riverine wetlands, excluding subtidal and subterranean wetlands and also excluding estuarine wetlands, coral reefs, seagrass meadows, oyster reefs and aquifers. **Natural wetlands** refer specifically to wetlands without any anthropogenic structural or hydrological change to the wetland, or within its catchment. **Near-natural wetlands** refer to wetlands without any anthropogenic structural change to the wetland, but with anthropogenic structural or hydrological change occurring within the broader catchment.
Nutrient species	TN = Total nitrogen; TDN = Total dissolved nitrogen; DIN = Dissolved inorganic nitrogen; NO_3^- = Nitrate; NH_4^+ = Ammonium; NO_x = Nitrogen oxides; TP = Total phosphorus; FRP = Filterable reactive phosphorus; PO_4 = Phosphate, TIP = Total inorganic phosphorus.
Pesticides	A pesticide is a substance or mixture of substances used to kill, repel, or control pests, such as insects, rodents, or plants. Pesticides include herbicides, insecticides, and fungicides.
Pollutant	Any contaminant above natural background levels which may or may not cause an adverse effect. In this review, pollutants include nutrients, sediments and pesticides.
Rehabilitation	Action or actions to repair, enhance and/or replace ecosystem processes and/or components, to improve intrinsic values and/or ecosystem services (DESI 2022b).
Restoration	Action, or actions to bring back a former, original, normal, or unimpaired condition (DESI 2022b).
Treatment (or constructed) wetlands and other treatment systems	Treatment (or constructed) wetlands, and related treatment systems for improving water quality, are engineered wetlands that are designed to intercept, slow down, and remove sediments, nutrients, and other pollutants (e.g., pesticides) from water. DESI (2022a) defines **treatment or constructed wetlands** as: "…engineered systems that replicate and enhance the physical, biological, and chemical treatment processes occurring in natural wetlands. They differ from restored or natural wetlands in that they are designed and managed primarily to improve water quality". Other names for treatment wetlands are constructed wetlands, surface flow wetlands, and free-water wetlands. **Treatment systems**

	include floating wetlands, vegetated drains, recycle pits, swales, buffer strips and sediment basins.
Urban / non-agricultural	In this review, urban and non-agricultural land uses are considered together and are defined as those activities which may occur at a high level of intensity, with mixed application of pervious and impervious land surfaces and the generation of both diffuse and point sources of nutrients and other contaminants.
Water quality	The physical, chemical, and biological characteristics of water and the measure of its condition relative to the requirements for one or more biotic species and/or to any human need or purpose.
Wetland systems	Wetlands systems are lacustrine, palustrine, riverine, marine, estuarine and subterranean wetland systems (des.qld.gov.au). See Table 3 for detailed definitions of these systems.

Executive Summary

Wetlands in the Great Barrier Reef (GBR) catchment area hold immense significance due to the intrinsic values and the range of ecosystem services they provide, from biophysical (e.g., nutrient cycling), biological (e.g., biodiversity) and environmental (e.g., flood control and foreshore stabilisation) to economic (e.g., tourism) and cultural (e.g., aesthetic). The quality of water in wetlands can be highly variable which is a function of hydrology including surface/groundwater interactions, timing and frequency of rainfall and other weather conditions, human use, runoff from adjacent land uses and biota (including weeds and pests). Depending on the land use immediately surrounding or nearby wetlands, and modifications to natural processes, wetland water quality conditions can change or be altered from natural cycling conditions to alternative modified states.

Since European settlement (~1850), land-use changes and extensive modifications to floodplains in the GBR catchment area have contributed to the degradation and loss of wetland habitats, along with the critical ecosystem services they provide. While 78 to 97% of pre-European wetlands remain, this proportion varies widely between Natural Resource Management regions and basins. These losses are primarily due to changed land management including land clearing, draining or infilling of wetland on coastal floodplains. A growing area of interest is the capacity of wetlands to improve water quality by reducing pollutant concentrations and loads through biotic and abiotic processes. This is especially relevant to GBR catchments, where anthropogenic and climatic stressors such as increased loads of nutrients, sediments, and pesticides can pose significant risks to aquatic ecosystems.

This report provides evidence to demonstrate the current understanding of the effectiveness of wetlands in water quality improvement. It is underpinned by four questions in the 2022 Scientific Consensus Statement which reviewed GBR, national and international evidence, and previous syntheses where relevant. The report was based on international, national and local (GBR) evidence including 238 tropical and sub-tropical studies in agricultural areas and 145 studies in non-agricultural areas, and 17 studies from the GBR across different wetland and treatment systems. The report draws on and will inform Wetland*Info,* a comprehensive resource developed by the Queensland government to synthesise science to support managers and decision making for wetland protection and management in Queensland. The key findings will also inform the development of a wetland hydrology and water quality model for the GBR. These interactions are represented in Figure i.

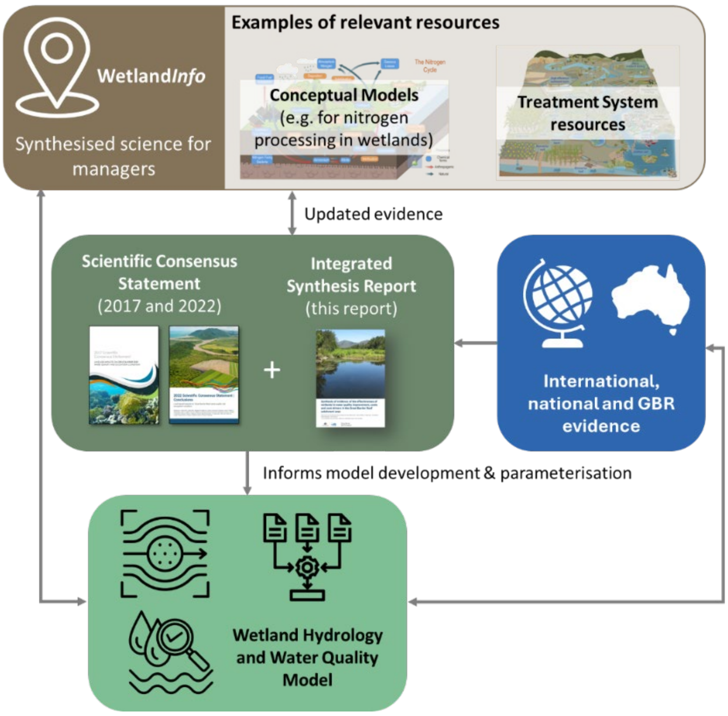

Figure i. High level illustration of the role of this review, the 2022 Scientific Consensus Statement, WetlandInfo, and the development of a Wetland Hydrology and Water Quality Model for the GBR, and the interactions.

Wetland and treatment systems and their effectiveness

The review scanned over 2,400 publications and drew on almost 400 global studies that have examined the water quality improvement efficiency of natural, restored, and constructed wetlands in agricultural and non-agricultural areas of tropical and subtropical regions. In studies that were conducted in the GBR, 63 wetlands (including natural, constructed, and bioreactors) were investigated, showing varying efficacy depending on the wetland type and pollutant.

The report evaluated the ability of several wetland and treatment systems to reduce pollutants:

Natural and Near-Natural Wetlands

- Global evidence shows that natural and near-natural wetlands, which include lacustrine, palustrine, and riverine wetlands, are typically more effective at nutrient and pesticide removal than constructed or restored wetlands, and that sediment is often retained in wetlands but can be remobilised in large flow events.
- Their performance depends heavily on hydrological factors including residence time and flow, vegetation cover, pollutant concentrations and loading rates which influence rates of denitrification. High denitrification rates can lead to high removal of nitrogen, particularly in first flush events. Only a few studies in the GBR have measured denitrification rates in palustrine and lacustrine wetlands.
- The review of global studies suggest that natural wetlands can remove nitrogen at efficiencies of 63% and phosphorus at efficiencies of 75%, and near-natural wetlands at efficiencies of 34% for nitrogen and 54% for phosphorus, although this varies across different wetland systems and environmental conditions. These rates refine the findings of the 2017 Scientific Consensus Statement and are now supported by a much larger evidence base.
- There is limited evidence about the effectiveness of sediment processing in natural and near-natural wetlands, with some trapping occurring, and very few studies for pesticides globally. There are no studies that measure pesticide/herbicide removal in GBR wetlands, only studies that measure *in situ* concentrations. However, the potential negative impacts of pesticides on wetland ecosystems are an important consideration.

Constructed/Treatment Wetlands

- Generally, constructed/treatment wetlands are effective in nitrogen removal if appropriately designed and built, with effectiveness increasing over time if the system is properly maintained.
- Constructed wetlands in the GBR catchment area have demonstrated nitrogen removal rates ranging from -5% to 90%, depending on design, vegetation, and inflow concentrations. Studies have demonstrated that constructed wetlands with higher vegetation cover (which promotes denitrification) and optimised hydrology perform better.
- There is more confidence in the evidence about the effectiveness of treatment wetlands than other wetland systems due to the higher number of studies (including in the GBR) and the fact that measuring removal efficiencies in constructed systems is more straightforward than in natural wetlands.
- Bioreactors, which use organic materials like woodchips to enhance microbial nitrogen removal, have demonstrated significant potential in agricultural settings when effectively designed and strategically placed. For instance, studies in the GBR have shown that bioreactors can remove up to 80% of nitrogen, with performance largely dependent on nitrate concentration, subsurface flow pathways and water flow rates. However, challenges related to design (including placement in the landscape) and maintenance can impact their overall effectiveness. There were limited studies that reported removal efficiencies for total phosphorus, sediments, and pesticides.
- Although less studied in the GBR, stormwater wetland systems in urban areas have shown global success in treating runoff when properly designed and maintained. These systems are particularly effective in removing coarse sediments and nutrients from urban stormwater.

Factors influencing wetland performance

Several factors influence the ability of wetlands to remove pollutants (including nutrients, sediments and to a lesser extent, pesticides):

- **Hydrology:** Wetland performance is highly dependent on water flow, water residence time, loss pathways, overall connectivity within the landscape and wetland size relative to the contributing catchment area. Wetlands that retain water flow for longer periods tend to remove more pollutants, particularly nitrogen and phosphorus. Different forms of nitrogen can be removed at significantly different rates, with observations that total nitrogen can be removed more effectively in static water relative to flowing water, and that ammonia removal is highest in high-flowing waters. These are important considerations for the placement and design of wetland and treatment systems.
- **Vegetation:** The type and density of wetland vegetation play a critical role in pollutant removal. Dense, diverse vegetation improves nitrogen cycling, facilitates sedimentation, and supports denitrification, a key process for nitrogen cycling and removal. The establishment of productive vegetation communities can take time, resulting in increased effectiveness in more mature systems.
- **Nitrogen load:** Wetland performance is closely related to the direct inputs it receives, specifically nitrate. Higher concentrations of nitrate entering the system can enhance its ability to process nitrogen, but if not properly managed, excessive input may overload the system leading to diminished effectiveness.
- **Wetland configuration:** The size, shape, and landscape position of a wetland influence its effectiveness in pollutant removal. Wetlands that are well-integrated into their surrounding environment particularly in terms of hydrological characteristics tend to perform better over time.
- **Maintenance:** Ongoing performance is dependent on continued and regular maintenance to maintain hydrologic and vegetation processes within the system.
- **Climate and seasonal variability:** Climatic factors such as rainfall, temperature, and extreme weather events (e.g., floods or droughts) significantly affect the performance of wetland systems, impacting their capacity to retain or process pollutants.
- **Other factors:** Additional factors can be important and include landscape and local conditions including upstream land use characteristics, and biogeochemical processes within the wetland including carbon-nitrogen ratios, sediment processes and microbial communities.

While the importance of these factors varies between different wetland and treatment systems, hydrology and vegetation are key drivers to pollutant processing in most systems. Other pollutants, such as pesticides and sediment can be reduced within wetland systems, but can cause negative effects such as vegetation loss, siltation and impacts on biological communities. Assessment of the impacts of pollutants on wetland ecosystems in the GBR is outside of this review but is reviewed in other sources such as the 2022 Scientific Consensus Statement and Wetland*Info*.

Table i presents a summary of the main findings for each wetland and treatment system and type from this review in relation to their effectiveness in water quality improvement and influencing factors.

Policy, cost and investment considerations

The Australian and Queensland governments have implemented a range of policies and programs to manage wetlands within the GBR catchment, emphasising their ecological importance and the need for sustainable management practices. These initiatives, including the Queensland Wetlands Program and the *Reef 2050 Wetlands Strategy - A strategy to manage wetlands in the Great Barrier Reef and its catchments*, provides the overarching direction for wetland science, planning, coordination and management in the GBR and its catchments. It promotes an integrated approach among the many government and non-governmental organisations, Traditional Owners and Custodians of Country, landowners and managers, businesses and industries involved in wetland and catchment management activities. These programs, supported by tools like Wetland*Info*, enable better decision-making and resource allocation for wetland management. Market-based instruments, such as the Australian Carbon Credit Units, offer financial incentives for landholders to engage in wetland restoration. Options for credits for water quality improvement activities are currently being developed. While these programs highlight the potential for co-benefits like carbon sequestration and biodiversity which can help offset costs, further research is required to quantify the benefits.

Wetland restoration projects can come with significant costs, influenced by factors such as hydrology, location, and design requirements. International studies highlight key cost drivers for wetland construction and restoration, including design, construction, and ongoing maintenance. Understanding these cost drivers is crucial to designing effective and sustainable wetland projects, particularly with regard to incentivising landholder participation and ensuring long-term sustainability. Key considerations for assessing cost-effectiveness of projects and programs include:

- **Initial construction and design:** Upfront costs vary widely based on wetland or treatment system and complexity. Constructed wetlands, in particular, require significant investment in design and setup to optimise pollutant removal. These systems can also take longer to reach optimal performance following establishment.
- **Maintenance and monitoring:** Ongoing operational costs, including regular monitoring and maintenance, are critical to sustaining wetland performance. In addition, long-term costs increase when landholder participation or intensive management of the system is required.
- **Economic and environmental co-benefits:** Wetlands can provide multiple co-benefits (intrinsic values and ecosystem services) including habitat provision, carbon sequestration, and recreational opportunities, which enhance their overall value. The integration of these services into project planning can improve cost-effectiveness, especially when aiming to deliver both environmental and socio-economic outcomes.

The current understanding of cost variations between different wetland and treatment systems is constrained, although international research offers insights, albeit with contextual challenges. While detailed cost data is limited for GBR wetlands, there is one GBR study that reported consistent cost metrics across wetland sites, each with different wetland designs in different contexts, land uses and size. All but one site applied the same discount rate and timeframe and reported both the annualised cost per hectare and the total cost per hectare.

GBR evidence base, knowledge gaps and future work

The evidence in this report specific to the GBR includes 17 studies comprising both published studies and reports, primarily focusing on nitrogen removal, with few studies on sediment processing and none for pesticides. Most studies in the GBR were conducted after 2019 and were concentrated in the Wet Tropics and Dry Tropics, with few studies from other regions. The GBR studies show significant variability in methodologies, monitoring approaches, and hydrological conditions, which limits the ability to make comparisons of the effectiveness of different wetland and treatment systems in water quality improvement.

Despite recent and significant advancements in this research area in the GBR, knowledge gaps remain. Many of these have been known for some time but have not been fully addressed. Priority needs include:

- **Long-term monitoring and modelling to quantify water quality outcomes**: There is a need for consistent, long-term monitoring and supporting models to track and quantify the effectiveness of different wetland systems in pollutant removal over time. This information is required in different settings (i.e., land uses, groundwater contribution, climates, and soils), with configuration of multiple systems in the landscape to understand the spatial and temporal drivers of variability, quantification of delivery pathways (surface and groundwater), pollutant removal efficiencies and potential pollutant stores (particularly those found to impact GBR ecosystems such as pesticides), and evidence of the timescales over which management interventions are likely to be effective. Quantification of the role of natural, and to a lesser extent, near-natural wetlands in long-term water quality management is particularly limited. Importantly, many studies are site specific and do not report results in the context of total pollutant load or volume entering the GBR, which can be improved through modelling.
- **Standardising monitoring methods to evaluate performance**: Inconsistent methodologies across studies limit the ability to compare wetland performance globally and at local scales. A standardised approach to measuring pollutant loads and wetland efficacy is needed to inform better management and investment decisions for the GBR.

- **Understanding wetland function under changing climate conditions:** The impacts of climate change including more frequent extreme weather events on wetland systems need to be further explored to develop robust wetland management strategies.

Future research and investment should focus on addressing these gaps to enhance our understanding of the role of wetlands in water quality management and improve the planning, design, and maintenance of wetland systems in the GBR.

Conclusions

Wetlands can play a vital role in improving some water quality parameters within the GBR catchment area, and provide significant ecosystem services that support biodiversity and community well-being. However, there has been significant historical loss of natural wetlands, particularly in floodplains, in some areas and degradation of the condition of those wetlands remaining. The evidence shows that natural/near-natural and constructed/treatment wetlands can be effective in water quality improvement in certain conditions, however, it is crucial to equally prioritise wetland health, intrinsic values and the ecosystem services they provide. There are several important considerations for maximising pollutant removal efficiency and determining future protection and management opportunities for GBR wetlands. These include the hydrological characteristics (water flow and residence time), presence of vegetation, type and density, pollutant concentrations and loads, wetland configuration, climate and seasonal variability, biogeochemical processes, and the need for ongoing and long-term maintenance of wetlands and treatment systems. The performance of the system in water quality improvement depends on careful design, ongoing maintenance, and integration within the broader landscape. To maximise the benefits of wetland and treatment systems, it is essential to invest in long-term monitoring, research, and adaptive management strategies such as the Whole-of-System, Values Based Framework that account for the diverse services that wetlands provide and the challenges they face in a changing environment.

Table i. Summary of the main findings for each wetland and treatment system and type from this review; n=number of studies. TN = Total nitrogen, TP = Total phosphorus, TSS = Total suspended sediments/solids and NH_4^+ = Ammonium, NO_3^- = Nitrate, PO_4 = Phosphate, TDN = Total dissolved nitrogen, DIN = Dissolved inorganic nitrogen, TIP = Total inorganic phosphorus.

Wetland System	Wetland Type (Section 2)	Contextual information (See Section 2[1])	Effectiveness in water quality improvement (Section 3)	Main factors influencing water quality improvement (Section 4)
Natural/ near-natural	Overall	For this review, natural and near-natural wetlands refer to lacustrine, palustrine and riverine wetlands, excluding subtidal and subterranean wetlands, estuarine wetlands, coral reefs, seagrass meadows, oyster reefs and aquifers. Natural wetlands refer specifically to wetlands without any anthropogenic structural or hydrological change either to the wetland itself, or within its catchment. Near-natural wetlands refer to wetlands without any anthropogenic structural change to the wetland, but with anthropogenic structural or hydrological change occurring within the broader catchment.	From the global review, natural (n = 5) and near-natural (n = 6) wetland studies that reported annual loading and removal rates of TN, showed an average removal efficiency of 64% and 34% respectively. For TP, the average removal efficiency of natural wetlands was 75% (n = 3) and 55% (n = 2) for near-natural wetlands. For TSS, natural wetlands had an average removal efficiency of 45% (n = 2, ranging from -1 to 91%). From the body of evidence, natural and near-natural wetlands were reported to remove NH_4^+ the most efficiently (64% and 48%).	Vegetation (including vegetation community composition, species and density), water flow rate and hydraulic residence time, inflow and loading rates of nutrients.
	Palustrine	Palustrine wetlands are vegetated (more than 30% emergent vegetation), non-riverine or non-channel systems. They include billabongs, swamps, bogs, springs, soaks etc.	From the global evidence, there are not sufficient studies that clearly identify the wetland as palustrine with reported pollutant removal efficiency. One study conducted in the GBR measuring denitrification rates across 10 palustrine wetlands, found rates varying between 1.1 and 52 mg m^2 h^{-1} which can convert to TN removal of up to 80% and NO_3^- removal up to 70%.	Nitrate inflow/ concentration in the water column, vegetation cover, soil/sediment composition/content, and landscape context.
	Lacustrine	Lacustrine wetlands (lakes) are dominated by open water. Although lakes may have fringing vegetation, most of the wetland area is open water.	From the global evidence, there are not sufficient studies that clearly identify the wetland as lacustrine with reported pollutant removal efficiency. The one study conducted in the GBR on two coastal lagoons found denitrification rates of 24 and 52 mg m^2 h^{-1}.	Vegetation (macrophytes), hydrology, depth, and nutrient concentrations.
	Riverine	Riverine wetlands are all wetlands and deepwater habitats within a channel, periodically or continuously containing moving water. Riverine wetlands include	From the global evidence, riverine wetlands (natural and near-natural) studies showed on average, a TN removal of 45% (n = 5, range 24 to 88%), a TP removal	Vegetation, hydrology, landscape context, and environment factors.

Wetland System	Wetland Type (Section 2)	Contextual information (See Section 2[1])	Effectiveness in water quality improvement (Section 3)	Main factors influencing water quality improvement (Section 4)
		rivers, streams, creeks, brooks, rivulets, canals, channels, watercourses and tributaries.	of 37% (n = 4, range 10 to 80%). There were no studies on the removal of pesticides and TSS. There is only one study conducted in the GBR in a riverine wetland, and this wetland showed limited efficacy in removing TN or TSS and a small reduction in TP of 14%.	
Constructed	Treatment wetland Other names: Constructed wetlands, landscape wetlands, embellished wetlands, surface flow wetlands, free-water wetlands	Treatment wetlands are engineered systems that replicate and enhance the physical, biological and chemical treatment processes occurring in natural wetlands to remove fine sediments, nutrients and other pollutants (e.g., pesticides, heavy metals).	From the global evidence, treatment wetland studies in agricultural areas showed on average, a TN removal of 46% (n = 40, range -4 to 97%), a TP removal of 49% (n = 38, range 2 to 97%), TSS removal of 57% (n = 10, range 1 to 94%), pesticide removal of 69% (n = 16, range 4 to 100%), NH_4^+ removal of 64% (n = 11) and PO_4 removal of 38% (n = 5). From the evidence in the GBR (7 studies) the efficacy of treatment wetlands for TN ranged between -5 and 100%, for TDN between -5 and 50%, for DIN -15 and 90%, for NO_3^- between -30 and 100%, NH_4^+ between -90 and 50%, and one study found the efficacy in removing sediments to be equal to 86%. Free flow stormwater wetlands in Melbourne showed a TN removal was 41% (-36 to 70%) and performed poorly in removing TSS and TP from urban stormwater.	Influent loads, macrophyte presence, open water surface area, sediment basin, high-flow bypass, bathymetry, depth, shape and length to width ratio and retention time, nitrate, low oxygen levels and appropriate redox conditions, and wetland age.
	Floating wetland (CFW)	Floating wetlands consist of a suspended matrix planted with wetland plants. This facilitates microbiological and plant processing of nutrients. Floating wetlands work by encouraging settling and biological processing of suspended sediments, particulate and dissolved nutrients and pollutants, and also by directing the water through the suspended root mass.	No studies were identified that review the effectiveness of treatment wetlands in pollutant processing in urban or agricultural areas in the GBR. Three studies conducted at the same site (2 CFWs) in southeast Queensland showed a TN removal of 17%, TSS removal of 80% and a TP removal of 52%. For agricultural areas in the global review, seven studies looked at floating treatment wetlands, with three of these providing removal efficiencies for nutrients and pesticides, which showed an average removal efficacy of 10% for TP (n = 1, range 2 to 15%), 7% for TN (n = 1, range -6 to 31%), 22.6% for NO_3^- (n = 2, range -13 to 78.4%), and a range of 4 to 74% removal of pesticides (n = 1).	Influent loads, root mass, root length, depth, anchoring, hydrology, plant selection, and detention time.

Wetland System	Wetland Type (Section 2)	Contextual information (See Section 2[1])	Effectiveness in water quality improvement (Section 3)	Main factors influencing water quality improvement (Section 4)
	Recycle pits	A recycle pit is a structure designed to capture irrigation runoff (known as tailwater) for re-use in the production area. Unlike many other treatment systems used in agricultural production systems recycle pits do not treat the water. Rather they rely on the water being captured and re-used on the farm.	One study modelled the efficacy of recycle pits in agricultural waters in the GBR and reported an overall DIN removal efficacy between 1 and 89%, and DIN removal average efficacy in wet weather was 37% (range 7 to 89%).	Size, bypass system, base and walls, capacity and pumping design.
	Vegetated drains Vegetated buffers and swales	Vegetated drains are open channels for conveying water, where vegetation covers most of the banks and bed. The difference between drains and swales, is that swales are dry most of the time whereas drains often hold water for extended periods.	No studies were identified that review the effectiveness of swales in pollutant processing in the GBR in non-agricultural areas. Two studies were reviewed from South-East Queensland. One study in Brisbane showed a TN removal of 44 to 57% and the other in the Sunshine Coast showed only limited effectiveness of swales for removing nitrogen. Global evidence in agricultural land uses showed an average TN reduction of 43% (n = 3, 13 observations, range -5.4 to 76%), a 57% NO_3^- reduction (n = 2, 7 observations, range 49 to 64%), 57% TP reduction (n = 1 , 9 observations, range 24 to 75%), 31% TIP reduction (n = 2, 7 observations, range 19 to 42%), 79% TSS reduction (n = 2 , 3 observations, range 68 to 89%), pesticide removal was also recorded (up to 99% Imidacloprid, up to 100% permethrin and 23% for chlorpyrifos). In the GBR, one study examined the efficacy of vegetated drains in removing DIN and found an efficacy ranging from 50 to 80%.	Plant community density, root density, and flow rates.
	Riparian buffer strips (RBSs)	Riparian buffer strips are vegetated areas that separate waterways from agricultural activity and other land uses.	Global evidence found an NO_3^- average removal of 59% (n = 3, multiple sites, range 3 to 99%), TN average removal of 44% (n = 2, multiple sites, range -43 to 66%), TP average removal of 13% (n = 2, multiple sites, range -33 to 64%), TSS average removal of 11% (n = 1, 3 sites, range -51 to 46%). In the GBR, one study identified the efficacy of riparian buffers in processing pollutants. For TN, efficacy	Vegetation, width, slope. Larger widths were associated with better performance.

Wetland System	Wetland Type (Section 2)	Contextual information (See Section 2[1])	Effectiveness in water quality improvement (Section 3)	Main factors influencing water quality improvement (Section 4)
			ranged between -43 and 45%, for TP between 33 and 64% and for TSS, efficacy ranged between 8 and 46%.	
	Algae treatment	Wetland algae treatment uses harvested freshwater or saline/marine macroalgae cultivated in ponds to remove pollutants. Algal biomass is regularly harvested.	Global evidence found one study in algal ponds in Korea that had a removal rate of 85% for TN and 89% for TP. There are no studies in algal ponds in the GBR.	Hydrology (specifically detention time and perennially), wetland design and size, inflow DIN concentration, bypass, and algae species.
	Sediment basins	Sediment basins are designed to trap and store sediment and debris through the process of sedimentation.	No evidence from the global or the GBR literature search was found.	
	Bioreactors	Bioreactors are systems designed to intercept surface and/or subsurface flows and direct them through anoxic zones that have high carbon content, usually through the addition of woodchips or other organic matter. These anoxic zones promote denitrification which can be particularly effective in removing DIN.	From the global evidence (agricultural), bioreactors studies showed TN removal of 80% (n = 1), there were no results for TP or TSS, and pesticide removal was 47% (n = 2, range 14.3 to 100%). From the evidence in the GBR (agricultural), bioreactors show a TN reduction of 41% (n = 1), NO_x reduction of 41% (n = 1), NO_3^- reduction of 84% (n = 1), and a pesticide removal of 40% (n = 1). No specific studies were identified that reviewed the effectiveness of biofilters in pollutant processing from urban stormwater, in the GBR. From the seven studies conducted elsewhere in Australia, biofilter studies showed TN removal between 47 and 89% (n = 5), TP removal between 40 and 93.5% (n = 5), and TSS removal between 68 and 90% (n = 2).	Hydrology, vegetation, carbon content, dissolved oxygen concentrations and interception of the sub-surface flow path.
Restored		Restored or rehabilitated wetlands refer to wetlands where ecological and/or hydrological processes have been recovered where natural wetlands previously existed.[2]	From the global evidence, restored wetlands showed a TN reduction of 38% (n = 1), TP reduction of 52% (n = 2, range 26 to 59%), TSS reduction of 35% (n = 2, range -4 to 74%) and there were no results for pesticides. For urban stormwater one study outside of the GBR on restored wetlands showed no reduction of pollutants.	As above.

[1]DESI (2022a); [2]Note that Wetland*Info*, an extensive resource for information about the status and management of wetlands in Queensland, separates restored and rehabilitated wetlands into two different categories: Rehabilitated wetlands are wetlands where actions or interventions have sought to repair, enhance and/or replace ecosystem processes and/or components, to improve intrinsic values and/or ecosystem services; Restored wetlands are the result of actions that return the wetland to a former, original, normal, or unimpaired condition (DESI 2022b).

1. Introduction

Wetlands are highly significant ecosystems due to their intrinsic values and the broad range of ecosystem services they provide. These services include biophysical functions (e.g., nutrient cycling), biological contributions (e.g., biodiversity), environmental benefits (e.g., flood control and foreshore stabilisation), economic values (e.g., tourism) and cultural importance (e.g., aesthetic) (Sah and Heinen 2001; Fisher et al. 2011; Findlay and Fischer 2013; Gopal 2013). Wetlands can contain permanent or temporary water depending on their attributes and composition (Boulton et al. 2014) and in some cases the water is extracted for use by humans. Wetland water quality is highly variable and is influenced by hydrological interactions (surface and groundwater), rainfall patterns, weather conditions, human use, runoff from adjacent land uses and the presence of biota (including weeds and pests). Land use and other modifications to natural processes can significantly alter wetland water quality, shifting it from natural cycling conditions to degraded or modified states. This is particularly relevant in the Great Barrier Reef (GBR) catchment area, where it is widely acknowledged that wetlands play an integral role in contributing to the overall health and condition of the broader GBR ecosystem (DESI 2023a).

Since European settlement (c.1850), land-use changes and extensive modifications to floodplains in the GBR catchment area have contributed to the degradation and loss of wetland habitats, along with the critical ecosystem services they provide. While 78 to 97% of pre-European wetlands remain, this proportion varies widely between Natural Resource Management (NRM) regions and basins. Coastal floodplains, for example, have experienced substantial wetland losses, with the Wet Tropics NRM region losing 30.5% of its pre-development wetland extent and the Burnett Mary NRM region losing 28.5% by 2017 (DESI 2019). Palustrine wetlands have been particularly affected, with losses of 48.6% in the Wet Tropics and 43.6% in the Mackay Whitsunday NRM region (DESI 2019).

These losses are primarily due to changed land management including land clearing, draining or infilling of wetland on coastal floodplains. Although wetland loss rates have generally slowed in recent decades with slight increases in overall wetland extent observed in some catchments due to the construction of artificial wetlands (e.g., farm dams), natural wetland areas continue to decline. Between 2011 and 2017, the GBR catchments saw a net loss of 7,688 ha of natural and near-natural wetlands (i.e., excluding artificial/highly modified), with riverine wetlands experiencing the greatest reductions (DESI 2019). While the ecological implications of these changes are challenging to quantify, conceptual understanding suggests that wetland degradation has significantly modified the biological, biogeochemical, and physical functions critical to supporting the health of the GBR and its connected freshwater and coastal ecosystems (Creighton et al. 2021; Great Barrier Reef Marine Park Authority 2012).

A growing area of interest is the capacity of wetlands to improve water quality by reducing pollutant concentrations and loads through biotic and abiotic processes. This is especially relevant to GBR catchments, where anthropogenic and climatic stressors such as increased loads of nutrients, sediments, and pesticides (herein referred to collectively as 'pollutants'), pose significant risks to aquatic ecosystems (refer to Waterhouse et al. (2024) for a summary of ecological impacts).

Wetland systems are diverse, and their function depends on their type. In Queensland, wetlands are defined in the Queensland Wetland Definition Guideline (DESI 2023c) as 'areas of permanent or periodic/intermittent inundation, with water that is static or flowing, fresh, brackish or salt, including areas of marine water the depth of which at low tide does not exceed six metres. To be a wetland the area must have one or more of the following attributes:

- at least periodically, the land supports plants or animals that are adapted to and dependent on living in wet conditions for at least part of their life cycle, or
- the substratum is predominantly undrained soils that are saturated, flooded or ponded long enough to develop anaerobic conditions in the upper layers, or
- the substratum is not soil and is saturated with water or covered by water at some time.

Lakes, swamps, marshes, billabongs, rivers, creeks, fens, peat bogs, saltmarshes, mudflats, and mangroves are all wetlands. Queensland even has underground wetlands. Wetlands can include marine plants (e.g., seagrass), coral and other GBR species and form part of the wider GBR. This report, however, focuses on freshwater wetlands (lacustrine, palustrine and riverine wetland systems) and does not include estuarine and marine wetlands.

Wetlands play a dual role in the GBR: they contribute to its outstanding universal values, while also contributing to improving water quality. They also provide direct connectivity through water flows and faunal movement. The Reef 2050 Long Term Sustainability Plan (Reef 2050 Plan; Australian and Queensland governments, 2023) and Reef 2050 Water Quality Improvement Plan (Reef 2050 WQIP; (Australian and Queensland governments, 2017) recognise that natural wetlands are important for reducing pollutants and the benefits of applying a whole-of-catchment approach to improve water quality and the health of GBR ecosystems (DESI 2023a). Further discussion of the policy context is provided in Section 5.

Efforts to incorporate wetlands into the overall treatment regime in the GBR to improve water quality, both through rehabilitating existing degraded systems and new constructed systems, requires a comprehensive understanding of wetland processes, the key factors influencing these processes and their role within the landscape. While a significant body of work has already been completed and synthesised in this field (DESI 2021 Wetland*Info*), further knowledge is needed to determine when and where wetlands can most effectively enhance water quality. This understanding is crucial to protect and enhance their existing functions while minimising future declines. Importantly, using wetlands for water quality improvement must not compromise their intrinsic values or other ecosystem services they provide (DESI 2023a).

This report provides a comprehensive synthesis of the evidence supporting these concepts and addresses some of the current knowledge gaps in this area. The report builds on previous documented evidence in this field and expands it by reviewing additional national and international literature. It does not make specific management recommendations but complements the *Reef 2050 Wetlands Strategy: A strategy for managing wetlands in the Great Barrier Reef and its catchments* (Reef 2050 Wetlands Strategy, DESI 2023a), which provides overarching direction for wetland science, planning, coordination and management in the GBR and its catchments, and importantly takes a whole-of-ecosystem approach to the connection between catchments, wetlands and the reef ecosystems.

1.1 Project background

In 2021 the Great Barrier Reef Foundation (GBRF), became engaged in investment in wetland rehabilitation, restoration and construction for the objective of water quality improvement in the GBR catchment through the Reef Trust Partnership.

The role of wetlands in water quality improvement was highlighted as a key knowledge gap in the 2017 Scientific Consensus Statement (see Eberhard et al. 2017). While several significant syntheses and scientific papers have documented this information for many years, over the last five years, there have been a range of global and GBR studies that have helped to further advance our understanding of the role of wetlands in water quality management (e.g., Eberhard et al. 2017; Schaffelke et al. 2017; Waterhouse et al. 2017; Adame et al. 2021a and represented in Wetland*Info*). The evidence base for the Reef 2050 Wetlands Strategy is documented through the Wetland*Info* website developed by the wetlands team within the Department of Environment, Tourism, Science, and Innovation (DETSI). This resource reflects the work of DETSI and research partners in demonstrating current understanding of wetland processes and functions (e.g., Adame et al. 2021) and management in the GBR. Another project commissioned in 2022 aims to establish a model to evaluate wetland performance in pollutant removal to help in the planning of treatment wetland projects in the GBR catchments, based on previous work commissioned by the Queensland Water Modelling Network (Alluvium 2021). In addition, there are several other recently commissioned projects including the establishment of a detailed water quality monitoring project for natural wetlands in the catchments of the GBR.

To inform future investment programs, GBRF commissioned this project to collate and synthesise evidence and develop robust scientific consensus regarding the efficacy and value of wetlands for water quality improvement in the GBR. Robust scientific consensus was sought in the following areas of evidence:

- Knowledge of the effectiveness of wetland systems in the processing of sediments, nutrients and pesticides.
- The key wetland characteristics or attributes required to effectively improve water quality.
- The cost-effectiveness of investing in wetland construction, restoration or rehabilitation for the purpose of reducing pollutant inputs to the GBR.
- Considerations for accounting for the wider ecosystem services and benefits of wetlands including wetland values, health and function.

C_2O Consulting was engaged in November 2021 to deliver this project in conjunction with their role in the coordination of the 2022 Scientific Consensus Statement (SCS). This report is underpinned by four questions in the 2022 SCS which reviewed GBR catchment area, national and global evidence, and previous syntheses where relevant. The report draws on and will inform Wetland*Info*, and the key findings will also inform the development of a wetland hydrology and water quality model for the GBR. These interactions are represented in Figure 1.

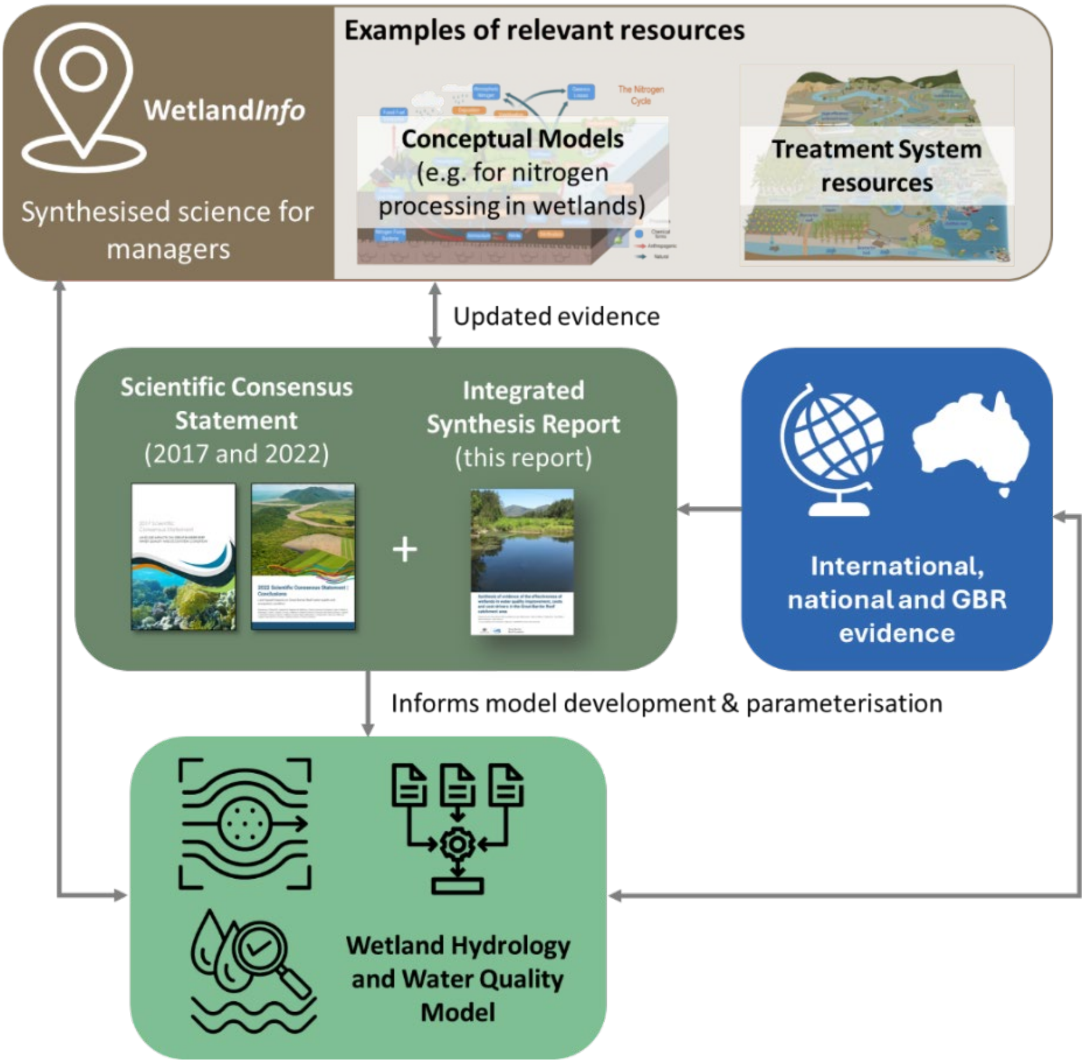

Figure 1. High level illustration of the role of this review, the 2022 Scientific Consensus Statement, WetlandInfo, and the development of a Wetland Hydrology and Water Quality Model for the GBR, and the interactions.

1.2 Links to the 2022 Scientific Consensus Statement

The *2022 SCS on land-based impacts on GBR water quality and ecosystem condition* brings together the latest scientific evidence to understand how land-based activities can influence water quality in the GBR, and how these influences can be managed. The SCS is updated periodically with the latest peer reviewed evidence and the 2022 SCS builds on previous Statements which were published in 2002, 2008, 2013 and 2017[1]. The SCS is used as a key evidence-based document by policymakers when making decisions about managing GBR water quality. In particular, the SCS provides supporting information for the design, delivery and implementation of the Reef 2050 WQIP. The Reef 2050 WQIP describes actions for improving the quality of the water that enters the GBR from the adjacent catchments. All of the Statements have included information about wetlands in the GBR, with greater emphasis since 2017.

The 2017 SCS highlighted the critical role of remnant wetlands in the GBR catchment, including their contributions to biodiversity, aesthetic and cultural values, and their role in connecting freshwater and marine ecosystems. Wetlands were recognised for delivering ecosystem services such as water quality improvement and carbon storage (Schaffelke et al. 2017). Their capacity to process and filter land-based pollutants was also noted, however, the detrimental effects of these pollutants on wetlands was also acknowledged. Additionally, the SCS identified that engineered treatment systems (technologies such as constructed wetlands, denitrifying bioreactors, floating wetlands, high efficiency sedimentation basins and algae nutrient removal) could be effective as a management option for reducing sediment, nutrients and pesticides loads entering the GBR (Eberhard et al. 2017).

The 2022 SCS addressed 30 priority questions developed in consultation with scientific experts, policy and management teams and other key stakeholders (e.g., representatives from agricultural, tourism, conservation, research and Traditional Owner groups). These questions were categorised into eight themes: values and threats, sediments and particulate nutrients, dissolved nutrients, pesticides, other pollutants, human dimensions, and future directions. The themes explored topics ranging from ecological processes, delivery and source, through to management options.

This report uses the outputs of four questions (see Table 1) that specifically addressed wetlands-related topics. These questions were answered using a structured synthesis of evidence approach (based on peer reviewed and publicly available literature, and typically incorporated evidence from 1990 to the end of 2022.

At the beginning of the process, it was agreed that further investigation of the best evidence synthesis method to be applied was warranted as part of this GBRF project, noting that there were some concerns regarding the availability of relevant GBR evidence for answering these questions. Consequently, a Rapid Systematic Map of evidence was undertaken by Evidentiary and C_2O Consulting in 2021 / 2022 (Richards and Molinari 2023). This informed the selection of synthesis methods by evaluating:

- The volume and scope of the evidence base.
- Potential limitations for further synthesis including variations in relevant definitions in international literature, the relevance of climatic conditions, lack of quantified data of pollutant removal and uncertainty of the time period of studies reported.
- Appropriate synthesis methods including qualitative analyses where needed.

Two methods were ultimately adopted – an Evidence Summary and an Evidence Review. While largely consistent, these methods differed in the level of appraisal required with the Evidence Review requiring additional quality assessment.

[1] https://reefwqconsensus.com.au/process/previous-scientific-consensus-statements/

Table 1. Final questions included in the 2022 SCS relevant to the GBRF Wetland Synthesis Project.

SCS Question	Questions	Synthesis of evidence method	Reference
4.6	What are the most effective management practices for reducing dissolved nutrient losses (all land uses) from the GBR catchments, and do these vary spatially or in different climatic conditions? *Note that the non-agricultural section contains information specifically related to wetland treatment systems in urban areas.*	Evidence Review	Thorburn et al. 2024 (Non-agricultural component: T. Weber)
4.7	What is the efficacy of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality (nutrients, fine sediments and pesticides)?	Rapid Systematic Map Evidence Review	Richards and Molinari 2023 Waltham et al. 2024a
4.7.1	**Sub-question to 4.7:** What are the key factors that affect the efficacy of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality and how can these be addressed at scale to maximise water quality improvement?		
4.8	What are the measured costs, and cost drivers associated with the use of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality?	Scoped as part of Systematic Map for 4.8 Evidence Summary	Star et al. 2024
4.9	What role do natural/near-natural wetlands play in the provision of ecosystem services and how is the service of water quality treatment compatible or at odds with other services (e.g. habitat, carbon sequestration)?	Evidence Summary	Waltham et al. 2024b

1.3 Scope of this report

In this report, the 2022 SCS evidence syntheses are supplemented by additional GBR-specific evidence items such as technical reports, website material (primarily Wetland*Info*) and case study outputs that did not meet the 2022 SCS eligibility criteria of peer reviewed and publicly available evidence items, as well as expert advice (Appendix 1: Great Barrier Reef studies included in the review). Additional evidence items were screened and assessed in a similar way to the formal 2022 SCS process.

The report presents evidence for agricultural and non-agricultural areas and considers different wetland and treatment systems. It draws on international, national and local (GBR) evidence and covers:

- Wetland processes associated with pollutant processing.
- Understanding the effectiveness of wetlands in water quality improvement.
- Factors that influence the effectiveness of water quality improvement in wetlands.
- Policy, cost and investment considerations.

2. Wetland function and processes associated with pollutant processing

Wetlands, both natural and constructed, can play a significant role in water quality improvement by processing land-based pollutants. Different wetland systems and types are capable of processing land-based pollutants including sediments, nutrients and pesticides to varying degrees. Constructed wetlands come in various types, including surface flow, subsurface flow, and hybrid systems, each designed to optimise specific processes like sedimentation, filtration, and nutrient removal. Understanding the characteristics and functions of wetland systems is critical for quantifying their effectiveness in water quality improvement. This review categorises wetland systems into two broad groups: natural/near-natural and constructed/treatment wetlands.

It should be noted that the formal definitions adopted by the Queensland Wetlands Program are underpinned by hydrological characteristics and grouped into natural, modified, slightly modified, highly modified and artificial wetlands (DESI, 2023c).

2.1 Natural and near-natural wetlands

For the purposes of this review, **natural and near-natural wetlands** are defined as lacustrine, palustrine, and riverine wetlands. Subtidal, subterranean and estuarine wetlands are excluded, along with coral reefs, seagrass meadows, oyster reefs and aquifers. **Natural wetlands** refer specifically to wetlands without any anthropogenic structural or hydrological change to the wetland, or within its catchment. **Near-natural wetlands** refer to wetlands without any anthropogenic structural change to the wetland, but with anthropogenic structural or hydrological change occurring within the broader catchment. In the Queensland Wetland Mapping Method, these are defined as Natural[2] or Modified wetlands[3] (DESI 2023c).

Freshwater wetlands are generally hydraulic low points in the landscape that receive water from surface and subsurface pathways (Mitsch and Gosselink 1993), channelling it downstream to lower catchment and nearshore areas. Wetlands may provide an effective means of processing and storing land-based nutrients, sediments, and other contaminants, though their functionality can diminish if poorly managed or designed, such that wetlands can become a source of contaminants (Adame et al. 2021a; Moustafa et al. 2011). Other constituents, such as pesticides and sediment can be reduced within wetland systems, but in excess will cause significant deleterious effects such as vegetation loss and siltation.

Wetland functionality is influenced by numerous factors, including complex interrelationships between these factors. A review by Weber et al. (2021) identified 17 key processes and 22 separate configuration components that may be important in understanding wetland function. These are summarised in Table 2. While not all of these factors are necessarily linked to pollutant processing, they may still need to be assessed to understand wetland performance in tropical and subtropical environments in the GBR.

Table 2. Key wetland processes and components identified as important in understanding wetland function. Source: Weber et al. (2021).

Processes
Algal growth and decay
Adsorption/desorption
Biological uptake at plant/water column interface
Biological uptake at sediment/water column interface
Dissolution/flocculation
Litterfall/organic matter accumulation

Components (may be related to several processes)
Algal species
Biofilm species
Denitrification species
Depth
Dissolved oxygen
Inflow configuration

[2] *Natural wetlands* refer to wetlands where activities that modify wetland hydrology and/or structures associated with these activities cannot be observed from aerial or satellite imagery and are not known from field survey data.

[3] *Modified wetlands* are existing wetlands which were also former natural wetlands, where activities that modify wetland hydrology and/or structures associated with these activities have been observed from aerial or satellite imagery or from field survey data. *Slightly modified* wetlands are modified wetlands where the hydrological modifications allows the resulting wetland to retain many of their functional and ecological characteristics. *Highly modified* wetlands are modified wetlands where the hydrological modifications are considered to significantly degrade the wetland's functional and ecological characteristics.

Processes
Macroinvertebrate grazing/predation
Nitrification/denitrification
Nitrification/nitrogen fixing
Nitrogen assimilation/annamox /ammonification
Oxygenation/deoxygenation (including conditions related to the presence of potential Acid Sulfate Soils)
Photosynthesis/respiration
Plant uptake (root zone)
Sediment/water column nutrient flux
Sedimentation/resuspension
Stratification
Wetting/drying

Components (may be related to several processes)
Inflow volume
Nitrogen concentration
Organic carbon
Outflow configuration
Perturbing events (cyclone, flood, feral animals, desilting, draining, clearing, weeds, vegetation failure)
pH
Plant density
Plant species
Redox
Sediment composition (including nutrients, toxicants, particle size etc.)
Shape
Surface area
Temperature
Time (duration)
Time (frequency)
Time (seasonality)

Additionally, it is important to recognise that different types of natural wetlands will process pollutants differently, and the mechanisms that underpin this removal will also differ. Table 3 summarises the three floodplain wetland types and the main processes associated with nitrogen (N) removal in the GBR as outlined in Adame et al. (2021a).

Table 3. Description and main processes associated with nitrogen balance in natural wetlands based on Adame et al. (2021) and detailed in WetlandInfo.

Natural wetland system	Description	Main nitrogen processes
Palustrine	"Palustrine wetlands are vegetated (more than 30% emergent vegetation), non-riverine or non-channel systems. They include billabongs, swamps, bogs, springs, soaks etc. The vegetation in these wetlands includes grasses, sedges, shrubs, and trees. In the GBR, Melaleuca species are dominant."	Palustrine wetlands usually act as nitrogen sinks. These wetlands store large amounts of carbon and nitrogen in their soils. The soils are usually anoxic and favourable for **denitrification**. Higher denitrifications rates are also associated with high NO_3^- inflows. The high productivity of palustrine wetlands can lead to the accumulation of nitrogen in plant biomass with occasional export (litter, DON and NH_4^+) during floods.
Lacustrine	"Lacustrine wetlands (lakes) are dominated by open water. Although lakes may have fringing vegetation, the majority of the wetland area is open water."	Nitrogen cycling in lakes involves both aerobic processes at the surface and anoxic processes at the bottom. Nitrogen is integrated into the food web through phytoplankton, epiphytes and macrophytes (**plant uptake**). Lakes also experience **nitrogen fixation** in specific conditions. In the littoral zone, macrophytes can significantly contribute to nitrogen storage and processing. In terms of nitrogen export, plant biomass and litter can be flushed out, especially during rainfall events or storms.

Natural wetland system	Description	Main nitrogen processes
Riverine	"Riverine wetlands are all wetlands and deepwater habitats within a channel, periodically or continuously containing moving water. Riverine wetlands include rivers, streams, creeks, brooks, rivulets, canals, channels, watercourses and tributaries."	Riverine wetlands usually transform nitrogen. In-stream processes such as **denitrification, sediment storage** and **plant uptake** help remove nitrogen. Denitrification is less effective in rivers with shorter water residence times. In terms of nitrogen export, **nitrogen mineralisation** in sediments contributes to the export of inorganic nitrogen.

To facilitate understanding of wetland processes and system components which may be important for wetland function, the Queensland Department of Environment, Science and Innovation (DESI) developed a series of conceptual models. This resource is available on the Wetland*Info* website (DESI 2023b) and summarised here.

The models address three spatial scales: 1) the individual wetland scale (Figure 2), 2) the relationship with contributing catchments (Figure 3), and 3) the placement of wetlands within an overall landscape (Figure 4). Understanding these processes is particularly important in understanding how they may be simulated in numerical models, which is of particular interest to decision makers.

The conceptual model for the wetland site scale (Figure 2) illustrates inputs to a wetland hydrologic model, the hydrological and nutrient processes within the wetland, the wetland components that influence water flows and processes, and the outputs from a wetland model, with a focus on **nitrogen processes** (DESI 2023b). The following description is sourced directly from (DESI 2023b).

Forcing factors are the factors (e.g., temperature, evaporation, rainfall intensity, duration, volume, timing and event frequency, solar radiation, wind) that influence the model inputs, including the amount and flow of water in the catchment or area of interest and the constituents (e.g., nutrients and pesticides) carried by the water. **Landscape factors** affect the forcing factors and contribute to the resultant runoff and constituent inputs.

Wetland configuration (components) describes the physical components of the wetland itself, which in turn affect flow within the wetland (e.g., size, depth, shape, bathymetry, inflow configuration (controls and inlet type such as pipe, channel, rock weir)), and vegetation configuration (species, shape, density).

Model inputs are water inputs to the wetland which can be divided into: overland flow, channelised flow, and subsurface flow. The flow rates, volume, and timing (frequency, seasonality) of the inflows will vary depending on the forcing factors and landscape parameters (see contributing catchment conceptual model).

Wetland processes – the nitrogen processes in wetlands vary depending on the environment:

- Open water processes occur in the water column of the wetland, separate from the vegetated or benthic zones. Nitrogen processes include: nitrification, nitrogen fixing by algae, macroinvertebrate grazing and predation, algal growth and decay, stratification in terms of temperature or oxygen levels, adsorption and desorption, dissolution and flocculation of contaminants.
- Benthic zone processes occur in the sediment and substrate of the wetland. Nitrogen processes include: sedimentation, and resuspension, sediment nutrient flux, nitrification, denitrification, assimilation, annamox, ammonification, litterfall, organic matter accumulation, biological uptake and filtration including biofilms at the sediment water interface, and oxygenation/deoxygenation.
- Vegetated zone processes occur in the vegetated zones of the wetland. Vegetation provides carbon which is used by microbes in processes such as denitrification, and biofilms which host microbes. Nitrogen processes include: denitrification, nitrification, nitrogen fixing, nitrogen assimilation, plant uptake in the root zone, litterfall, biofilm growth, decay, sloughing and scour, wetting and drying, adsorption, desorption, photosynthesis, respiration, and transpiration.

Model outputs are water outputs from the wetland which can be divided into: bypass or overflow, channelised flow, and subsurface flow. The flow rates, volume, and timing (frequency, seasonality) of the outflows will vary depending on the inputs, and water flow and storage within the wetland itself.

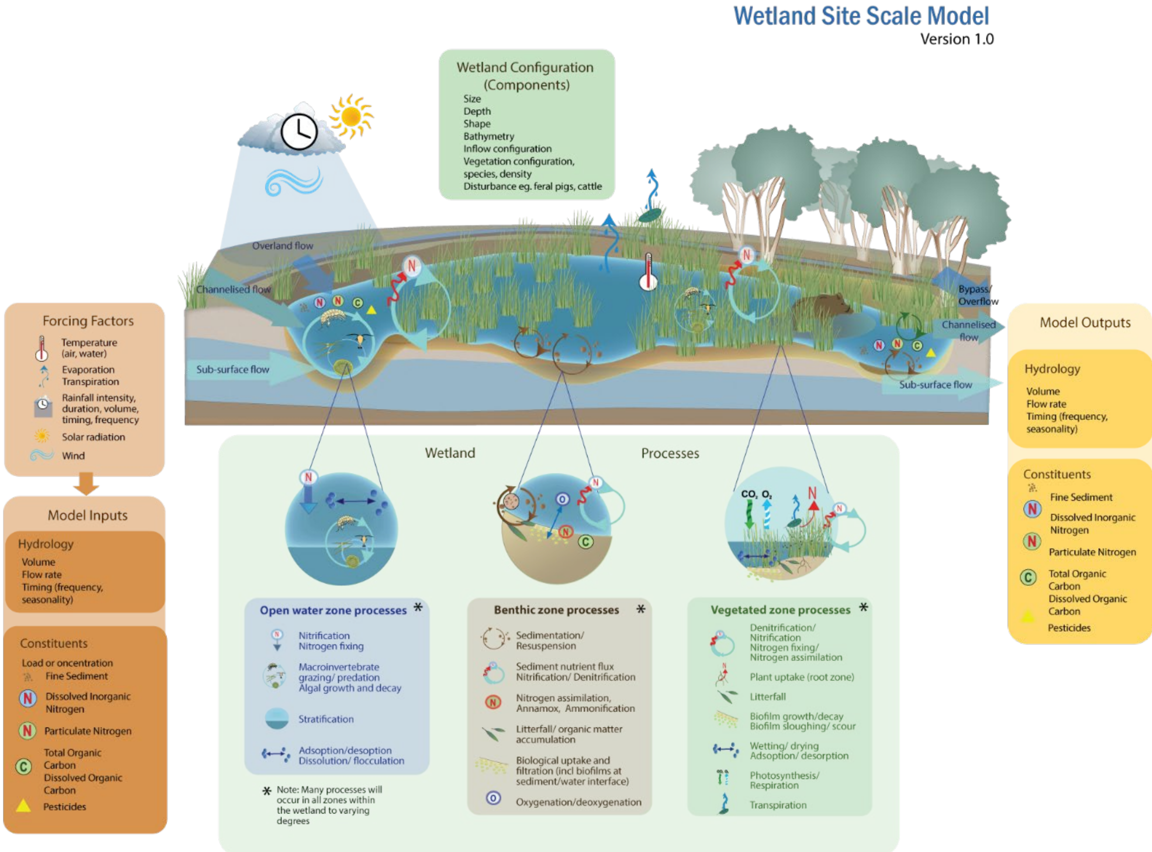

Figure 2. Wetland site scale conceptual model (DESI 2023b).

Figure 3 represents the wetland in terms of its local contributing catchment, and includes a broader range of hydrological and landscape factors to more accurately represent the inputs to the wetland. Forcing factors act upon the landscape to generate runoff, and the runoff in turn generates the model inputs in terms of surface runoff and groundwater. The climate forcing factors also influence the processes that occur in the wetland itself.

Landscape components affect water inflows to wetlands depending on land use (e.g., cropping, vegetation, urban impervious areas), soil type, geology (which influences percolation to groundwater), slope, vegetation cover, and management practices. Runoff flow rates and volumes from the wetland catchment as a result of rainfall are variable due to seasonality (e.g., wet and dry seasons), connectivity (e.g., the degree to which a wetland is connected to waterways), and landscape features, and may exhibit first flush characteristics in terms of pollutant concentrations and loads.

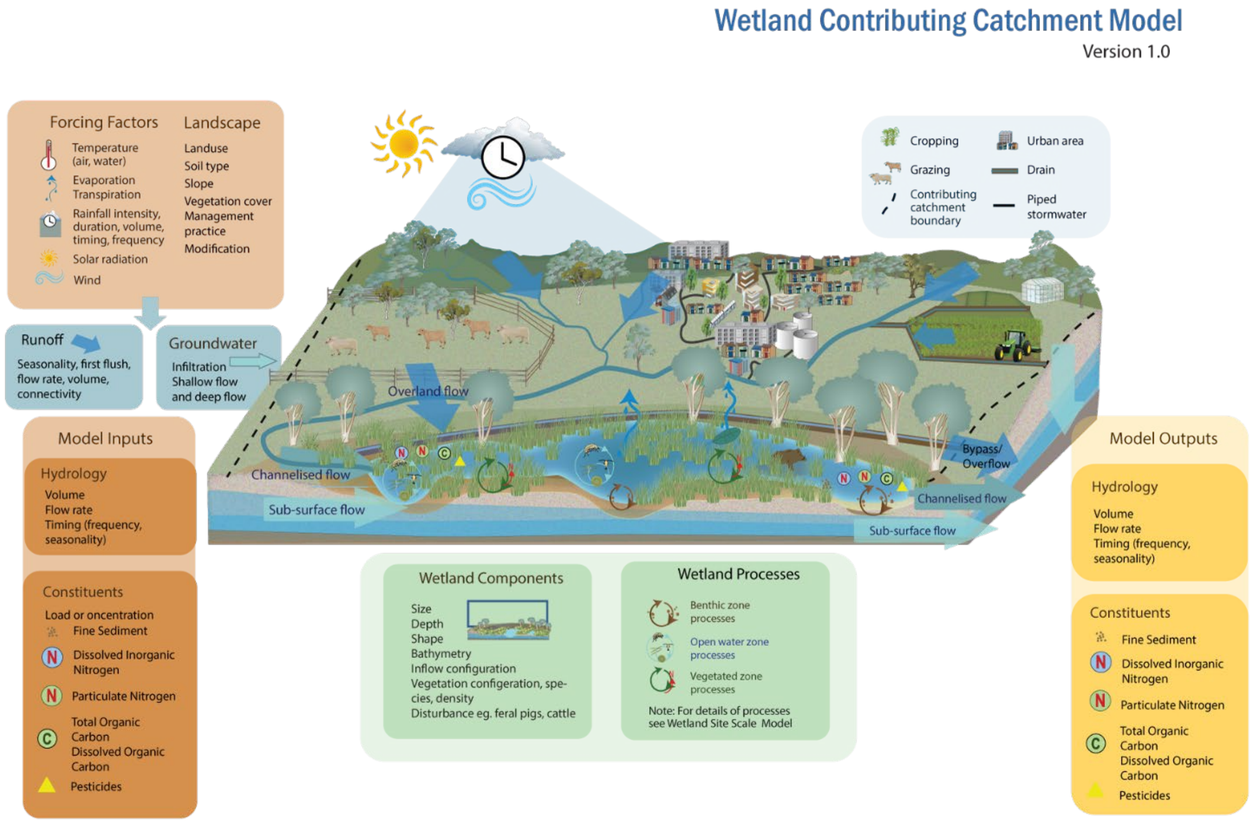

Figure 3. Wetland contributing catchment scale conceptual model (DESI 2023b).

Figure 4 shows wetlands within a sub-catchment or catchment scale and aims to represent multiple wetlands within a catchment or sub-catchment, which can be complex. A catchment or sub-catchment model can incorporate individual site scale wetlands as part of a node and link network often used to describe a catchment. These incorporate the same forcing factors, constituents and processes as the [catchment model](), but may be less detailed as noted above.

These models highlight the importance of scale and the context of the wetland within a catchment or sub-catchment, and also identify the need to evaluate wetland configuration components (size, shape, bathymetry), processes (benthic, macrophyte and water column related), and forcing factors to evaluate their performance.

Taking into account the information presented here, 21 dominant wetland processes were identified (and grouped into three zones (open water, benthic and vegetated). Each process was linked to related components or drivers, and each component was then evaluated to determine their significance (low, moderate or high) in wetland function, especially with respect to changes in water quality. For example, in the open water zone, nitrification/nitrogen fixing was associated with ten components/drivers, four of which were identified as highly significant - temperature, inflow volume, denitrification species and redox. These characteristics are presented in Appendix 2: Wetland processes and related components.

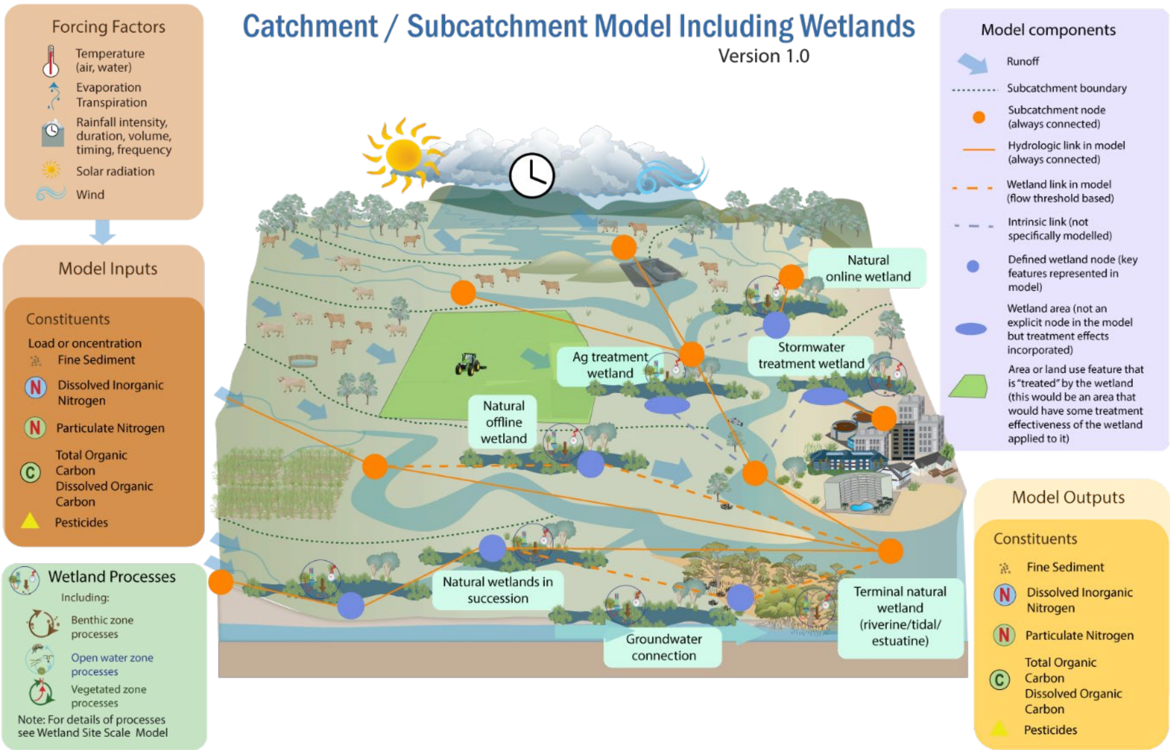

Figure 4. Wetland catchment scale conceptual model (DESI 2023b).

Treatment (or constructed) wetlands are engineered systems that replicate and enhance the physical, biological, and chemical treatment processes occurring in natural wetlands to remove fine sediments, nutrients and other pollutants (e.g., pesticides, heavy metals) (DESI 2022a). In the Queensland Wetland Mapping Method, these are defined as 'Artificial Wetlands'[4] (DESI 2023c). Other names for treatment wetlands are landscape wetlands, embellished wetlands, surface flow wetlands, or free-water wetlands. These wetlands are designed for the specific service of pollutant processing, and each treatment wetland will have different pollutant targets and different processes involved for water quality improvement. Treatment or constructed wetlands should generally not be considered as wetlands for providing habitat services in the landscape (Waltham et al. 2024a).

Table 4 summarises the main treatment systems that are used within agricultural land uses, and the processes associated with them. This information was derived from the Wetland*Info* website (DESI 2022a).[5]

Table 4. Overview of treatment system including the targeted pollutants and main treatment processes involved. Source: DESI (2022a). Further detail is accessible through the hyperlinks in the 'Treatment system' column.

Treatment system	Description	Primary pollutant removed	Main treatment processes involved
Treatment wetland	"Treatment wetlands are engineered systems that replicate and enhance the physical, biological and chemical treatment processes	• Nitrogen • Phosphorus • Fine sediment • Toxicants	• Nitrification and denitrification • Plant absorption/uptake • Microbial degradation

[4] *Artificial wetlands* refer to anthropogenically constructed wetlands where no natural or modified wetlands existed prior to the commencement of construction.

Treatment system	Description	Primary pollutant removed	Main treatment processes involved
	occurring in natural wetlands to remove fine sediments, nutrients and other pollutants (e.g., pesticides, heavy metals)."		• Adsorption • Filtration • Precipitation
Bioreactor	"Denitrifying bioreactors are systems filled with organic matter (e.g., woodchip) designed to remove nitrate through the process of denitrification."	• Nitrogen	• Denitrification
Algae treatment	Wetland algae treatments use harvested freshwater or saline/marine macroalgae cultivated in ponds to remove pollutants. Algal biomass is regularly harvested.	• Nitrogen • Phosphorus	• Plant absorption/uptake
Floating wetlands	"Floating wetlands consist of a suspended matrix planted with wetland plants. This facilitates microbiological and plant processing of nutrients. Floating wetlands work by encouraging settling and biological processing of suspended sediments, particulate and dissolved nutrients and pollutants and also by directing the water through the suspended root mass."	• Nitrogen • Phosphorus • Fine sediment	The potential mechanisms that provide treatment in floating wetlands include: • Denitrification • Plant absorption /uptake • Microbial degradation • Adsorption • Filtration • Sedimentation
Sediment basins	Sediment basins are designed to trap and store sediment and debris through the process of sedimentation.	• Coarse-medium sediment	• Sedimentation
Recycle pits	"A recycle pit is a structure designed to capture irrigation runoff (known as tailwater) for re-use in the production area. Unlike many other treatment systems used in agricultural production systems, recycle pits do not treat the water. Rather they rely on the water being captured and re-used on the farm."	• Nitrogen • Sediment • Toxicant	• Reuse • Sedimentation
Vegetated drains Vegetated buffers and swales	"Vegetated drains are open channels for conveying water, where vegetation covers most of the banks and bed. The difference between drains and swales, is that swales are dry most of the	• Coarse-medium sediment • Nitrogen • Phosphorus	• Sedimentation • Infiltration • Filtration • Plant absorption /uptake • Adsorption

Treatment system	Description	Primary pollutant removed	Main treatment processes involved
	time whereas drains often hold water for extended periods.”		

2.2 Summary of processes influencing wetland function and pollutant processing

In summary, wetlands, both natural and constructed, play a critical role in improving water quality by processing pollutants such as nutrients and in some conditions, sediments, and pesticides. Natural wetlands, including palustrine, lacustrine, and riverine systems vary in their effectiveness depending on their specific hydrological and ecological characteristics. Many of these systems act as sinks for pollutants, such as nitrogen, where processes like denitrification, plant uptake, and sediment storage help in reducing pollution loads. However, the evidence shows that poorly designed and/or managed wetlands can release stored pollutants and become net overall contributors of those pollutants instead of solutions. Key processes like algal growth, plant uptake, sedimentation, and nitrogen cycling, along with wetland configuration factors such as depth, surface area, and vegetation type, determine how efficiently wetlands function in pollutant removal. Comprehensive resources including conceptual models developed by DESI document and help illustrate these processes.

Constructed wetlands or treatment wetlands, are engineered systems designed to replicate and enhance natural wetland processes for the specific purpose of pollutant removal. These systems include various types such as surface flow wetlands, bioreactors, algae treatment ponds, and floating wetlands, each with distinct mechanisms for filtering and processing pollutants. For example, bioreactors use organic materials to promote denitrification, while floating wetlands use plants suspended in a matrix to filter nutrients and sediments from the water. These treatment systems are particularly effective in agricultural landscapes for removing nitrogen, phosphorus, and sediment, but their design and operation must be tailored to the specific pollutants they are targeting. The Wetland*Info* website provides extensive resources on these systems, highlighting the importance of selecting the right wetland type and understanding the underlying processes for effective water quality improvement.

3. Evidence of the effectiveness of wetland systems in pollutant processing

Management actions to improve water quality in the GBR catchments are typically organised by land use, and this is also reflected in the design of investment programs. Accordingly, this review distinguishes the water quality improvement efficiency of wetlands in agricultural areas from those intercepting urban runoff and wastewater. Summaries of the findings are presented at the end of each relevant section in Tables 9, 10 and 12, respectively.

3.1 Agricultural areas

3.1.1 Global studies

Interest in the role of wetlands and their ability to provide a water quality improvement service in agricultural areas is of considerable interest globally. The 2022 SCS Question 4.7 (Waltham et al. 2024) initially identified over 2,000 studies published globally between 1990 and 2023 that addressed the water quality improvement efficiency of natural, restored and constructed/treatment wetlands in tropical and subtropical regions. If temperate studies had also been considered, the number of papers would have been significantly greater. Following a screening process to ensure that the publications were relevant to the GBR context, 238 studies were selected for inclusion in the 2022 SCS synthesis (see 2022 SCS Question 4.7 for search methods). Significant variability was found in the type of data collected and how it was reported. In many cases, extracting critical details was challenging – either because the details were not collected during the research, not provided in the publication, or the details and sampling procedure had not been adequately designed during experimental planning. This highlights the importance of comprehensive reporting in these types of experimental and research projects to facilitate broader assessment and comparisons.

3.1.2 GBR studies

In the GBR catchment, 73% of the catchment area is used for grazing, 1.2% for sugarcane, 2.8% for irrigated and dryland cropping and 0.2% for horticulture crops. It is estimated that agricultural land uses within the GBR catchment contribute approximately 75% of the total loads of fine sediments, with dominant contributions from the Burdekin and Fitzroy regions (Prosser and Wilkinson 2024a), around 65% of the total dissolved inorganic nitrogen (DIN) load (Prosser and Wilkinson 2024b) and a majority of the PSII herbicides transported to the GBR (Templeman and McDonald 2024; Waters et al. 2014). It is also estimated that streambank erosion accounts for approximately 25% of the fine sediment loads delivered to the GBR (Prosser and Wilkinson 2024a).

Wetlands are common in agricultural areas in the GBR catchment and can be natural, restored, or constructed/treatment systems. Natural wetland types considered in this review are riverine, palustrine and lacustrine. While they remain present in agricultural areas, their extent has notably decreased, particularly riverine wetlands and those located in intensive agricultural zones. Treatment/constructed wetlands in agricultural landscapes include floating wetlands, vegetated drains, recycle pits, swales, buffer strips, sediment basins and bioreactors (DESI 2022a). Wetland systems in these areas are highly diverse, complex, and vary widely in their capacity to process, remove, and store pollutants. A detailed understanding of wetland function, the processes that underpin water quality improvement, and the conditions that optimise these processes is essential for appropriately designing, managing, and protecting wetlands to maximise their water quality improvement efficiency (refer to Section 4 and 5).

In the **evidence from the GBR catchment**, seventeen studies investigated 61 wetlands, comprised of 33 constructed/treatment wetlands, including irrigation ponds, recycle pits and vegetated drains; 18 natural wetlands; and 10 bioreactors, including membrane and woodchip denitrifying bioreactors (Table 5). A list of studies is presented in Appendix 1: Great Barrier Reef studies included in the review.

Table 5. The type and number of wetlands investigated for their water quality improvement efficiency within the GBR catchment (1990–2023).

Wetland Type		Count of wetlands
Constructed/ Treatment Wetland	Embellished wetlands	3
	Landscape wetland	1
	Biodiversity pond	1
	Constructed/Treatment wetland *(nondescript)*	8
	Irrigation pond	1
	Macrophyte zone	1
	Recycle pit *(modelled)*	10
	Riparian buffer	4
	Sediment basin	1
	Vegetated drain	2
	Off-channel facility	1
Natural	Freshwater marsh	1
	Wetland *(modelled)*	8
	Riverine	2
	Melaleuca spp. (*M. quinquenervia*)	1
	Melaleuca spp. (*M. viridiflora*)	1
	Ephemeral (*Melaleuca*–Eucalyptus spp.)	1
	Coastal lagoon with water lilies (*Nymphaea* spp.)	1

Wetland Type		Count of wetlands
	Coastal lagoon with emergent grasses	1
	Aggregation of floodplain wetlands	1
	Aggregation of palustrine, lacustrine, and riverine wetlands	1
Bioreactors	Bioreactor *(nondescript)*	8
	Membrane bioreactor	1
	Woodchip denitrifying bioreactor beds - mixed hardwood	1
		61

A data extraction spreadsheet was created to provide more detail of the GBR studies than the assessment undertaken for Question 4.7 of the 2022 SCS. The evidence included the GBR references used in Question 4.7 and additional references that were not captured in the search process (Adame et al. 2019a; Adame et al. 2019b; Adame et al. 2021a; Adame et al. 2021b; Kavehei et al. 2021a; Kavehei et al. 2021b; Manca et al. 2021; McJannet et al. 2012; McKergow et al. 2004; Navaratna et al. 2012; Rafiei et al. 2022; Wallace et al. 2022; Wallace and Waltham 2021), one reference that was published after the cut-off date for evidence in the 2022 SCS (Cheesman et al. 2023) and additional reports that were not initially included in the 2022 SCS due to the understanding of their peer review status (Alluvium 2016; Canning et al. 2021a; Wallace et al. 2020; Wegscheidl et al. 2021).

The data extraction spreadsheet was very comprehensive and included 113 fields. The objective was to extract all the information needed to model wetland efficiency in removing nutrients, sediments and pesticides from the GBR literature. There were nine main categories: Wetland Configuration (components), Wetland Hydrology and Hydraulics, Wetland Efficiency, Wetland Processes, Forcing Factors, Landscape Characteristics, Wetland Management, Costs, Monitoring information. The factors cited in the literature are described in Section 4. In all instances, the data from the 17 studies was not sufficient to complete all the columns. Numerous knowledge gaps are very apparent, necessitating further exploration to elucidate the factors influencing the efficiency of wetlands within the GBR. This is discussed further in Section 4.

3.1.3 Natural/near-natural and restored wetlands

Global studies

In natural wetlands it can be challenging to identify defined inlet and outlet points due to complex landscape variability and hydrology. As a result, fewer studies have quantified nutrients, sediments, and pesticides removal in natural and restored wetlands compared to engineered treatment/constructed wetlands which are often specifically designed with inflows and outflows. From the available literature, most studies on natural/near-natural wetlands have been conducted in the United States (US) (n = 30) and China (n = 10), followed by Australia (n = 6). Research topics range from modelling the water quality benefits of redirecting polluted waters through natural wetlands (Lane et al. 2003), examining spatial and temporal variability in river-floodplain interactions (Primost et al. 2022), monitoring water quality improvement efficiency and nitrous oxide generation in seepage wetlands (Zaman et al. 2008), to re-routing agricultural drainage water through a forested wetland (Lindau et al. 1997).

An overview of the efficiency rates (reductions in concentrations) for studies assessing natural, near-natural and restored wetlands are presented in Table 6. The 2017 SCS concluded that 'globally wetlands have been found to remove N at a median rate of 93 g m^2 yr^{-1} and P at a rate of 1.2 g m^2 yr^{-1} with a removal efficiency of 39% and 46% respectively' (Eberhard et al. 2017). These efficiencies align well with the Total Nitrogen (TN) and Total Phosphorus (TP) removal rates observed in near-natural and restored wetlands reported in this review.

Table 6. Overview of the efficiency (% reductions in concentrations) reported by studies included in the global review. All data presented are the average removal efficiencies (%) (number of studies, minimum and maximum efficiencies) of all reported water quality variables for natural, near-natural and restored wetlands.

Parameter	Natural wetlands	Near-natural wetlands	Restored wetlands
Total nitrogen	63% (5, 27 to 96)	34% (6, 12 to 83)	38% (1, 38)
Ammonium	80% (2, 73 to 86)	64% (1, 64)	48% (2, 48)
Nitrate	78% (2, 76 to 80)	61% (3, 6 to 97)	49% (3, 26 to78)
Total suspended solids	45% (2, -1 to 91)	n/a	35% (2, -4 to 73.8)
Total phosphorus	75% (3, 59 to 98)	54% (6, 6 to 93)	52% (2, 26 to 59)

Studies on restored wetlands are also predominantly from the US (n = 11) and Australia (n = 3). Common restoration approaches examined include hydrological restoration (n = 3), vegetation planting (n = 3), and weed removal (n = 2; e.g., Bruland et al. 2003). Research areas varied from regenerative stormwater conveyance (Thompson et al. 2018), ditch-filling and planting (Bruland et al. 2003), modelling the water quality impacts of 8,000 km^2 of wetland restoration (Evenson et al. 2021), hydrological restoration (Kahara et al. 2022) and adding carbon to enhance nitrate removal (Yang et al. 2019).

One study provided annual loading and removal rates of TN, reporting a 38% removal efficiency. Two studies reported removal efficiencies for TP of 26% and 52%. For TSS, restored wetlands showed removal efficiencies ranging from -4% to 78% (n = 2). Although based on limited evidence, restored wetlands were most effective at removing TP (up to 52%) and least effective at removing TSS (35% on average).

GBR studies

Four studies reviewed the effectiveness of natural wetlands in pollutant processing in the GBR. Direct measurement of nutrient inflows and outflows (and nitrogen specifically) is often not feasible due to the difficulty measuring all inputs and outputs accurately. Instead, nitrogen fate and removal are often inferred by measuring the underlying processes that control nitrogen transformation and/or through modelling.

Adame et al. (2021a) developed a conceptual framework to visualise and synthesise nitrogen fate and transport from catchments to the GBR. This framework was developed through a comprehensive review of published information and is expanded in the resources found on Wetland*Info* (DESI 2021). In addition, eight workshops were held involving scientists, program managers, policymakers, and extension staff working on the GBR and south-east Queensland between 2016 and 2019, to support the framework and corresponding model development. That study concluded that denitrification is a key mechanism for reducing nitrogen in wetlands, particularly in those that are highly connected, where soil/sediment had higher concentrations of carbon, and those that receive consistently high concentrations of nitrate.

Accordingly, further studies in the GBR measured denitrification rates within wetlands, but these studies were not designed to measure or assess the complete hydrologic balance which would have enabled the determination of the overall efficacy of the wetland with respect to the overall water balance. Adame et al. (2019a) found potential denitrification rates in melaleuca floodplain wetlands ranging from 1.1 to 9.7 mg N m^2 h^{-1}, (average rate 5.0 mg N m^2 h^{-1}), influenced primarily by the input nitrate concentrations, with denitrification rates higher where there were higher inputs. Adame et al. (2021b) reported denitrification rates in the sediment of a floodplain wetland with varying vegetation cover - native aquatic grass, native waterlilies and open water, and also denitrification rates of the epiphyton on macrophytes. Denitrification in the sediment varied from 3.3–51.5 mg m^2 h^{-1}, which was higher than the rates found in the epiphyton (1.9–3.7 mg m^2 h^{-1}). Higher denitrification rates occurred under vegetation (e.g., grasses, waterlilies) compared to open-water sediments.

Canning et al. (2021a) estimated denitrification rates in a palustrine wetland in the Dry Tropics using two approaches (Adame et al. 2019b; Land et al. 2016). Land et al. (2016) conducted a global meta-analysis of 93 studies and 203 wetlands (58 wetlands in equatorial locations) and demonstrated that wetlands can remove N at a median rate of 93 g m^2 yr^{-1} and P at a rate of 1.2 g m^2 yr^{-1} (39% and 46% removal efficiency, respectively). This was estimated by developing a denitrification model based on factors like hydraulic loading rate, temperature, wetland size, and inflowing N concentration. In contrast, Adame et al. (2019b) focused on predicting potential denitrification rates for eight coastal wetlands in North Queensland using inflow nitrate-nitrogen concentrations, with a simpler regression model. The predicted mean annual actual denitrification rates were 15 t N yr^{-1} and 12.6 t N yr^{-1}, corresponding to long-term removal efficiencies of 80% and 62%, using respectively the Adame et al (2019b) and Land et al. (2016) approaches.

Two studies used modelling to estimate N removal in GBR floodplain wetlands. Adame et al. (2019b) modelled nitrate removal by a floodplain wetland system (2,213 ha) in a sub-catchment of the Tully and Murray basins during flooding events. Model simulations indicated that the wetland could remove up to 70% of the incoming nitrate load within the first day of flooding. Similarly, Rafiei et al. (2022) applied a modified Soil and Water Assessment Tool model to quantify nitrate exchange between wetland, groundwater, and surface water for 28 wetlands (660 ha in total) in the North Johnstone River catchment, showing that these wetlands could reduce annual nitrate discharge by 13.5%. The potential ecological impact of the nitrate inputs on the wetlands was not assessed.

McJannet et al. (2012) measured the nutrient balance of a tropical riverine wetland over three years in the Tully-Murray floodplain. A small proportion of N (-4% removal efficacy for TN) was exported, sediments were neither imported or exported (-1% removal efficiency), and 14% of TP was removed. The low overall removal efficacy was attributed to short water residence times (less than six hours, 90% of the time).

3.1.4 Constructed/treatment wetlands

Global studies

Research on treatment/constructed wetlands is most extensive in the US (n = 67) followed by China (n = 30). Among these systems, the use of vegetation (n = 22), buffer strips (n = 20) and non-specific constructed wetlands (n = 15) were studied most often. Some treatment wetlands also included chemical additions to enhance pollutant removal (Ann et al. 1999; Bachand et al. 2019) and the use of on-farm irrigation tanks (Shao et al. 2013). Studies of treatment/constructed wetlands span a wide array of topics including modelling the effectiveness of treatment wetlands in reducing runoff from agricultural hillsides (Zhang et al. 2020), evaluating subsurface horizontal flow systems (De Ceballos et al. 2001), assessing constructed tidal marshes (Etheridge et al. 2015), examining riparian buffer strips and drainage ditches (Iseyemi et al. 2016; Schoonover et al. 2010) and identifying optimal sampling strategies for constructed wetlands (Moustafa and Havens 2001). Seven studies assessed floating treatment wetlands (FTWs, e.g., Chance et al. 2020; Shahid et al. 2018; Wang et al. 2022; Yamasaki et al. 2022). Three of these provided removal efficiencies for nutrients and pesticides (Pavlidis et al. 2022a; Pavlidis et al. 2022b; Rigotti et al. 2021).

The average annual loading and removal rates of the studies that assessed treatment/constructed wetlands are presented in Table 7. Overall, reported removal efficiencies of all of these pollutants tend to be lower than those documented in natural and near-natural wetlands, however there are significantly fewer studies on natural and near-natural wetlands.

Table 7. Overview of the efficiency (% reductions in concentrations) reported by studies included in the global review. All data presented are the average removal efficiencies (%) (number of studies, minimum and maximum) of all reported water quality variables for treatment wetlands.

Parameter	Treatment wetland
Total nitrogen	46% (40, -4 to 97)
Ammonium	65% (11, -14 to 99)
Nitrate	43% (23, -22 to 99)

Parameter	Treatment wetland
Dissolved inorganic nitrogen	44% (5, 7 to 61)
Total suspended solids	57% (10, 1 to 94)
Total phosphorus	49% (38, 2 to 97)
Phosphate	38% (5, -15 to 60)
Pesticides	69% (16, 4 to 100)

GBR studies

Seven studies examined constructed/treatment wetlands in the GBR region. Two focused on N removal.

Kavehei et al. (2021a) evaluated N removal in eight constructed wetlands, five in the Wet Tropics and three in the Dry Tropics and Mackay Whitsundays regions as well as two vegetated drains in the Wet Tropics. DIN reductions in the constructed wetlands ranged from -15 to 90% (average 33%), total dissolved nitrogen (TDN) from-5 to 50% and total nitrogen (TN) from -5 to 80%. DIN removal from the vegetated drains was between 50 and 80%. The study found that although there was variability, the most effective wetlands had relatively high inflow concentrations, extensive vegetation cover, high length to width ratios and low suspended solids concentrations.

In another study, Kavehei et al. (2021b) quantified nitrogen removal rates in four constructed treatment wetlands in the Tully and Johnstone catchments of the Wet Tropics. Testing different inflow rates, vegetation conditions, and nutrient concentrations, nitrate removal was found to vary between -30 and 100% and ammonia removal between -90 and 50% (reported in Table 10). The highest N removal rates occurred at higher NO_3^- inflow concentrations and slower water flows. The results highlight the importance of the design of treatment wetlands and management for improved nutrient removal. The study also highlighted that water pH, electrical conductivity, soil carbon content, oxidation-reduction potential and vegetation cover, may serve as useful indicators for assessing treatment wetland effectiveness in N removal.

Five studies modelled the efficiency of constructed wetlands in the GBR. Wallace et al. (2020) modelled a treatment wetland near Babinda (Russell catchment, Wet Tropics), an area dominated by sugarcane and found mean total potential denitrification rates in the water column of 8.1 mg m^2 h^{-1}. Simulations suggested a 100% TN reduction after 26 days, with gaseous denitrification accounting for 42% of the N loss, followed by sedimentation at 21% within the first 12 days, and drainage contributing 37% until N in the wetland was depleted.

Wallace and Waltham (2021) combined a hydrological water balance model with a simple first order rate-based denitrification model to estimate the long-term N and sediment filtering capacity of the same treatment wetland near Babinda. The results showed a reduction of 86% TSS and ~52% of TN over a year.

Wallace et al. (2022) modelled TN and TSS removal in a system of four constructed wetlands in the Mackay Whitsunday region, finding a TN removal efficiency of ~37%. Low water residence time was identified as a key factor influencing N removal, with most of the water passing through the wetland within three days.

Alluvium (2016) modelled DIN reduction in recycle pits and wetlands across irrigated sugarcane areas in the Wet Tropics, Burdekin, and Burnett Mary regions under different weather conditions. For wetlands operating in both wet and dry conditions, modelled efficiencies ranged from 1 to 89% depending on the proportion of the wetland area in the catchment and the location of the system. The most effective ratios were 5 to 10% wetland area in dry tropical climates, and 10 to 20% wetland area in wetter tropical regions.

McKergow et al. (2004) measured the performance of grass and rainforest riparian buffers at four sites in the Johnstone catchment (Wet Tropics). TN removal ranged from -43 to 45%, TP from -33 to 64% and TSS from 8 to 46%. Grass buffers outperformed riparian buffers, likely due to higher vegetation. The study was conducted under extreme conditions: steeply cultivated land with high rainfall intensities, which contributed to the considerable variability observed in the removal efficiencies.

3.1.5 Bioreactors

Global studies

Six studies examined bioreactors in tropical environments (Table 8). The maximum bioreactor removal efficiencies were only reported in four studies, and the maximum rates for TN were 80%, 98% for NO_3^-, 68% for NH_4^+, 55% for TP and 66.2% for the pesticide ametryn (David et al. 2015; Du et al. 2016; Hunt et al. 2008; Navaratna et al. 2012).

Table 8. Types of bioreactors featured within the body of evidence and the number of bioreactor studies conducted according to country.

Bioreactor	Australia	China	United States
Aerated and non-aerated biofilm reactors		1	
Column bioreactor			1
Eco-soil reactor		1	
Membrane bioreactor	1		
Tile woodchip bioreactors	1		1
Total	**2**	**2**	**2**

GBR studies

Within the GBR, four studies assessed bioreactors for pollutant removal in agricultural catchments. Manca et al. (2021) evaluated four bioreactors on sugarcane farms in the Johnstone and Tully catchments (Wet Tropics), and Haughton and Lower Burdekin catchments (Dry Tropics). These systems removed NO_3^- at rates of 0.15 to 7.1 g N m^3 d^{-1}. Beds performed better than wall designs. For both designs, higher NO_3^- concentrations were associated with higher N removal performance. Additionally lower flow rates and longer residence times were associated with more effective bioreactor performance.

Cheesman et al. (2023) assessed woodchip denitrifying bioreactor beds on a sugarcane farm in the Russell catchment (Wet Tropics) and found a 41% reduction of nitrate in intercepted waters. However, removal rates were limited by NO_3^- availability, for example, a load reduction over the 2018/19 season was just 0.11 kg N ha^{-1} yr^{-1} (approximately 0.00003 g N m^3 d^{-1}), significantly lower than the rates reported by Manca et al. (2021) and Wegscheidl et al. (2021). The limited performance was attributed to the 'first flush', where a high proportion of the annual nitrogen load bypasses the system during the first rainfall event of the wet season (refer to the nutrient transport and delivery characteristics for the GBR catchments in Question 4.5; Burford et al. (2024)). Consequently, these bioreactors remain N-limited for most of the year, emphasising the importance of considering annual performance when evaluating bioreactor efficacy.

Wegscheidl et al. (2021) analysed denitrifying bioreactors on farms in three case studies in the Lower Burdekin, Johnstone and Russell catchments. One case (Case 3) consisted of an in-line drain bioreactor bed below the floor of a pre-existing drain and showed an 84.3% reduction in NO_3^-. Another case (Case 6) consisting of six parallel bioreactor beds showed a nitrate removal rate between 9.4 and 13.1 g N m^3 d^{-1}. Finally, a case consisting of twin in-line bioreactor beds (below floor/invert of drain) (Case 7) showed a 41% reduction in TN.

Navaratna et al. (2012) tested a different type of bioreactor combined within a treatment system. A membrane bioreactor (MBR) and a hybrid treatment system consisting of a MBR, ultraviolet (UV) disinfection unit and a granular activated carbon column, were assessed in a laboratory setting. While the MBR alone removed 40% of ametryn, the hybrid system removed ametryn to below detection levels, both with a hydraulic retention time of 7.8 hours. These results highlight the potential of combined treatment approaches for pesticide removal.

Comparison among wetland and treatment systems

Table 9 (global studies) and Table 10 (GBR studies) present average efficiency values for nutrient and TSS removal across the wetland types included in this review. The minimum and maximum values are shown in parentheses. While these averages do not capture the full range of performance (as wetlands can operate above or below the mean), they provide a useful indication of relative differences between wetland types and pollutant parameters. It is important to note that these reported rates reflect 'on-site' performance and may not represent overall catchment-level loads once delivery ratios are considered. Consequently, they may only represent a small fraction of the total load exported downstream from catchment areas (delivered to end of catchment). This distinction is critical when assessing overall effectiveness in pollutant processing.

Due to the limited data on pesticide removal and the wide range of pesticides studied, results have been aggregated. It is recognised that pesticides behave in different ways depending on their properties and environmental conditions, but this level of detail was not typically presented in the studies.

The global evidence suggests that natural wetlands generally display higher nutrient removal rates, while treatment wetlands tend to show lower rates. However, the data is limited (less than five studies in some cases), particularly for natural and near-natural wetlands. Pesticide removal was generally high in a range of systems, with most data derived from treatment systems. Sediment removal was most efficient in treatment wetlands but can also accumulate in wetland systems. These findings align with other literature, such as Forbes et al. (2012) which reported similar trends (again acknowledging data limitations). Wang et al. (2019b) suggest that these differences are related to factors such as hydraulic loading and the size and configuration of treatment wetlands such as vegetated drains.

A more detailed discussion about the factors affecting pollutant processing in different systems is provided in Section 4.

*Table 9. Overview of the efficiency (% reductions in concentrations) reported by studies included in the **global review**. All data presented are the average removal efficiencies (%) (number of studies, minimum and maximum) of all reported water quality variables. For sites/studies where multiple results or a range are presented, the average value has been extracted for this analysis, suggesting that efficacy could be higher or lower depending on how the data were generated and presented in published studies. TN = Total Nitrogen, DIN = Dissolved Inorganic Nitrogen, NO_3^- = Nitrate, NH_4^+ = Ammonium, TP = Total Phosphorus, PO_4^{3-} = Phosphate, TSS = Total Suspended Sediment. Shaded cells indicate average values derived from less than five studies and therefore low confidence in the averages provided. Empty cells are where no data could be found. Note that the efficacy reported in Cheesman et al. (2023) and Wegscheidl et al. (2021b) were not included in the averages (extracted from Waltham et al. 2024b), due to the date of publication or peer-review status.*

Wetland type	TN	DIN	NO_3^-	NH_4^+	TP	PO_4^{3-}	TSS	Pesticide
Natural wetland	63.5 (5, 27–96.4)		78.0 (2, 76–80)	79.5 (2, 73–86)	74.5 (3, 59–97.6)		45.0 (2, -1–91)	98.5 (2, 97–100)
Near-natural	33.5 (6, 11.6–83)		60.8 (3, 6–96.5)	64.0 (1, 64)	54.6 (6, 6–93)			
Restored wetland	38.0 (1, 38)		48.9 (3, 25.7–77.9)	48.2 (2, 48–48.3)	52.4 (2, 25.7–59)		34.9 (2, -4–73.8)	
Treatment wetland	46.4 (40, -4–97)	44.2 (5, 6.6-60.5)	42.6 (23, -22–99)	64.6 (11, -14–99)	49.3 (38, 1.8–96.5)	38.0 (5, -15.1–59.5)	57.1 (10, 1.1–94)	69.2 (16, 3.6–100)
Bioreactor system	80.0 (1, 80)		82.2 (1, 82.2)					47.0 (2, 14.3–100)
Combination	19.0 (1, 19)		93.0 (1, 93)		45.0 (4, 24–94)		50.0 (1, 50)	

*Table 10. Range of pollutant removal efficiencies from observational and modelling studies of **GBR wetlands** receiving agricultural runoff. TN = Total Nitrogen; TDN = Total Dissolved Nitrogen; DIN = Dissolved Inorganic Nitrogen; NO_3^- = Nitrate; NH_4^+ = Ammonium, NO_x = Nitrogen Oxides; TP = Total Phosphorus; TSS = Total Suspended Sediment. Blank cells contain no data.*

Wetland type (number of sites), study type	Location	TN	TDN	DIN	NO_3^-	NH_4^+	NO_x	TP	TSS	Pesticide (Ametryn)	Reference
Constructed/treatment wetlands (8) Monitoring	Wet Tropics, Dry Tropics, Mackay Whitsunday	-5 to 80%	-5 to 50%	-15 to 90%*							Kavehei et al. (2021a)
Constructed/treatment wetlands (*Vegetated drains)* (2) Monitoring	Wet Tropics			50 to 80%							Kavehei et al. (2021a)
Constructed/treatment wetlands (4) Monitoring	Wet Tropics				-30 to 100%	-90 to 50%					Kavehei et al. (2021b)
Constructed/treatment wetlands (4) Modelling	Mackay Whitsunday	36.7%									Wallace et al. (2022)
Constructed/treatment wetlands (1) Modelling	Wet Tropics	52.2%							86%		Wallace & Waltham (2021)
Natural wetland (Riverine) (1) Monitoring	Wet Tropics	-4%						14%	-1%		McJannet et al. (2012)
Constructed/treatment wetlands *(Riparian buffer) (4)* Monitoring	Wet Tropics	-43 to 45%						33 to 64%	8 to 46%		McKergow et al. (2004)
Natural wetland (1 aggregation of palustrine, lacustrine, and riverine wetlands) Modelling	Wet Tropics				70%						Adame et al. (2019)
Natural wetlands Modelling	N/A - Laboratory Scale				13.5%						Rafiei et al. (2022)

Wetland type (number of sites), study type	Location	TN	TDN	DIN	NO_3^-	NH_4^+	NO_x	TP	TSS	Pesticide (Ametryn)	Reference
Bioreactors (5) (Woodchip) Monitoring	Wet Tropics, Dry Tropics				0.15 and 7.1 g N m^3 d^{-1} (no percentage was provided)						Manca et al. (2021)
Bioreactors (Woodchip) (1)) Monitoring	Wet Tropics		48%				41%				Cheesman et al. (2023)
Bioreactor (Membrane) Laboratory	N/A - Laboratory Scale									40%	Navaratna et al. (2012)
Bioreactor (Membrane + UV disinfection unit + granular activated carbon column) Laboratory	N/A - Laboratory Scale									Below detection levels	Navaratna et al. (2012)
Recycle pit (5) Modelling	Burdekin (4) Burnett Mary (1)			18 to 89% 1.1% (Dry weather)							Alluvium (2016)
Constructed wetlands (8) Modelling	Burdekin (4) Wet Tropics (4)			8 to 80% 1 to 49%							Alluvium (2016)
Bioreactors *(woodchip)*: Case #3 (1) Case #6 (1) Case #7 (1) Monitoring	 Dry Tropics Wet Tropics Wet Tropics	 41%			 84.3%						Wegscheidl et al. (2021)

* The large range of DIN reductions were associated with different constructed wetlands, with higher reductions occurring in constructed wetlands receiving high DIN concentrations and those with extensive vegetation cover.

3.2 Non-agricultural areas

In non-agricultural areas, wetland systems are predominantly constructed vegetated systems with engineered inflow and outflow controls. These often incorporate pre-treatment measures such as sediment basins, as well as extensive areas of emergent and floating macrophytes. In recent years, floating wetland systems have emerged as a novel approach to processing nutrients passing through treatment wetlands with recent studies focusing on their performance. As there are no studies examining constructed/treatment wetlands, swales or biofilters in non-agricultural settings in the GBR, the evidence presented below comes from other Australian and international studies.

Constructed/treatment wetlands

No studies were identified that specifically review the effectiveness of constructed/treatment wetlands in pollutant processing from urban areas in the GBR. However, 25 studies with similar climates or environmental conditions to non-agricultural areas in the GBR were included.

Awad et al. (2022) investigated constructed floating wetlands planted with different vegetation. While these systems may not be suitable for mitigating peak flows, they accumulated between 0.48 and 2.0 g of TN per m^2 and 0.04 to 0.46 g of TP per m^2 with the sedge *Baumea rubiginosa*, and between 0.2 and 2.3 g of TN per m^2 and 0.02 to 0.2 g of TP per m^2 with the reed *Phragmites australis*. Under low nutrient conditions (TN ≤ 0.4 mg L^{-1} and TP ≤ 0.2 mg L^{-1}), *Phragmites* growth was not supported, whereas *Baumea* species did grow.

A study by Ryder and Fares (2008) in a Hawaiian watershed demonstrated the performance of three natural/constructed wetlands, with TSS removal of 74 to 85%; TP of -81 to -256%; TN of -47 to 22%, NH_4^+ of -53 to 23% and NO_3^- of -4 to 6%. These results indicate that wetlands can both export and/or process and retain nutrients depending on their design and size of the contributing catchment. Similarly, Phillips et al. (2021) reported that urban wetlands can remove up to 100% of pesticides and 95% of sediment loads in vegetated drains, while other studies (Jordan et al. 2003) observed conditions where wetlands exported nutrients (e.g., -11% for TP, -8.4% for TN, and -4% for TSS in restored wetlands).

Schwammberger et al. (2017; 2019; 2023) noted that nutrient removal in constructed floating wetlands was low when influent stormwater concentrations were low. However, when these systems were applied in catchments with higher input nutrient concentrations typical of urban catchments in Australia, TN removal was 17% and TP removal was 52%. This highlights the importance of influent pollutant loads in the performance of floating wetland systems. These studies also showed that constructed floating wetlands removed large amounts of nutrients from urban stormwater through plant uptake. Since only total nutrients were measured and not dissolved forms, it remains unclear whether the wetlands were processing bioavailable or dissolved nutrient forms.

A recent study by Szota et al. (2024) examined 17 “free flow” stormwater wetlands (0.15 to 18 ha , constructed between 1999 and 2011) in Melbourne and found a median TN removal of 41% (ranging from -36 to 70%). Wetlands with greater than 80% vegetation achieved higher removal rates. TN removal was largely driven by NO_x removal, effective even in wetlands with low vegetation cover, demonstrating the importance of denitrification. In contrast, TSS and TP removal was poor (median removal efficiency of 11% and 17%, respectively).

Riparian wetlands in an urban area in Calamvale, Brisbane, which were remnants of a larger system of natural channels, were highly effective in removing nitrate and phosphate compared to constructed systems in the same catchment. The 600 m of remnant channel, lagoons and associated vegetation were reported by Greenway (2007) and Greenway et al. (2002) as being effective in reducing nutrients in both wet and dry weather conditions. These wetlands were part of a treatment train including sediment basins, ponds and gross pollutant traps, some of which showed elevated nutrient concentrations at the outlets compared to the inlets, suggesting that at times, export of nutrients from the treatment train did occur.

In Florida, US, Griffiths and Mitsch (2017) evaluated an urban tropical stormwater wetland and found it to be a net sink of nutrients. Nitrate concentrations reduced from 0.13 mg L^{-1} at the inlet to less than

0.002 mg L^{-1} at the outlet. N removal was expected to increase over time as denitrification increased with increased organic carbon in the soils, while TP removal was expected to reduce.

Headley et al. (2001; 2005) demonstrated that in reed beds treating elevated nutrient concentrations, the removal of nitrate-nitrogen occurs predominantly through plant uptake and denitrification. The TN concentration in the influent of the 'Low' and 'High' nutrient loadings typically ranged between 9 and 15 mg L^{-1}, predominantly in the form of oxidised nitrogen (NOx-N). In both treatments, virtually complete removal of TN (effluent concentration <0.5 mg L^{-1}) was achieved when the influent loading rates were kept below 0.75 g m^2 d^{-1}. This corresponds to a hydraulic retention time of 2.7 days at an inlet TN concentration of 10 mg L^{-1}. They also observed that denitrification was low initially, from 17 to 22% of the removal of TN, increasing to greater than 49% after 17 months. This reinforces evidence from elsewhere that denitrification capacity develops over time as the biological communities are more established.

Variable performance was noted in a secondary review study by Mitchell et al. (1995) finding that wetland performance under varying hydrologic conditions was inconsistent. Similarly, Nguyen et al. (2018) noted only minor TN and TP reductions when modelling wetlands along the Torrens River in Adelaide, South Australia, but improved outcomes when wetlands were combined with other interventions such as buffer strips and streambank restoration into a treatment train. Bourgues and Hart (2007) found that epiphytic biofilms sheltered bacterial populations able to potentially carry out denitrification at comparable rates to those measured in nearby sediments, emphasising the role of plants in wetlands designed to treat nutrient-rich stormwater.

In summary, these studies show that wetlands treating nutrient runoff in urban areas of similar climates and conditions to the GBR can be effective, especially as vegetation and organic carbon increase over time to support denitrification. Vegetation density is key, as are hydraulic loading rates and residence times. Floating wetlands show potential for nutrient removal, but their effectiveness may be limited under low influent stormwater concentrations or system design. In GBR catchments, where high nutrient loads and marked wet/dry seasonality occurs, floating wetlands might not mitigate existing aquatic weed problems and could even exacerbate them (Waltham et al. 2020b; Waltham and Fixler 2020).

Swales

No studies on swales in non-agricultural GBR settings were identified, however two studies from south east Queensland provide some insights. Vegetated swales are typically incorporated with other treatment measures in a "treatment train" and studies on them as individual treatments are limited. Fletcher et al. (2002) found that vegetated swales in Brisbane, Queensland were effective stormwater treatment measures, removing 44 to 57% of TN, and 58 to 72% of TP concentrations, with similar load reductions TN (40 to 72%) and TP (12 to 67%). Performance declined with increasing flow rate, but TN and TP were less affected than TSS, reflecting the likely influence of rapid chemical processes. In contrast, Kachchu Mohamed et al. (2013) reported limited nutrient removal by new swales on the Sunshine Coast, Queensland, possibly due to nutrient leaching from the newly established swale.

Biofilters

No specific studies on biofilters were identified but seven relevant studies from elsewhere were reviewed (Table 11). Biofilters are soil filtration systems planted with emergent macrophytes and with underflow collection systems such that stormwater is collected on the surface, then infiltrates through the vegetated soil filter where the treated water is collected through underflow drains before flowing out to receiving waters.

Table 11. Studies reporting the effectiveness of biofilters included in this review.

Study	Location
Denman et al. (2016)	VIC, Australia
Jhonson et al. (2022)	Malaysia
Ng et al. (2018)	Australia
Kandasamy et al. (2008)	NSW, Australia
Lloyd and Wong (2008)	VIC, Australia
Lucke and Nichols (2015)	Sunshine Coast, QLD, Australia

Macnamara and Derry (2017)	NSW, Australia

Denman et al. (2016) evaluated the potential to include trees in biofiltration studies through mesocosm column experiments. Vegetated soil profiles reduced nitrogen oxides (NOx) by 2 to 78% and Filterable Reactive Phosphorus (FRP) by 70 to 96% depending on the filtration media (not reported). Jhonson et al. (2022) found maximum removal efficiencies of 86.4% for TN, 93.5% for TP and 90% for TSS in a Malaysian setting, while Ng et al. (2018) reported TN and TP reductions of 47% and 69% respectively in urban biofilters planted with vegetable crops.

In evaluating the performance of sand filters in removing N and P, Kandasamy et al. (2008) showed that two different grades of sand media performed similarly with 61% of TN, 70% of Total Kjeldahl N (organic N + ammonium) and 53% NOx removed from stormwater runoff. TP showed a 40% reduction and was similar to other values for the effectiveness of sand filters reported in the literature.

A paired catchment study by Lloyd and Wong (2008) noted that biofilters reduced pollutant loads by both retaining runoff and through physical and/or chemical treatment processes achieving 100% gross pollutant removal over ten events, 68% TSS removal, 68% TP removal and 57% TN removal. However, Lucke and Nichols (2015) observed variable performance of the treatment of synthetic stormwater including a range of influent concentrations ranging from 'no added synthetic pollution' to five times the typical urban stormwater pollutant loads. Overall, the performance under different hydrologic conditions was highly variable but always positive, with TSS removal being variable and not correlated with influent concentration. For the treatment of 'no-pollution' in the influent, the bioretention systems were shown to have negative removal for TN, however TP was effectively removed across all systems. The field study showed that the performance for biofilters was highly variable and dependent on a range of factors including inflow pollutant concentrations, filter media, construction methods and environmental factors.

Macnamara and Derry (2017) tested monophasic (single type of media) and biphasic (two media types) filter media designs for potential stormwater filter systems in Sydney and found median TN removal efficiencies of 84.1% and 89.0% for monophasic and biphasic designs respectively. TP median removal efficiencies were 77.8% and 68.5% respectively.

Collectively, these studies indicate that biofilters can significantly improve urban stormwater quality. They function well in tropical climates but may experience nutrient leaching from their media as N accumulates and this would need to be managed (Kavehei et al. 2021c). In addition, plant species selection is also an important consideration. System maturity is important, as denitrification rates and nutrient removal often improve as systems age.

Comparison among wetland systems

Table 12 presents the range of reported efficacy values for each water quality parameter for each wetland type used in non-agricultural/urban areas included in this review. Although this review included 25 studies of wetlands treating urban stormwater runoff, some studies did not report the efficiency in percentage of the loads making it difficult to include all of the results. Calculating averages or drawing conclusions is not appropriate at this stage until further evidence becomes available.

Table 12. Overview of the efficiency (% reductions in concentrations) reported by studies included in this review for non-agricultural sources (global). The data presents the range of observed removal efficiencies (%) of all reported water quality variables. n= number of studies. Empty cells are where no data could be found.

Wetland type	Total Nitrogen	Nitrogen Oxides	Total Phosphorus	Filterable Reactive Phosphorus	Total Suspended Sediment	Pesticides
Floating wetlands	17% (n = 1)		52% (n =1)			
Vegetated drains					95% (n = 1)	100% (n = 1)
Restored wetlands	-4% (n = 1)		-11% (n = 1)		-4% (n = 1)	
Reed beds	17 to 49% (n = 1)					
Swales	44 to 57% (n = 1)		58 to 72% (n = 1)			

Wetland type	Total Nitrogen	Nitrogen Oxides	Total Phosphorus	Filterable Reactive Phosphorus	Total Suspended Sediment	Pesticides
Free flow wetlands	Median 41% (n = 1, 17 sites)		Median 17% (n = 1, 17 sites)		Median 11% (n = 1, 17 sites)	
Biofilters	47 to 89% (n = 5)	2 to 78% (n = 2)	40 to 93.5% (n = 5)	70 to 96% (n = 1)	68 to 90% (n = 2)	

3.3 Summary of the effectiveness of wetland and treatment systems in water quality improvement

The collective evidence base showed high variability in the effectiveness of wetlands and treatment systems in processing pollutants in agricultural areas. Global studies show that natural wetlands generally exhibit higher pollutant removal efficiencies compared to constructed wetlands. However, there are fewer studies that evaluate the effectiveness of natural and near-natural wetlands in pollutant processing. Constructed or treatment wetlands such as floating wetlands, vegetated drains, and bioreactors show diverse pollutant removal capacities depending on design factors like inflow pollutant loads, vegetation cover and water residence time, but can be highly effective for nutrient removal in the right conditions.

Wetland systems in non-agricultural areas, primarily constructed vegetated systems, can also play an important role in improving water quality. These systems are designed with engineered inflow and outflow controls and often include pre-treatment methods like sediment basins and floating macrophytes. Research shows variable efficacy in nutrient removal based on the local context, with some wetlands in urban areas achieving high rates of pollutant removal, while others have been less effective, at times even exporting nutrients. As above, factors such as inflow pollutant loads, vegetation cover and system design can greatly impact performance.

Studies also highlight that denitrification is a key process in nutrient removal, improving over time as vegetation and organic carbon increase. Constructed floating wetlands have demonstrated significant potential for nutrient uptake, though results vary based on stormwater inflow concentrations.

In GBR-specific studies, 61 wetlands (including natural, constructed, and bioreactors) were investigated, showing varying efficacy depending on the wetland type and pollutant. Constructed/treatment wetlands and bioreactors were found to be effective at removing nitrogen, but with high variability in removal rates.

4. Factors influencing the effectiveness of pollutant processing in wetlands

Numerous factors influence the capacity of wetlands to process nutrients, sediments, and pesticides. Understanding these factors is essential for designing, maintaining, and predicting the performance of wetlands, as well as evaluating the broader ecological benefits or trade-offs.

Global evidence from tropical and subtropical climates identified several key factors affecting water quality improvement. Prominent among these were:

- **Pollutant type and concentration:** The nature and initial concentration of the targeted pollutant strongly influence removal efficiency.
- **Vegetation community:** The presence, density, and maintenance of local plant species is widely cited (36% of studies) as a critical factor (e.g., Bhomia and Reddy 2018; Min et al. 2015; Zhao et al. 2012).
- **Hydrology and residence time:** Controlled water flow and adequate residence time are highlighted in 20% of studies (e.g., Kim et al. 2004; Knox et al. 2008; Sim et al. 2008; Wilcock et al. 2012). Managing these factors can significantly enhance wetland efficacy.

These factors combined, if understood, controlled, and managed appropriately, can improve the efficacy of wetlands for the objective of water quality improvement. Variations in wetland size, age and drainage area also contribute to differences in removal efficiency (e.g., Bason and Kroes 2017; Campaneli et al. 2021; Jia et al. 2019; Khare et al. 2019; Tanner and Kadlec 2013). Seasonal sampling also affects outcomes – studies that monitor both wet and dry seasons provide a clearer understanding of long-term performance (e.g., O'Geen 2006). Measurement of hydrology including retention time, rainfall and groundwater influence alongside water quality is also an important consideration (Appelboom et al. 2008; Cai et al. 2017; Wallace et al. 2022). Uncertainties in monitoring may also contribute to variability in the reported performance of systems, but also the inability to attribute pollutant removal to specific factors, for example, if only surface water runoff and not groundwater is measured in assessing hydrological factors.

In addition to the aforementioned factors, a number of variables have been described in the literature as important, but to a lesser degree when compared to vegetation and hydrology. These include, for example, landscape context (e.g., land use upstream of wetlands – 5% of studies), environmental factors including climate change (9%), and sediment delivery and storage/accumulation (2%). Surprisingly, management actions such as maintenance were rarely mentioned as a contributing factor to reduced efficacy (7%), which is interesting given vegetation maintenance was considered as highly important. A range of other factors were highlighted (16%) including the accumulation of rubbish, which provides important insight into the need to consider local nuanced conditions for each project site. The landscape context of a wetland is important information that is not always reported, inhibiting the ability to understand the wetland performance relative to the surrounding land use. The number and spatial pattern of wetlands within a catchment can also be an important consideration (Eberhard et al. 2017). This aspect has been found to impact wetland efficiency in temperate regions (e.g., Eberhard et al. 2017; Hansen et al. 2018), and a more recent study conducted in the Tully and Johnstone basins in the Wet Tropics region considered wetland aggregations and their position in the landscape to measure end-of-system efficacy in processing DIN (Askildsen et al. 2020).

The 2017 SCS (Eberhard et al. 2017) also highlighted that at a local scale, wetlands can be considered important for decreasing nutrient loads based on three lines of evidence: 1) the natural process of denitrification in wetland systems, 2) the ability of wetland soils to store nutrients and carbon, and 3) the high productivity of wetlands and their ability to absorb nutrients through plant growth. It was recognised that these functions depend on several factors, and in the GBR catchments, residence time during flood conditions is a key limiting factor for nutrient removal.

The following sections present further detail on the factors influencing wetland effectiveness in pollutant processing.

4.1 Hydrology/hydraulics and residence time

Wetlands are dynamic, transitional ecosystems that vary over complex spatial and temporal scales. Wetland hydrology varies with seasonal rainfall, runoff, land-based activities, groundwater interactions and tidal influences. This variability affects their biological functions and water quality improvement capacity.

Agricultural land uses

From the global review of tropical and subtropical climates, water flow rate and hydraulic residence time were paramount for nutrient removal efficiency. Residence time, nitrate distribution, moisture, and vegetation species all influence denitrification. TN removal is more effective under low flow or in static water conditions, whereas ammonium (NH_4^+) removal can be higher in fast flowing waters (e.g., Kröger et al. 2012; Mu et al. 2020; Yi et al. 2010). High variability in nutrient removal is frequently linked to variations in inflow and loading rates, inflow volumes and thus hydraulic residence time (Zhang et al. 2017). Studies that measure water quality and hydrology over both wet and dry seasons capture this variation more effectively (Jordan et al. 2003; Kaplan et al. 2011; Laterra et al. 2018; Moustafa 1999; Niu et al. 2016).

Engineering wetlands to control water passage and increase residence times can enhance treatment efficiency. Larger wetlands, appropriately sized wetlands, or natural settings in an appropriate design, combined with sufficient residence times, can provide greater treatment opportunities (Birgand et al. 2016; Littlejohn et al. 2014; O'Geen et al. 2007; She et al. 2018). Using modelling approaches, Ji and Jin (2016) also found that increasing water depth in wetlands can improve retention time and pollutant processing. In constructed wetlands, low levels of denitrification and therefore low TN removal efficiency were attributed to hydraulic residence time (Lin et al. 2015). Hydrology can also influence the denitrification rate by influencing the distribution of nitrate-N and moisture in riparian ecosystems, thereby determining the rate and location of denitrification (Schnabel et al. 1997).

Seasonal variation is also an important influence on effectiveness, and therefore an important consideration in design. About 36% of the studies examined both wet and dry seasons, whereas 5% examined the wet season only, ~2% examined the dry season only and the remainder did not provide seasonal information. Without this context, it is challenging to interpret the results reliably. Of the studies conducted over both the wet and dry seasons, 26% (n = 19) and 25% (n = 18) found that vegetation and hydrology respectively affected the water quality improvement efficiency of wetlands. Sampling across both wet and dry seasons is important to effectively understand wetland water quality processes, function and removal efficiencies in the long-term. For pesticides, several studies only sampled during the wet season and many others did not provide this contextual information.

For soluble pesticides such as neonicotinoids, hydrology may be particularly important as longer residence times may increase exposure to UV light, enhancing pesticide breakdown. High intensity rainfall events that shorten the residence time can reduce water soluble herbicides (atrazine, metolachlor and glyphosate) efficiencies (Lerch et al. 2017). Sampling in a single season therefore biases understanding of a wetland’s capacity for longer-term water quality improvement (e.g., Tanner et al. 2005; Zhao et al. 2019).

The 2017 SCS (Eberhard et al. 2017) recognised the potential limitations of wetland processing in flood conditions when residence time is a major limiting factor. The following information is sourced from Eberhard et al. (2017). In the GBR during large floods, nutrients are rapidly mobilised through the river channels into the coastal zone and marine environments (Davis et al. 2016), and wetlands may provide little protection from the large amounts of nutrients discharged into the GBR during these events (McJannet et al. 2012). The exceptions to this could be small sub-catchments, where the ratio of wetlands to other land uses allows processing to occur, or in deltaic systems, where large areas of wetlands and floodplain are flooded for long enough to process the nutrients before they are exported to the GBR. During low flow events, pollutants may be removed through natural wetlands and treatment systems, meaning they are not available for mobilisation during high-flow events. Some wetlands or treatment systems may capture first-flush events, even in high-flow events, if a bypass system is in place. Deltas usually have slower flows even during high-flow events (because of the flat landscape) and operate more like slow flow systems. In irrigated systems, pollutants are constantly removed by wetlands and are not as available in high-flow events for movement.

Non-agricultural land uses

Hydrology also matters in non-agricultural contexts. Using 240 mesocosm columns in a laboratory setting, Payne et al. (2014) found that biofilters performed well for TN removal during wet periods but were less reliable following a 15-day dry period. These authors also highlighted that the plant species was of limited importance under wet conditions provided the filter medium was carefully specified to reduce nutrient leaching, though the species selection became a differentiator for performance during extended drying. This may have implications for the design of biofilters in tropical climates in the GBR catchment area if systems are allowed to dry out extensively in the dry season.

Bourgues and Hart (2007) examined the roles of epiphytic biofilms and sediment processes in 13 urban stormwater wetlands in Melbourne and noted that in systems with a supply of nitrate, low oxygen levels and appropriate redox conditions, high levels of denitrification were observed.

Surface area and hydrologic length of contact between the riparian zone and stream sources of nutrients were important for N removal in riparian wetlands in North Carolina in the US. Although not climatically similar to regions in the GBR, it does highlight that these factors need to be considered for the design of wetland systems. Sakadevan and Bavor (1999) examined five experimental wetlands in Richmond, NSW, and found that low hydraulic loading and greater retention times enhanced the removal of N and P, though this was for wastewater rather than stormwater runoff.

4.2 Vegetation community

After hydrology, vegetation is the second most influential factor in pollutant processing. Vegetation plays a significant role because it accumulates nutrients and supports denitrification.

Agricultural land uses

From the global review of tropical and subtropical climates, the presence of vegetation often explains differences in nutrient removal efficiency (e.g., Lu et al. 2010; Menon and Holland 2013). For instance, one study reported efficiencies ranging from 59 to 65% in unvegetated plots compared with 86 to 88% in vegetated plots, with alien and indigenous plant assemblages performing similarly (Jacklin et al. 2020). While species composition can influence removal efficiencies, the magnitude of variability is also affected by factors such as inflow volume/rate, hydraulic residence time, and vegetation presence/absence (Jacklin et al. 2020; Lu et al. 2014; Wang et al. 2019a).

Vegetation is particularly influential in denitrification rates, with denitrification in a tropical Australian floodplain greatest in sediments sampled beneath grass, followed by sediments beneath waterlilies, and lowest in sites without aquatic plants (Adame et al. 2021b). Species of vegetation can also affect denitrification rates and nitrogen removal processes. For example, drainage ditches vegetated with *Myriophyllum elatinoides* were found to remove NH_4^+-N through both plant uptake and microbial nitrification-denitrification, whereas ditches vegetated with *Pontederia cordata* mainly removed NH_4^+-N through sediment sorption (Zhang et al. 2016).

Based on the premise that longer residence time and more vegetation results in a higher processing rate of nutrients, some experimental studies have combined both flow and vegetation communities to optimise flow rates for specific vegetation species (Wang et al. 2019a). Zhang et al. (2021) found that TN removal efficacy in wetlands increased from 17.95% in unvegetated wetlands to 29.8% in vegetated wetlands; Tyler et al. (2012) found that TN efficacy increased from 26.9% in unvegetated to 50% in vegetated wetlands; and Sasikala et al. (2009) found that TN improved from 44% to 58.2% removal with vegetation, and this increased further to 67.4% with fluctuating water levels.

In treatment wetlands in the GBR, Kavehei et al. (2021a) found that nutrient removal was highest when vegetation cover was greater than 50%. Different macrophyte species vary in their ability to process nutrients (Ibekwe et al. 2007; Rigotti et al. 2021). Additionally, Adame et al. (2019b) modelled the denitrification rates of a range of natural wetlands from within the Tully-Murray catchment (Melaleuca, Melaleuca-Eucalyptus, saltmarsh, freshwater marsh, mangrove, floodplain wetlands, coastal lagoons with water lilies and coastal lagoons with emergent grasses). Whilst the vegetation densities within these habitats were not reported, the study found that the coastal lagoon wetland, with emergent grasses and water lilies (*Nymphaea* spp.), had the highest denitrification rates of 52 mg m^2 h^{-1} and 24 mg m^2 h^{-1} respectively, whilst the mangrove forest dominated by *Bruguiera gymnorhiza* had the lowest denitrification rate reported of 3.8 mg m^2 h^{-1}. Excessive growth can present a challenge for water quality objectives as the presence of excessive aquatic weeds can contribute to poor water quality conditions for aquatic species such as fish (Veitch et al. 2007).

The extent of the vegetation is also an important consideration. Larger and intact grass swales were more effective in slowing flow sufficiently for improved water quality than riparian areas, particularly for nutrient species and some pesticides (Welsh et al. 2019; Yorlano et al. 2021). In addition, Alemu et al. (2017) showed that increasing riparian buffer strip widths from 3 to 10 m improved nitrate removal efficiency from 50% to 85%, TP from 47% to 99% and TSS from 76% to 94%. Mature tree species are also capable of removing nutrients from within the edge areas of wetlands, given that wetlands can expand and contract depending on local hydrology and rainfall (Adame et al. 2019a).

Vegetated treatment systems and buffer strips are also effective at increasing sediment removal efficiencies (Arora et al. 2010). Efficiencies can be increased further through the addition of retention basins and sediment traps (Alemu et al. 2017; Phillips et al. 2021; Zhao et al. 2016). Plant community density and root density are also important variables in determining a wetland's water quality improvement efficiency, although this is not a linear relationship (Ibekwe et al. 2007; Lv and Wu 2021; Shahid et al. 2018). Evidence of enhanced denitrification over time was demonstrated in three experimental field trial ponds, where plant biomass uptake was important initially, and sediment became increasingly important as a sink (Lund et al. 2001). In four constructed stormwater urban wetlands, denitrification was also shown to be a dominant nutrient reduction process by Rahman et al. (2019a, 2019b, 2019c), with organic carbon and high nitrate concentrations important influences in favouring of denitrification over nutrient recycling by bacteria. Water column processes such as carbon to nitrogen (C:N) ratios can also affect N removal through denitrification. The presence of vegetation can affect this ratio and subsequently denitrification, with high plant productivity potentially increasing C:N ratios in the water column (Hume et al. 2002). The artificial addition of carbon also significantly increased denitrification rates in treatment wetlands (Liang et al. 2020; Martínez et al. 2018).

Non-agricultural land uses

Vegetation is also key in biofilters. Denman et al. (2016) evaluated the potential to include trees in biofiltration studies through mesocosm column experiments, showing that vegetated soil profiles generally reduced NOx concentrations and FRP consistently greater than unplanted profiles. Vetiver grass out-performed blue porterweed, hibiscus, golden trumpet and tall sedge under tropical conditions (Malaysia) (Jhonson et al. 2022). Biofilters planted with vegetable crops showed TN and TP reductions of 47% and 69% respectively (Ng et al. 2018). Hatt et al. (2006; 2007; 2009) investigated filter clogging, different media performance, and different climate conditions in biofilters. In general, non-vegetated systems are net producers of N (except for sand media) and exhibit variable performance for P. Collectively, these studies show that vegetation is a critical component of the treatment process to achieve high N removal in biofilters.

Bourgues and Hart (2007) examined the roles of epiphytic biofilms and sediment processes in 13 urban stormwater wetlands in Melbourne and noted a high degree of heterogeneity across the systems. Biofilms on macrophytes were important to shelter bacteria that were able to carry out denitrification at comparable rates to those in adjacent sediments, highlighting the importance of plants to treat nutrient rich stormwater. Headley et al. (2001; 2005) also observed that nutrient uptake by plants was important and accounted for greater than 70% of the nutrient removal initially. Lund et al. (2001) also showed that plant biomass uptake was important initially. Vegetation also played a significant role in the capture of suspended solids and nutrients in a wetland system in Brisbane, Queensland and limited the resuspension of these pollutants into the water column (Kasper and Jenkins 2007).

4.3 Wetland area, shape and configuration

From the global review of tropical and subtropical climates, differences in wetland area, volume and residence times helped explain variations in removal efficiencies (Kaplan et al. 2011). Highly variable removal efficiencies were associated with altered inflow and hydraulic residence time (Jordan et al. 2003; Kato et al. 2007; Niu et al. 2016).

Engineered constructed wetlands ranged in size (and depth), with most wetlands from the global review less than 1 km^2 in total size (e.g., Jordan et al. 2003). When natural wetlands are included, wetland area increases, (e.g., 2–3 km^2 in Adame et al. (2019a) to 9 km^2 in Neubauer et al. (2019)). Basic site information relating to the location and dimensions of study wetlands, their spatial pattern within a watershed, as well as the presence and number of other, connected wetlands, should be a standard set of details reported in publications. Of particular interest but omitted largely by the subtropical/tropical body of evidence, is the relationship between these variables and the size and shape of wetlands, as well as the cumulative effect of multiple wetlands within a watershed. Unfortunately, wetland size is rarely reported (7% of global studies).

Of the 16 GBR studies featured within the body of evidence, two studies reported wetland volume, four reported wetland depth, five reported wetland age, and nine reported wetland area. Table 13 shows the removal efficiencies of natural/near-natural, treatment/constructed, and bioreactors in the GBR catchment which are ordered according to their area in hectares. Overall removal efficiencies were highly variable with no apparent relation to their area.

Table 14 shows the removal efficiencies of wetlands in the GBR catchment and their age in years. The table showcases the range of removal rates for TN and nitrate/DIN, highlighting variations in efficiency over different stages of wetland development. Removal efficiencies fluctuate considerably across age categories and wetland systems and detecting any age-related patterns is hindered by the few studies available for each age group.

Table 13. Overall removal efficiencies (%) of natural/near-natural, treatment/constructed wetlands and bioreactors from studies in the GBR catchment according to their area (ha). Blank cells contain no data.

	Removal efficiency (%)						
	Bioreactor	Constructed/Treatment	Natural				
Area (ha)	NO_3^-	DIN	TN	NH_4^+	NO_3^-	TP	TSS
0.002	84%						
0.3		80%					
0.5		50%					
1.2		-15%					
1.3		90%					
1.6		80%					
1.8		10%					
2.1		45%					
2.5		10%					
8.5		70%					
25			-4%			14%	1%
2,213			29%	19%	86%		

Table 14. Removal efficiencies of natural/near-natural, treatment/constructed wetlands and bioreactors from studies in the GBR catchment according to their age (years). N-values represent the number of wetlands that informed the range. Blank cells contain no data.

Age (years)	Removal efficiency			
	Bioreactor		Constructed/Treatment	
	TN	NO_3^-	DIN	TN
1.5		84.3% (n = 1)		
1.6	41% (n = 1)			
3			10% (n = 1)	-5% (n = 1)
9			45 to 90% (n = 2)	25 to 80% (n = 2)
10			10% (n = 1)	-5% (n = 1)
11			-15 to 80% (n = 2)	
12			70% (n = 1)	
13			50 to 80 (n = 2)	

The association of the shape of the wetland and its efficiency was studied by Kavehei et al. (2021a) who evaluated N removal across eight constructed wetlands and two vegetated drains in the GBR catchment area. The study reported that rectangular wetlands, with a width to length ratio of at least 1 to 3, had a better hydraulic efficiency, and consequently an improved retention time and nutrient removal. Another study conducted by Kavehei at al. (2021c) in bioretention basins showed that a C:N ratio of greater than 20 is required in the bioretention soil to reduce N leaching and N_2O production. Of the 25 systems evaluated in sub-tropical Australia, the results showed that most C:N ratio values were above 25 which was important to promote N removal, and the oldest systems showed high C:N ratios with the denitrification potential increasing significantly with the age of the system. This shows the importance of the role of carbon in understanding denitrification potential in vegetated systems.

Wetland size to catchment area ratio (%) influenced the removal efficiency of DIN from treatment wetlands (recycle pits and constructed wetlands) in a synthesis of data for the GBR in 2016 (Alluvium 2016; Table 15), indicating that both the wetlands included and recycle pits needed to occupy significant areas to have a reasonable treatment efficiency. From the results, wetlands need around twice the area of recycle pits to get the same removal rate until the rate between the wetland and catchment size is about 10%, where both the recycle pit and a constructed wetland seem to have similar performance.

Table 15. Estimated DIN efficacy for Burdekin wetlands and recycle pits and Wet Tropics wetlands (Alluvium 2016).

Wetland type	Catchment	Weather	Wetland size to catchment area ratio (%)	Removal efficiency (%) - DIN
Recycle pit	Lower Burdekin	Recycle pit - operating in dry and wet weather	2% w:c	18
			5% w:c	38
			10% w:c	67
			20% w:c	89
Constructed Wetland	Lower Burdekin	Wetland - operating in dry and wet weather	2% w:c	8
			5% w:c	24
			10% w:c	62
			20% w:c	80
	Wet Tropics	Wetland - operating in dry and wet weather	2% w:c	1
			5% w:c	10
			10% w:c	42
			20% w:c	49

4.4 Other Factors

Several additional factors can influence wetland pollutant processing including:

- **Climate:** Antecedent conditions can be an important factor when measuring the water quality improvement efficiency of wetlands, particularly following storm activity (Gall et al. 2018). For example, seasonal and long-term variations in soil moisture deficit and evapotranspiration can significantly impact wetland hydrology, leading to large variations in sediment removal efficiency among similarly sized storm events (Gall et al. 2018).
- **Temperature:** Temperature strongly affects denitrification rates (Adame et al. 2021a; She et al. 2018). For example, in experiments replicating near-natural wetlands in Louisiana in the US, denitrification rates doubled when temperatures increased from 14 to 20 °C (Bowes et al. 2022).
- **Geology:** Soil composition, erosion and physicochemical conditions influence nutrient and pesticide sorption or leaching (Axt and Walbridge 1999; Cao et al. 2018; Kao et al. 2002; Lerch et al. 2017). This is especially important for organochlorine, organophosphate and synthetic pyrethroid pesticides that have low solubility in water and tend to bind with particulate matter and become deposited in sediments.

  While acid sulfate soils were not covered in great depth within the body of evidence, it is important to highlight that the drainage of wetlands for urban development and agriculture can disturb and/or create ideal conditions for the development of acid sulfate soils. Moreover, the re-wetting of wetlands containing acid sulfate soils (e.g., as a wetland restoration initiative), can potentially lead to large-scale sulfuric acid generation and runoff, compromising wetland ecosystem service provision (Luke et al. 2017). The sulfuric acid in these soils can leach into groundwater and urban drainage systems, compromising potable water quality. Acid soils and drainage water can negatively impact local wildlife, reduce biodiversity, fisheries and agricultural production, as well as damaging infrastructure.

4.5 Summary of the factors influencing the effectiveness of pollutant processing in wetlands

The evidence demonstrates that several factors influence the effectiveness of pollutant processing in wetlands with key factors such as hydrology, vegetation community and wetland size playing important roles. Hydrology, particularly water flow rates, hydraulic residence time, and seasonality, directly impacts nutrient, sediment, and pesticide removal efficiencies. Wetlands with longer residence times generally show higher pollutant removal rates. The role of vegetation is equally significant, as wetlands with higher vegetation density or cover have shown better performance in processing pollutants, especially nitrogen. The presence of plants also helps with sediment trapping and supports microbial processes such as denitrification, which enhances nutrient removal.

Other factors that influence effectiveness include climate, temperature, size and location in the landscape (including adjacent and upstream land uses). Larger wetlands with an ideal size-to-catchment ratio tend to be more effective, particularly in agricultural settings close to the source where pollutants are concentrated. Additionally, certain biogeochemical processes, including carbon:nitrogen ratios and sediment processes, contribute to improved pollutant removal.

5. Policy, cost and investment considerations

Managing wetlands for the protection of wetland values has been the focus of past Australian and State government programs in Queensland. There have also been a large number of long-term wetland programs overseas which provide policy and program insights. This is important context for understanding the potential role of wetlands in water quality improvement in the GBR.

This section describes the current policy context in the GBR catchments, the implications for wetland projects and programs, and the drivers and costs of managing wetlands for water quality improvement. It also highlights local learnings about the implementation of on-ground projects, and considerations for investment. Evidence from outside of the GBR was also reviewed and included where relevant.

5.1 Current wetland policy

The Queensland Government shares responsibility for the management of wetlands with the Australian Government, local governments, Traditional Owners, landholders and the wider community. These responsibilities are formalised in laws passed by the Queensland and Commonwealth governments, through international obligations, agreements and a suite of policies and programs.

Relevant Australian Government policy and programs

A range of laws, policies and programs administered by different government agencies operate to regulate and manage the different wetlands in our environment. Australia, through the Commonwealth government, is also signatory to many international agreements which also apply to State jurisdictions, including the international Ramsar Convention which aims to halt the worldwide loss of wetlands and to conserve remaining wetlands through wise use and careful management.

The Australian Government has commitments under the Ramsar Convention and responsibilities under the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC Act). The EPBC Act lists the matters of national environmental significance as: world heritage properties, national heritage places, wetlands of international importance (often called 'Ramsar' wetlands after the international treaty under which such wetlands are listed), nationally threatened species and ecological communities, migratory species, Commonwealth marine areas, and the Great Barrier Reef Marine Park.

Relevant Queensland Government policy and programs

In 2003, the Queensland Wetlands Program was established to protect wetlands in the GBR catchment and throughout Queensland and has been funded by Queensland for the past decade.

Since 2003 the program has supported projects that have delivered a range of new mapping, information and decision-making tools that has enabled government agencies, landowners, conservationists and regional natural resource management bodies to better protect and manage wetlands into the future.

The first formal strategy for wetland management in the GBR was established in 2016, *Wetlands in the Great Barrier Reef Catchment Management Strategy 2016-2021* (DEHP 2016). Projects that support the themes within the Strategy include: the intertidal and subtidal ecosystem mapping for Central Queensland; Walking the Landscape workshops; delivery of Ramsar Management Advisory Groups; shorebird monitoring activities; the reporting of wetlands extent change; litter and illegal dumping information being made available in Wetland*Info*; and making wetlands data available in QGlobe. An evaluation of the Strategy was carried out in 2021, confirming that over 60 organisations were involved in delivering relevant wetland activities in the GBR catchments and had used the Strategy to inform the development of their own strategies and work programs.

Of critical relevance, Wetland*Info* was developed in 2007 as a first-stop-shop for wetland information in Queensland, providing a range of tools and resources to assist with the sustainable management of wetlands. Wetland*Info* has continued to develop and integrate a range of resources including facts and interactive maps and maps for download, summaries of wetland information, including information on N processing by wetlands and information on treatment wetlands, online education resources, factsheets, management guides, case studies, project guidance including monitoring and evaluation, assessment tools and links to other resources including relevant programs, policy and legislation.

As noted in Section 1, the Strategy was updated in 2023 through an extensive consultation process (*Reef 2050 Wetlands Strategy: A strategy for managing wetlands in the Great Barrier Reef and its catchments,* DESI 2023a) and includes the following Themes and goals:

- Theme 1. Improving wetlands information for decision making and action: Up-to-date, scientifically robust and integrated information is available for evidence-based decision making and informing best practice protection and management of wetlands.
- Theme 2. Wetland planning: Statutory and non-statutory planning arrangements are in place to protect, manage and enhance wetlands.
- Theme 3. On-ground activities to protect, manage, rehabilitate and restore wetlands: Implement on-ground activities that improve the health of wetlands and enhance their contribution to the GBR's resilience.
- Theme 4. Engagement, education, communication and capacity building: Improved awareness of the value of wetlands, management tools and involvement in wetland planning and management.
- Theme 5. Monitoring, evaluation, reporting and improvement: An adaptive management approach incorporating effective monitoring, evaluation, reporting and improvement is implemented to improve wetland management.

The Queensland Government has also enacted several pieces of legislation for the protection of wetlands which are important when considering wetland management in the GBR catchment area:

- *Planning Act 2016*, which aims to facilitate ecological sustainability by providing a legal framework for land use planning, development assessment and dispute resolution in Queensland. Of particular relevance is State Development Assessment Provisions State Code 9: Great Barrier Reef wetland protection areas.
- *Environmental Protection Act 1994* and associated regulations, which aims to achieve ecologically sustainable development in Queensland and specifically the *Environmental Protection (Water and Wetland Biodiversity) Policy 2019* (Water and Wetlands EPP) which establishes a framework for

identifying environmental values (EVs) for waters and wetlands, and water quality objectives (WQOs).

- *Marine Parks Act 2004* and the *Fisheries Act 1994*, which protects important marine and estuarine areas in Queensland through the declaration and management of marine parks and fish habitat areas and protection of marine plants.
- *Nature Conservation Act 1992*, which safeguards Queensland's diverse range of protected animals and plants.
- *Vegetation Management Act 1999*, which regulates the clearing of native vegetation in Queensland.
- *Coastal Protection Management Act 1995*, which supports the protection and management of the coasts, coastal resources and biodiversity and minimises the impacts of coastal hazards.

The Queensland Government policies are also aligned with global commitments such as the Ramsar Convention, and Australian Government legislation like the EPBC Act. Through initiatives like the Queensland Wetlands Program and the Reef 2050 Wetland Strategy, a comprehensive framework has been developed to protect, rehabilitate, and manage wetlands in their own right, delivering critical environmental, social, and economic benefits, including biodiversity conservation, improved water quality, flood mitigation, and carbon sequestration.

The establishment of tools and their integration on Wetland*Info*, the enactment of relevant legislation, and the ongoing projects guided by the updated Reef 2050 Wetlands Strategy have created a robust foundation for sustainable wetland management. The focus on improving information for decision-making, implementing on-ground activities, enhancing engagement, and using adaptive management ensures the long-term resilience of wetlands and their contributions to the GBR ecosystem. Co-ordination across government agencies, landholders, and communities will be essential to safeguard these vital ecosystems and achieve national and global conservation objectives.

The Whole-of-System, Values-Based Framework is a management framework developed through the Queensland Wetlands Program that 'draws explicit connections between the biophysical environment, the beneficiaries of the services provided by that ecosystem and their values' (DESI 2022c). The Framework uses a holistic management approach to achieve outcomes that consider the biophysical environment alongside social, economic and cultural outcomes.

5.2 Policies and programs for multiple benefits

As described in Section 2, wetlands provide a diversity of ecosystem services and benefits to support ecological, social, economic and cultural values including First Nations values. In addition to the management strategies, policies and programs outlined above, there are also a range of policies that can account for 'co-benefits' that are specifically relevant to wetland management for water quality improvement in the GBR catchment area (refer to Star et al. 2024b).

Co-benefits can generate a range of additional benefits, such as carbon sequestration, biodiversity, fisheries habitat and N reduction (Hagger et al. 2022; Strand and Weisner 2013). In Australia there is increasing development of markets to facilitate water, carbon and biodiversity outcomes. Currently in Queensland, there are a number of market-based instruments (MBIs) for environmental policies that are operating in parallel to water quality outcomes, based on a quantity-based approach. The primary instruments are described in Section 5.1 and include, *inter alia*:

- Australian Carbon Credit Units (ACCUs), designed for the *Carbon Credits (Carbon Farming Initiative) Act 2011* established the Emissions Reduction Fund in 2015.
- Reef Credits[6], administered by EcoMarkets Australia and designed for the removal of N specifically from sugarcane catchments, reduced sediment losses as a result of gully restoration or grazing land management.

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[6] https://eco-markets.org.au/methodologies/

- Queensland's Land Restoration Fund (LRF), which includes increased incentives to landholders for participating in carbon credits and generating ACCUs. The LRF allows bundling and seeks socio-economic, environment and Indigenous co-benefits.
- Agriculture Biodiversity Stewardship Pilots, which aim to achieve biodiversity, carbon and water quality outcomes.

Each of these existing market mechanisms has different characteristics for landholders seeking to participate in the market. For example, Reef Credits and the Carbon Credit Markets do not allow additionality of co-benefits, whereas the LRF and the National Stewardship Trading Platform do allow additionality, but these are not open access markets and there are set periods for calls for projects.

Table 16. Market-based instrument programs and attributes relevant to co-benefits from management practices that improve water quality in the GBR.

MBI program	Allows participation in other MBI programs	Credit generation time frame	Market access
Australian Carbon Credits	No	25-year or 100-year	Open
Reef Credits	No	10 years	Open
Land Restoration Fund	Yes - ACCU	5 to 15 years	Call for applications during set periods
Agriculture Biodiversity Stewardship Pilots Carbon, Biodiversity and Carbon Plus	Yes - ACCU	10 years	Pilot areas - calls for applications during set periods

Each of the MBI's has a series of methods to assess the environmental change, with the most established market and method for assessment for ACCUs. For carbon reduction methods the generation of ACCUs includes active methods for agriculture in:

Cattle:

- Reducing greenhouse gas emissions by feeding nitrates to beef cattle
- Beef cattle herd management

Irrigated cotton:

- Reducing greenhouse gas emissions from fertiliser in irrigated cotton

Soil carbon:

- Estimating soil organic carbon sequestration using measurement and models method
- Estimating sequestration of carbon in soil using default values (model-based soil carbon)

Savanna fire management:

- Savanna fire management 2018—emissions avoidance
- Savanna fire management 2018—sequestration and emissions avoidance.

There are also vegetation methods established for different situations:

- Avoided clearing of native regrowth
- Designated Verified Carbon Standard projects
- Plantation forestry
- Reforestation and afforestation V2.0
- Tidal restoration of blue carbon ecosystems method (Blue Carbon – BlueCAM - see below).
- Reforestation by Environmental or Mallee Plantings Full Carbon Accounting Model (FullCAM)

In 2021, the Clean Energy Regulator (Australian Government) prepared a Blue Carbon method to activate market mechanisms for industry and investment schemes to fund restoration of coastal wetlands, including mangroves and tidal marshes for their greenhouse gas reduction services (Clean Energy

Regulator 2021). The method focuses on tidal re-introduction via a managed realignment of earthen bund walls, with ACCUs awarded for greenhouse gas abatement with coastal wetland restoration.

The LRF is Queensland's $500 million fund investing in high quality carbon farming projects and was established in 2017[7]. The LRF invests in land management projects that not only keep carbon in the ground, but also deliver positive impacts or co-benefits for the environment and communities which is of increasing interest in Australia's environmental policies (Australian Government 2022). The LRF defines co-benefits specifically, all of which are relevant to wetland management for water quality outcomes in the GBR:

- Environmental co-benefits: Improved biodiversity, habitat for threatened species and healthier soils, wetlands, and water.
- Socio-economic co-benefits: Improving the resilience and prosperity of regional communities by supporting jobs and skills and generating economic benefits for local communities.
- First Nations co-benefits: A broad range of co-benefits including customary, cultural, economic and business development benefits, such as providing new on-country and service delivery business opportunities and supporting cultural and customary connections.

ACCUs can be co-benefits linked to biodiversity, soil, and Indigenous outcomes. Projects that generate co-benefits are able to attach ACCUs generated by projects under the LRF. Essentially, co-benefits under these programs can be carried out on the same area that water quality improvement management practices are undertaken. Both the LRF and the ACCUs apply the Accounting for Nature Framework which requires projects to monitor and report the nominated outcomes to ensure additionality is achieved (e.g., wetlands and biodiversity outcomes).

The LRF directly identifies that GBR projects with environmental and/or social and/or First Nations co-benefits are eligible. To claim a GBR co-benefit, LRF projects must result in: a) a verified improvement to native vegetation in pre-clearing wetlands in a GBR catchment; and/or b) a verified improvement to both native vegetation condition and soil condition within a GBR catchment that has a sediment target in the Reef 2050 WQIP. Along with this soil health, wetlands, coastal ecosystems, threatened ecosystems, threatened wildlife, and native vegetation are all considered to contribute to the environmental co-benefits (DESI 2023a).

Opportunities also exist in the GBR coastal area for blue carbon projects, for example, through the recently developed Australian Government's Nature Repair Plan (DCCEEW 2022). These projects deliver outcomes through activities such as engineering wetlands designed to intercept and process available nutrients and sediments, or removing earth walls and allowing tidal waters to ingress which could potentially generate blue carbon credits (these low-lying areas would transition or return to mangrove and saltmarsh areas which sequester carbon) (Jenkins et al. 2010). There is also a call for caution in the consideration of removing or modifying earth walls or tidal restrictions built for ponded pasture wetlands, as in some places, these effectively provide some of the last remaining freshwater ecosystems. Removing earth walls and transitioning a freshwater ponded pasture habitat to a blue carbon ecosystem habitat could result in a negative outcome for freshwater dependent species (Abbott et al. 2020). In addition, the assumption is that once the tidal wall is breached, marine vegetation (including supratidal species like *Melaleuca*) will colonise and provide carbon sequestration abatement, however, this may not always be the case.

Despite the development of various MBIs in Queensland aimed at environmental outcomes, there are currently no MBIs specifically designed to support wetlands or water quality improvements in the GBR catchment area. Existing instruments, such as Reef Credits and the Land Restoration Fund, can address related outcomes like carbon sequestration and nitrogen reduction but do not provide direct incentives for wetland-specific water quality management.

[7] Land Restoration Fund

5.3 Cost Drivers

A cost driver is defined as any factor, index, event or coefficient that causes a change in the costs and which is the basis for cost allocation, and a measured cost has monitored and primary data for dollars per item or intervention associated with the management or improvement. The cost drivers of wetland rehabilitation and restoration projects are influenced by a range of factors including policy setting, program selection, and biophysical features. Understanding the overarching cost drivers is crucial for designing and implementing effective wetland management strategies that achieve water quality improvement objectives while minimising costs.

A significant proportion of the literature on cost drivers and costs of constructed treatment wetlands comes from the US. In the US, the Federal Clean Water Act does not regulate non-point source pollution from agriculture (Soldo et al. 2022). Harmful algal blooms triggered by excess nutrient concentrations are major concerns for the Great Lakes, the Gulf of Mexico and Chesapeake Bay (Aggarwal et al. 2022; Soldo et al. 2022; Stephenson et al. 2021). Lacking a regulatory policy, voluntary uptake of improved management practices in arable cropping and livestock grazing is incentivised to reduce non-point source N and P loads to receiving waters. Practice-based cost-sharing subsidy schemes are the predominant approach for incentivising constructed wetlands as a component of improved management practices (Cheng et al. 2020). Similarly, Sweden and Denmark have had long-term wetland programs which provide insights into the cost drivers of wetland rehabilitation and restoration projects over longer timeframes, and provide relevant learnings for this review.

Influence of policy and program selection on cost effectiveness

As described above, there are a range of policies, programs and instruments that are likely to influence the success of programs aiming to implement improvements to water quality via wetland systems. The mechanism used to incentivise wetland rehabilitation or wetland/bioreactor construction for water quality improvement varies between Australian and Queensland jurisdictions, and also depends on the water quality objective(s) and overarching policy context. Differences in the policy context and economic mechanism can influence the costs of wetland construction/rehabilitation, either directly at project level by imposing conditions on wetland design, advisory or extension services, intended outcomes and monitoring requirements, or indirectly at the program level by affecting the cohort of farmers who participate (Graversgaard et al. 2021; Mewes 2012; Stephenson et al. 2021).

Policy context can also be a driver of wetland construction/restoration cost as demonstrated in the evidence summarised below.

Incentives

- Differences in construction/rehabilitation or location requirements under incentive mechanisms can influence wetland costs directly, due to the timeframes involved, desired outcomes and biophysical landscape traits. These may be particularly relevant if wetland rehabilitation aims to deliver multiple benefits (e.g., carbon sequestration and storage, water quality improvement, biodiversity enhancement, hunting opportunities) (Hagger et al. 2022; Soldo et al. 2022; Stephenson et al. 2021).
- Under the Wetland Reserve Program (WRP), the US seeks to achieve eight benefit categories, with water quality just one aspect of the program resulting in costs incorporating a range of actions not just water quality. The eight benefits are: habitat to improve waterfowl wildlife, carbon sequestration, flood protection, nitrogen removal, species protection, open space, sediment removal, and groundwater recharge (Hansen 2015).
- The US has several programs with the Conservation Reserve Program (CRP) and WRP that both require landholders to maintain wetlands and have a contractual easement over the land. However they have different contract lengths (10, 15 and 30 years), the payment schedule varies and the WRP is a competitive process unlike the CRP (Hansen et al. 2015). Hansen et al. (2015) note that “The WRP is an investment that provides wetland benefits, in most cases indefinitely, whereas the annual CRP payments are more analogous to annual rental payments, and the stream of environmental benefits flowing from the retired land may decline after the contract expires.” This highlights that the

timeframes to achieve long term wetland outcomes and maintain them are an important consideration in program and policy design.

- Under incentive programs, differences in maintenance or monitoring requirements over time can influence the amount of compensation requirements and potential co-benefits (Hansson et al. 2012; Strand and Weisner 2013).
- A review and comparison of wetland incentive schemes in Denmark (1998 to 2021) and Sweden (1986 to 2021) by Graversgaard et al. (2021) indicated that in both countries the average payment ($/ha) required to incentivise voluntary participation in wetland construction/restoration schemes has increased substantially through time (even after allowing for inflation). In Denmark incentive payments increased from 25,000 DKK/ha in 1998 to 117,000 DKK/ha in 2016 and in Sweden (where only partial cover for costs is offered) incentive payments increased from 15,000 SEK/ha in 1989 to 50,000 – 60,000 SEK/ha in 1996 (Graversgaard et al 2021). Graversgaard et al. (2021) also report that between 1998 and 2021 the threshold N removal effectiveness for entry to incentive schemes had to be reduced from 350 kg N ha^{-1} yr^{-1} in 1998 to 90 kg N ha^{-1} yr^{-1} to achieve the desired level of participation (Graversgaard et al. 2021). These outcomes suggest that compensation requirements may have to increase once an initial pool of environmentally motivated farmers and highly effective wetland locations have signed up.

These findings highlight that various factors including timeframes, desired outcomes, biophysical landscape traits, maintenance requirements, project duration and terms with landholders can be critical when designing incentive programs for wetland conservation and restoration. Additionally, these studies highlight that compensation requirements, for example through incentives, may need to increase over time to achieve the desired levels of participation and effectiveness.

Market mechanisms

- Schemes where N credits from wetlands are traded on markets with landholders can result in landholders having different evaluation criteria, and some outcome objectives, that may differ from those of government who are typically the buyers of wetland outcomes via grants or incentive programs (Stephenson et al. 2021). These criteria include implementation costs (construction and maintenance), transaction/contracting costs (number of contracts required and the typical length of contract), regulatory risks (use of third-party contracts), certainty of N compliance (modelled or measured N removal outcomes), and a list of the pollutants reduced by the alternative and qualitative co-benefits (wildlife, aesthetics, flood control). The differences can lead to changes to landholder participation.
- While programs can have multiple benefits, the desired co-benefits such as water, carbon or biodiversity outcomes may be mutually exclusive. For example, managing a wetland for optimal carbon sequestration may limit other wetland uses, such as nutrient offsetting or fishing approaches (Waltham et al. 2016). It has also been demonstrated that while providing co-benefits could increase the overall cost of a project in some instances, securing payments for these additional ecosystem services could help cover the costs of on-ground works (Canning et al. 2023) and reduce the relative cost for water quality improvement. Conversely, Lentz et al. (2014) found in Illinois that for corn farmers installing wetlands, stacking benefits from a wetland N removal program in a market-based trading scheme may or may not satisfy additionality of wildlife as this would occur without payment for wildlife outcome. This highlights the importance of determining the mechanism and intended outcomes for the wetland during the planning and design phase.

Program costs for market mechanisms such as trading and offset programs are difficult to assess as they are poorly reported in Australia across many environmental issues. The US EPA has a very comprehensive system for reporting trading and offset program costs which are separate to the credit or offset. The available information suggests that program costs borne by the public administrator range from 10% to 80%. A number of examples are provided in Table 17. This highlights the need to consider the type of program (or approach) to suit the objectives to ensure that cost-effective outcomes are delivered, and the importance of capturing program costs to allow comparison among programs. In the GBR context, Coggan et al. (2024) report this as a major limitation to assessing the effectiveness of water quality investment programs for the GBR over the last 10 to 15 years.

Table 17. Summary of costs to administer nutrient trading and offset programs. Source: US EPA Compilation of Cost Data Associated with the Impacts of Nutrient Pollution (2015).

Name (Active or Closed)	Type of program	Nutrient(s) involved	Description of costs (Reported by EPA as US $ 2012. Indexed and converted to AUD 2024)	Number of projects that generated pollutant reductions that were offset or traded
Chatfield Reservoir Trading Program (Active)	Trading	Phosphorus	A $254 application fee to cover administrative costs is required for point sources to apply for increased discharge through trading. Credits that enter the pool are sold at a price that reflects the cost of non-point source reduction projects, costs associated with the pooling program, and costs incurred by the Authority to administer the trading program. Exact costs are unknown, but the monitoring program was estimated to cost $147,757/year.	9
Cherry Creek Basin (Closed)	Trading	Phosphorus	Coming from a combination of property taxes and user fees, the budget for 2003 was US$1.7 million (AUD $4.43 million), of which at least 60% had to be spent on the construction and maintenance of pollution reduction facilities. The remaining 40% was used in research, planning documents, technical reports, and administrative costs. State grants financed a smaller portion of the work, particularly that involving educational campaigns about non-point source pollution and construction of pollution reduction facilities.	4
Rahr Malting Company Permit (Active)	Offset	Nitrogen and phosphorus	During the two-year permitting phase, Rahr spent ~$41,658 ($30, 410 for consultants and $11,456 for staff time), while the Minnesota Pollution Control Agency (MPCA) spent ~$85,703 on staff time. During the implementation phase, Rahr spent ~$5,622 on staff time, the MPCA spent ~$83,317 on staff time, a local citizen's group spent ~$1,875, and non-point sources spent about $1,250 on legal assistance. The total for transaction costs during these two phases was ~$174,126, 81% of which were borne by the MPCA as it designed the overall program structure.	4
Tar-Pamlico Nutrient Reduction Trading Program (Active)	Trading	Nitrogen and phosphorus	The Tar-Pamlico Basin Association gave $379,093 to the State Department of Environmental Management during Phase I to fund a staff position, and the trading ratio includes 10% for administrative costs.	200
Great Miami River Watershed Water Quality Credit Trading Pilot Program (Active)	Trading	Nitrogen and phosphorus	Estimated three-year project cost of $5,063,224 including $820,000 to fund best management practices. The program received in-kind support primarily in the form of water quality monitoring, and the training of soil and water conservation professionals by other organisations.	345

Biophysical features influencing costs

As supported by the evidence in Section 4, the most effective wetland construction and treatment systems are those which have been selected, located and designed based on the components and processes of the landscape and which will effectively result in water quality improvement based on the required characteristics. Hydrology, input water quality including pollutant loads and concentrations, wetland type and topography of the landscape at a paddock and overall catchment scale are important considerations (Byström 1998; Cheng et al. 2020; Djodjic et al. 2022; Hansen et al. 2021; Lowe et al. 1992; Manca et al. 2021; Rodriguez et al. 2011; Singh et al. 2019; van der Valk and Jolly 1992; Wallace et al. 2020; Wegscheidl et al. 2021; Zimmerman et al. 2019). When it comes to costs, generally, cost-efficiency is poor when the constructed wetland area is large or rehabilitation of an area is large and incoming nutrient loads are low, as this generates both a high cost and low nutrient processing (Djodjic et al. 2022; Kavehei et al 2021a).

There are global examples where stacking agronomic and edge of field management practices such as improved timing or reductions to in-field N application, edge of field buffer strips with wetland construction, rehabilitation or treatment systems (bioreactors) resulted in more cost-efficient outcomes than individual measures (Balana et al. 2015; Christianson et al. 2018; Geng et al. 2019; López-Ballesteros et al. 2023). Further to the importance of scale, catchment-collective approaches for edge of field mitigation placement typically become more cost-effective than farm-based approaches when larger nutrient reductions are required (Weeber et al. 2022).

The sought after outcome of reduction of a specific pollutant type or other wetland outcomes is a key driver of the cost of an intervention. The dominant pollutant and/or water quality targets for a site will dictate system type, design and maintenance (Adame et al. 2022; Entry and Gottlieb, 2014; Kavehei et al. 2021a). For example Canning et al. (2021b) explored the impacts of ponded pastures south of Mackay on biodiversity outcomes, highlighting that a one size approach does not fit all wetland systems with different biodiversity outcomes being observed. This is also highlighted further in Adame et al. 2022 where each site had a varied shape, design and actions, demonstrating that a variety of factors can influence pollutant processing as described in Section 4.

The maintenance costs of interventions can also be significant, particularly where soil removal and vegetation re-establishment is required (Entry and Gottlieb 2014). Maintenance costs are likely to be greater for P reduction as P cycles through the system and accumulates in the sediment requiring regular sediment removal, whereas N can be permanently removed through the process of denitrification (Byström 1998, 2000; DESI 2023b). For bioreactors, cost is a function of bioreactor size (i.e., volume), as volume directly relates to residence time (DeBoe et al. 2017). The target residence time is a function of the inflow nitrate concentration and nitrate reduction objective (Wegscheidl et al. 2021). This highlights the importance of the definition of clear program objectives, extended program periods and project design specific to the focal landscape to achieve long-term outcomes, and the influence of this on costs.

Cost-effectiveness

Cost-effectiveness analysis is crucial in developing a strategic program, as it helps to evaluate the efficiency of different management actions in reducing pollutants and achieving environmental outcomes for a given level of funds or resources. Cost-effectiveness is essentially the measure of total costs to pollutant reduction and is generally reported as dollar per tonne. To date, the focus of studies assessing cost-effectiveness of management actions in the GBR has been on the cost of landholders changing management practices to reduce pollutants leaving agricultural lands (East and Star 2010; Van Grieken et al. 2010). However, cost information is also important to help understand barriers to adopting management changes (Rolfe and Gregg 2015), to evaluate different policy mechanisms or designs (Rolfe and Windle 2011) and to assess investment priorities (e.g., Alluvium, 2016; 2019). For these purposes cost-effectiveness has been assessed as the ratio of costs involved to achieve pollution changes.

Current policy frameworks for the GBR do not require consistent assessment of the benefits and costs of a project or program to assess cost-effectiveness with each program instead designing their own monitoring and evaluation, and therefore limiting comparability (Star et al. 2021). In Europe, the Water Framework Directive explicitly provides guidance regarding the application of economic principles, tools, and instruments (Balana et al. 2011; Carvalho et al. 2019; Lam et al. 2011; Martin-Ortega 2012) and in the US, detailed cost-benefit assessments are required under the Clean Water Act (Keiser and Shapiro 2019) for a range of actions including duck hunting, carbon sequestration, species protection, flood protection, greenhouse gas reductions, groundwater recharge, sediment removal and open space. Despite the lack of formal requirements in the GBR policy settings, the use of cost-effectiveness of management has been increasing as an assessment tool for two broad reasons. First, on the supply side, there is increasing availability of detailed data from various trials and management options about the pollutant reductions achieved and the costs involved that have allowed more estimates to be generated. Second, on the demand side, the requirements to meet ambitious pollutant reduction targets with set funding caps have focused greater attention on where activities need to be prioritised.

Information about costs helps to identify where investments are most effective and also identifies viable options for water quality improvements. However, the assessment of costs relating to agricultural water management is complex. Keiser et al. (2019) identifies that estimation of primary costs is difficult, because costs are difficult to apportion, cost signals are distorted because of market power issues, and taxes and regulations distort real costs. Other problems are that it can be difficult to measure physical changes such as pollutant reductions, and natural systems are often stochastic due to climate variability.

Given the limited data on cost-effectiveness of wetland treatment systems presented in the 2022 SCS, the findings are expanded here with the addition of studies that did not meet the 2022 SCS eligibility criteria of being peer reviewed and publicly available. Australian studies, particularly those derived from tropical regions, were supplemented by overseas studies which provide insights for cost drivers and program and policy design (Table 18).

Table 18. Summary of Australian studies relevant to the cost drivers and measured costs of wetland treatment systems.

Study	Cost driver and measured cost
Canning et al. (2023)	Representative landholder scenario of the participants, obtained from the costs incurred in constructing a representative scheme-subsidised lagoon on a medium-sized cane farm. Construction and maintenance costs estimated. Included biodiversity and water quality benefits.
Kavehei et al. (2021a)	Outcomes for constructed treatment wetlands, triangles of vegetated drains and squares of sewage treatment plant wetlands across the Wet Tropics and Mackay Whitsunday regions. Eight constructed wetland sites. Costs assessed include design, project management, and construction, maintenance and repair.
Waltham et al. (2021b)	Investigated the transition of low-lying, marginal sugarcane land to alternative land uses that require lower or no N inputs, such as treatment wetlands and ecosystem service wetlands which provide co-benefits of fish production. The costs assessed were reductions in annuity gross margins and land conversion cost.
Hagger et al. (2022)	Carbon focused study for wetland management, but noted the opportunity costs of landholders and the capacity to stack benefits such as carbon, biodiversity and water quality.
White et al. (2022)	Bioreactor in a blueberry farm, measures inflow and outflow of nutirents. Captured construction cost.
Waltham et al. (2021a)	Investigated the transition of low-lying, marginal sugarcane land to alternative land uses that require lower or no N inputs in the Wet Tropics region. Costs assessed were reductions in annuity gross margins and land conversion cost.
Wegscheidl et al. (2021)	Use of bioreactors on sugarcane farms to remove nutrients. Captured construction costs for different sites.
Pfumayaramba et al. (2020)	Presents actual monitored construction costs of bioreactors for pollutant removal.
DESI (2023b)	Summarises cost considerations for treatment systems.

To assess cost-effectiveness, essentially all the unit costs associated with the pollutant reduction are required to be captured and measured consistently across projects and programs. This requires a well-defined monitoring and evaluation program and reporting framework to capture the range of costs over the life of the project.

5.4 Measured Costs

Project-level costs

Project-level cost is defined as the actual cost incurred at the scale of the individual wetland or treatment system. A treatment system such as bioreactors and constructed wetlands may comprise several distinct smaller treatment units situated near each other in an agricultural farm that are designed to function as an integrated set to deliver the designed water quality treatment service.

A wetland project generally goes through three phases of measured progress and subsequent costs across its lifespan (Figure 5):

1. Pre-construction phase (e.g., conceptualisation, design, planning, landholder engagements, approvals).
2. Construction phase (e.g., earthworks, planting).
3. Post-construction phase (e.g., monitoring, maintenance, repair).

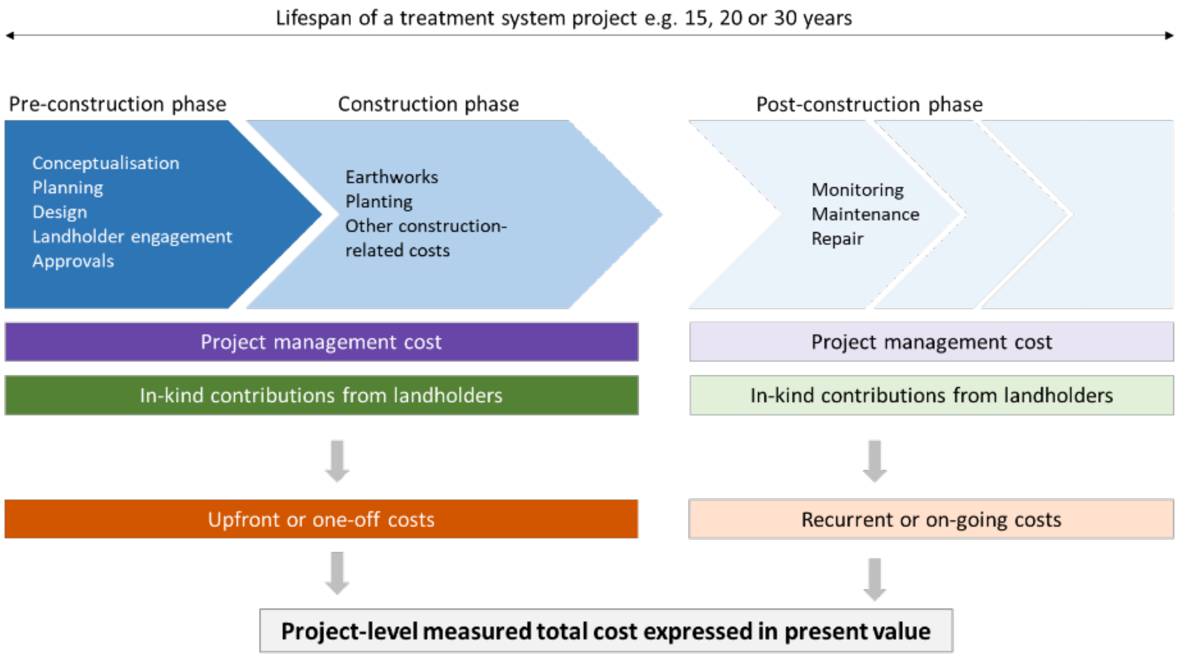

Figure 5. Types of costs over the lifespan of a wetland (Star et al. 2024).

Pre-construction

Wetland establishment and opportunity costs include design costs which involve surveying the site (Collins and Gillies 2014; Douglas-Mankin et al. 2021) and consideration of the hydrology in the context of the required works as they identify the site specific actions and associated costs that will be required to achieve water quality outcomes for the targeted pollutant (Byström 1998; Cheng et al. 2020; Djodjic et al. 2022; Hansen et al. 2021; Lowe et al. 1992; Manca et al. 2021; Rodriguez et al. 2011; Singh et al. 2019; van der Valk and Jolly 1992; Zimmerman et al. 2019).

Costs for restored wetlands involve considerable and semi-irreversible structural work as well as long-term opportunity costs, with a number of studies highlighting the opportunity cost of production as a cost to be considered over the long term (Beukes et al. 2023; Douglas-Mankin et al. 2021; Heberling et al. 2010; Roley et al. 2016). Opportunity cost can be the main cost over time for constructed or rehabilitated wetlands as land area is permanently taken out of production (Ribaudo et al. 2001; Roley et al. 2016).

Yang et al. (2016) considered the transaction costs of administration associated with a project. These transaction costs included site assessment, negotiation, and paperwork, which were distributed over the number of wetlands within one farm. Transaction costs associated with funding applications can be regarded as a significant disincentive from the landholder's perspective (Hansson et al. 2012; Stephenson et al. 2021). Yang et al. (2016) also considered what they termed a nuisance cost, which represent the annual costs associated with inconveniences to agricultural production (e.g., machinery operations) when wetlands are present within farm fields. For many studies, actual *in-kind contributions* are typically included in project costs, as upfront and/or ongoing costs, because such costs are seen as essential drivers for successful completion of wetland projects (Canning et al. 2023; Kavehei et al. 2021a), however given that opportunity costs can dominate the overall costs in the long-term, they must be captured (Roley et al. 2016).

This phase will also ensure that any legal restrictions and existing infrastructure are identified, and the relevant agencies can cooperate if required before construction begins, noting that it may increase the length of the planning and design process and subsequent costs (Byström 1998; Hansen et al. 2021).

Hansen et al. (2015) found that the key cost drivers for incentives to obtain an easement or pre-construction costs were related to:

- Current land values and proximity to urban development.
- Value of the land with an easement for the wetlands.
- The value of the land near the wetland.
- Nuisance cost of landholders.
- Good stewardship.

van der Valk and Jolly (1992) suggest that the major technical issues that need to be resolved before effective and realistic guidelines can be developed for restoring wetlands to reduce non-point source pollution include: 1) the effects of contaminants, particularly sediments and pesticides, on the wetlands; 2) the fate of organic contaminants in the wetlands; 3) the development of site selection criteria; and 4) the development of design criteria. There are also many social, economic, and political barriers to implementing restored wetlands which are highlighted Section 5.5.

Construction

Construction costs are based on the design aspects. They typically include soil and land conditioning, earth works such as use of excavator embankment, construction and engineering of water-flow structures (Comín et al. 2014; Kavehei et al. 2021a). These costs also include the planting of wetland specific plants (Aggarwal et al. 2022; Collins and Gillies 2014; Comín et al. 2014). If land is acquired for construction, then the cost for the acquisition needs to be captured in the construction costs. If this is the case, the opportunity costs for the landholder of not producing off this land is negligible (Christianson et al. 2013). These costs generally occur in the first year of analysis for cost-effective studies and therefore are not impacted by discounting over time.

Construction costs will vary depending on the type of wetland, and design aspects will vary between sites based on hydrology and biophysical features. These costs will also vary based on the location or access to heavy machinery, and or materials such as rocks or woodchips (Weeber et al. 2022).

Post-construction

Following completion of on-ground wetland construction, recurrent or ongoing costs are incurred annually or periodically until the end of the project lifespan. Ongoing costs include monitoring and evaluation costs, operating and maintenance costs and repair costs.

Monitoring and evaluation costs (annual): Monitoring and evaluation activities are necessary to ensure targeted pollutants are being reduced and are effective either in terms of wetland function (in the case of wetland restoration) or wetland extent and condition (in the case of new wetland installations) (Douglas-Mankin et al. 2021; Mewes 2012; Strand and Weisner 2013). If co-benefits are sought, these also need to be monitored to ensure that these outcomes are being realised (Strand and Weisner 2013).

Operating and maintenance costs (annual or periodical): Operating and maintenance works are undertaken to ensure that the restored or constructed wetlands remain effective at delivering water quality outcomes. These works may also include the areas close to the wetland such as buffer strips, or the actual wetlands (e.g., annual weed removal, mowing, periodical dredging of accumulated sediments) (Douglas-Mankin et al. 2021; Getahun and Keefer 2016; Soldo et al. 2022; Tamburini et al. 2020).

Without a long-term maintenance plan and a mechanism to fund ongoing works, rehabilitation sites have a high chance of returning to a degraded state. For example, the US Wetland Program highlights that long-term opportunity costs and ongoing maintenance costs must also be considered with some programs allocating 50 years ahead of time. This is important in the context of the wetland and treatment system and design, and the different time periods over which the wetlands are resourced to be managed and maintained. The cost of maintenance is also an important consideration in defining the minimum time of a project, as well as resourcing and monitoring requirements, which potentially (and most likely) extend well beyond the life of the initial funding program.

Potential disbenefits of actions should also be considered, for example, consideration that management practices can generate unintended negative impacts on landholders such as the introduction or attraction of invasive species (e.g., feral pigs) in buffer strips or difficulty in headland management. There are many examples of this occurring globally (e.g., Entry and Gottlieb 2014; Getahun and Keefer 2016; Hansen et al. 2021; Rao et al. 2012; Ribaudo et al. 2001; Sarris and Burbery 2018) and also within the GBR.

Waltham and Canning (2021) explored this further in floodplain wetlands within the Burdekin region (Sheep Station Creek), highlighting the need for ongoing spot spraying maintenance of aquatic weeds. The study demonstrated that investment in aquatic weed removal and ongoing maintenance was critical, not only for improved water quality but also survival of fish reaching the restored waterway. A funding model similar to the Sheep Station Creek maintenance program, which has continued for almost 20 years, is probably the best example of the type of long-term maintenance necessary on the floodplain (Waltham 2021). These costs obviously vary within the lifetime of the wetland, how it was designed and how it is managed in the landscape. This also highlights the requirement for ongoing monitoring, evaluation and adjustment of projects and programs.

Looking nationally, Firn et al. (2013) highlighted the costs of managing weeds at Lake Eyre over a 50-year period. The weed management program involved several strategies including a prevention and monitoring program which cost an estimated $9 million over the 50 years. The costs ranged from $1.7 billion required to manage all weeds and protect the values of the system to $113 million to only manage Weeds of National Significance over 50 years. This highlights the importance of considering the maintenance costs over the long-term.

Repair costs (as and when needed): Repair costs may be incurred post-wetland construction to account for unforeseen circumstances (e.g., mechanical failure, design oversight, flood damage) that have compromised wetland condition and function and may involve minor or major repair works (e.g., revegetation, apron re-shaping). These were poorly documented in the literature which may reflect the temperate or alpine nature of international studies, or the length of the study.

Timeframe and discount rate

The sum of upfront wetland construction cost, the ongoing monitoring, evaluation, operating and maintenance costs (discounted), and repair cost (discounted) constitutes the actual measured costs of wetlands, expressed in present value (in $). The timeframes that wetland cost assessments were completed on also varied with some considering costs over 50 years (Christianson et al. 2013), 40 years (Strand and Weisner, 2013), 30 years (Zammali et al. 2021) and others 15 and 10 years (Roley et al. 2016; Yuan et al. 2022). The timeframe is dependent on the type of wetland and treatment systems being assessed: for bioreactors, reflecting the life span; for buffer strips or best management practice approaches, reflecting management integration; and for natural or near-natural wetlands, reflecting the ongoing management that would be required for having a wetland in the landscape.

Discount rates also varied between studies (Canning et al. 2023; Christianson et al. 2018; Collins and Gillies 2014; Kavehei et al. 2021a) in terms of the time period, from a 10-year analysis for a bioreactor with a

10.6% discount rate through to a 40-year timeframe for a bioreactor and management and a 4% discount rate (Christianson et al. 2013). The total present value of measured costs is then multiplied by the inverse of the annuity factor to arrive at the annualised present value of measured costs (in $/year) (Canning et al. 2023; Kavehei et al. 2021a).

Reported costs and cost-effectiveness

No studies in Australia have captured all types of costs across the pre-construction, construction and post-construction phase, partly because implementation of such projects is relatively new. Introducing a consistent standard will facilitate comparable reviews over time as more projects are implemented (Aklilu and Elofsson 2022; Graversgaard et al. 2021; Strand and Weisner 2013).

Given there is limited data relating to the conditions of the GBR individual sites, costs must be considered with caution. Table 19 captures the actual costs reported in relevant Australian studies. These are difficult to compare as the studies have different approaches to capturing pre-construction, construction and post construction phase costs, and in some instances, the costs are not separated. Wallace et al. (2020) recognises these limitations by providing ranges to address the uncertainty and limited information. The data in Table 19 highlights the different types of costs that have been captured, but the difference in the way costs are measured means they are not directly comparable.

Further to this, the type of wetland or treatment system along with the location, different time periods of analysis, and discount rates that have been applied will influence the costs each study has reported as $/ha/year. Kavehei et al. (2021a) studied eight constructed wetland systems in the Wet Tropics and Mackay Whitsunday regions (Table 20). The study reported consistent cost metrics across wetland sites, each with different wetland designs in different contexts, land uses and size. All but one site applied the same discount rate and timeframe and reported both the annualised cost per hectare and the total cost per hectare. This allows policy to consider investment options over different time periods.

International projects in Denmark and Sweden have demonstrated that the average costs of N abatement typically increase over time (after correcting for inflation) as the number of willing participants and effective wetland locations is exhausted (Aklilu and Elofsson 2022; Graversgaard et al. 2021; Strand and Weisner 2013). Therefore, initial landscape design accounting for landscape processes, hydrology and topography is critical to achieving cost-effective outcomes (Cheng et al. 2020; DeBoe et al. 2017; Hassett and Steinman 2022; Roley et al. 2016). If the wetland is also to achieve other co-benefits, then these must be identified at the design phase of the project (Canning et al. 2022; Hagger et al. 2022).

For the US-based studies, the range of costs is reflected by the variance in costs collected, with some programs allocating percentages across the 50-year life of the wetland program, and others assessing the cost per ‘Average Annual Habitat Unit’ (Table 21). These programs highlight the complexity of assessing costs and the importance of capturing specific aspects, recommending minimum reporting measures for assessing and comparing annual N removal performance. For example, for bioreactors, these measures include bioreactor dimensions and installation date; fill media size, porosity, and type; nitrate-N concentrations and water temperatures; bioreactor flow treatment details; basic drainage system and bioreactor design characteristics; and N removal rate and efficiency.

Table 19. High-level summary of Queensland studies with reported cost types and values. Cost values are reported as annualised present value cost (APVC, in AU$ ha^{-1} yr^{-1}) and its equivalent total present value costs (TPVC, in $ ha^{-1}).*

Study	Treatment system(s)	Region	Types of upfront measured cost included	Types of ongoing measured costs included	Opportunity cost of production	Time frame(s) (years)	Discount rate(s) (% per annum)	Cost value
Kavehei et al. (2021a)	Eight constructed treatment wetlands§	Wet Tropics and Mackay Whitsunday	Design, project management, and construction	Maintenance and repair	Not included	15, 20, 25	3, 5, 7	Reported APVC at 5% discount rate over 20 years ranged between AU$4,197 ha^{-1} yr^{-1} (TPVC at AU$52,304 ha^{-1}) and AU$43,076 ha^{-1} yr^{-1} (TPVC at AU$536,824 ha^{-1}) with mean and median costs of AU$11,886 ha^{-1} yr^{-1} (TPVC at AU$148,125 ha^{-1}) and AU$7,789 ha^{-1} yr^{-1} (TPVC at AU$97,064 ha^{-1}), respectively. (AU$ expressed in FY2020/21).
Canning et al. (2023)	Constructed lagoon	Wet Tropics, Tully-Murray	Construction	Maintenance	Opportunity cost net of yield improvement and higher value alternative land use across a representative farm	15	5	For a representative 0.3 ha lagoon, total (upfront) construction cost was AU$39,867, of which AU$9,867 (~25%) was contributed by landholders as in-kind and cash contributions. Maintenance costs ranged between AU$99 yr^{-1} and AU$990 yr^{-1}. (AU$ expressed in year 2019). On a per hectare basis and expressing costs in AU$ in 2020, total construction costs were AU$134,015 ha^{-1} and the maintenance cost was between AU$333 ha^{-1} yr^{-1} and $3,328 ha^{-1} yr^{-1}.
Pfumayaramba et al. (2020)	Denitrifying bioreactor‡	Lower Burdekin	Construction	Ongoing measured costs not available	No	N/A	N/A	Total construction cost for 34 m^3 bioreactor was AU$20,951. (AU$ expressed in year 2020). Construction cost comprised fixed cost component (AU$11,600 per bioreactor) and variable cost component (AU$275.04 per m^3, AU$9,351 for 34 m^3).
Alluvium (2016)	Constructed wetlands	Wet Tropics	Construction	Ongoing measured costs not available	Not included	10	7	An assessment of the installation costs for 25, 50 and 100 ha of constructed wetlands or recycle pits in sugarcane growing areas. • Installation cost for small wetlands:

Study	Treatment system(s)	Region	Types of upfront measured cost included	Types of ongoing measured costs included	Opportunity cost of production	Time frame(s) (years)	Discount rate(s) (% per annum)	Cost value
								\$800,000/ha (low), \$900,000/ha (medium) and \$1,000,000. • Construction costs per ha [Small wetlands]: \$900,000/ha. • Construction costs per ha [Medium/Large wetlands]: \$343,913/ha. • Installation cost for medium to large wetlands: \$275,130/ha (low), \$343,913/ha (medium), and \$412,696/ha in addition to establishment cost of \$738,607.
Wallace et al. (2020)	Constructed treatment wetland	Wet Tropics	Construction, Post-construction	Maintenance	Not included	12, 16	5	The range of construction costs for wetlands constructed on-farm for nutrient management in this study is \$60,000 to \$900,000 per hectare. The range of annual maintenance costs for constructed treatment wetlands is assumed to be between \$1,800 ha^{-1} yr^{-1} and \$27,000 ha^{-1} yr^{-1}.
Waltham et al. (2016)	Wetland restoration	Wet Tropics	Post-construction	Maintenance	Included as lost production; one off capital cost of land acquisition not included	12, 16, 20	5	Maintenance costs for restored coastal wetlands: \$750 ha^{-1} yr^{-1}.

§ Vegetated drains are not included because the construction, maintenance and repair costs are not measured actual costs i.e., they were derived and/or estimated from other studies.

Ŧ Costs of two denitrifying bioreactors were provided: 34 m^3 trialled bioreactor bed and 100 m^3 hypothetical bioreactor; only the cost for the actual 34 m^3 bioreactor is reported in this table.

*Costs reported here are expected to be higher due to the supply chain issues and increased costs that occurred post Covid-19 pandemic.

Table 20. Reported cost types and measured cost values for individual constructed treatment wetlands (CW) as reported in Kavehei et al. (2021a). Cost values are reported as annualised present value cost (APVC, in AU$ ha^{-1} yr^{-1}) and its equivalent total present value costs (TPVC, in $ ha^{-1}).# The range of APVCs is based on combinations of timeframe and discount rates used in the study; corresponding TPVCs are calculated by dividing APVC by the annuity factor. Measured costs for CW8 wetland are reported for a 20-year timeframe at 5% per annum discount rate because APVC for this wetland is not provided for other timeframes and discount rates.

Treatment system	Size (ha)	Region	Characteristics	Cost value (in FY2020/21 AU$)#
CW1	1.6	Wet Tropics	Groundwater dominated, converted drain on a sugarcane farm. Very high length-to-width ratio. Vegetation cover is > 50%.	$APVC_{25yrs\_3\%}$ = 6,369 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 11,269 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 110,901 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 102,633 AU$ ha^{-1}
CW2	1.2	Wet Tropics	A square-shaped wetland on a banana farm with a sediment basin at the inlet. Low length-to-width ratio. Vegetation cover is < 25%.	$APVC_{25yrs\_3\%}$ = 31,588 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 57,665 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 550,046 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 525,204 AU$ ha^{-1}
CW3	8.5	Wet Tropics	A large landscape wetland with two inlet points draining sugarcane farms. Vegetation cover is > 50%.	$APVC_{25yrs\_3\%}$ = 3,075 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 5,629 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 53,538 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 51,273 AU$ ha^{-1}
CW4	10	Wet Tropics	Sugarcane paddock converted to wetland. Water level is regulated via manually operated gates. Vegetation cover is > 50%.	$APVC_{25yrs\_3\%}$ = 7,874 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 12,915 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 137,115 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 117,632 AU$ ha^{-1}
CW5	2.5	Wet Tropics	Treatment system draining a banana farm. Designed for a retention time of two days. Very high length-to-width ratio. Vegetation cover is < 25%.	$APVC_{25yrs\_3\%}$ = 5,244 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 8,705 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 91,306 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 79,283 AU$ ha^{-1}
CW6	1.8	Dry Tropics, Mackay Whitsunday	A square-shaped wetland draining sugarcane farm. Low length-to-width ratio. Comprised of two internal berms to increase residence time, a sediment basin, two inlets and an outlet wall. Vegetation cover is > 50%.	$APVC_{25yrs\_3\%}$ = 5,232 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 8,928 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 91,112 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 81,312 AU$ ha^{-1}
CW7	2.1	Dry Tropics, Mackay Whitsunday	A treatment train system draining sugarcane land comprising multiple ponds. High length-to-width ratio. Vegetation cover is between 25–50%.	$APVC_{25yrs\_3\%}$ = 5,775 AU$ ha^{-1} yr^{-1} $APVC_{15yrs\_7\%}$ = 9,997 AU$ ha^{-1} yr^{-1} $TPVC_{25yrs\_3\%}$ = 100,555 AU$ ha^{-1} $TPVC_{15yrs\_7\%}$ = 91,054 AU$ ha^{-1}
CW8	1.3	Dry Tropics, Mackay Whitsunday	A treatment train system draining sugarcane land comprising multiple ponds. High length-to-width ratio. Vegetation cover is between 25–50%.	APVC AU$ ha^{-1} yr^{-1}: 7,963 (20 years at 5%) TPVC AU$ ha^{-1}: 99,237 (20 years at 5%)

Table 21. Examples of costs for wetland restoration and bioreactors in the US.

Study	Treatment system(s)	Region	Types of upfront measured cost included	Types of ongoing measured costs included	Opportunity cost of production	Time frame(s) (years)	Discount rate(s) (% per annum)	Cost value
Aust (2006)	Wetland restoration	Louisiana	Pre-cost: engineering design, easements and land rights, federal supervision and administration, project management and inspection.	25% contingency costs. These costs are added to the monitoring costs and the operation and maintenance Post project: Maintenance costs over the total life of the project.	N/A	20	Not reported	Expressed as $/ Average Annual Habitat Units (AAHU). Range of $159–$79,631 (2006 indexed and converted to 2024 AU$).
Hansen et al. (2015)	Wetland restoration	US review	All cost covered by the contracts (specifics not detailed).		No	87% of the WRP contract acres are permanent easements, 6% are 30-year easements, and the remainder are 10-year agree-ments. In contrast, CRP wetland-related contracts range from 10 to 15 years (about 25% are for 10 years, 75% are for 14 or 15 years).	Not reported	$170 per acre (AU$816.58 per ha indexed and converted) in western Dakota, Montana, Arkansas and Louisiana $6,100 per acre (AU$29,308.17 per ha indexed and converted) in the major corn producing areas such as Iowa.
McMann et al. (2017)	Wetland restoration coastal	Baton Rouge, US	Varied across studies.	Proportional allocation of pre-construction, construction, post-			Not reported	Allocations, not $ values given however:

Study	Treatment system(s)	Region	Types of upfront measured cost included	Types of ongoing measured costs included	Opportunity cost of production	Time frame(s) (years)	Discount rate(s) (% per annum)	Cost value
				construction and contingencies.				Pre-construction: 10% of construction cost, 15–20% for largest sized wetlands. Construction: timelines based on types of categories defined. Contingencies: 20% Post-construction: Operating & maintenance based on a 50-year life.
Christianson et al. (2018)	Bioreactor Review	International	Varied across studies.				Not reported	Bioreactor bed installation costs have ranged from $$15,479 to $20,377 in South Dakota to more than $39,187 in the Chesapeake Bay watershed. (Indexed and adjusted to AU$).

5.5 Improving landholder participation in wetland management

A central component of wetland restoration and rehabilitation is landholder participation in programs, and adoption of management practices and activities that support wetland health and promote removal of pollutants. This section explores several key themes that are likely to influence the success of wetland management programs: the adoption of MBIs, the influence of regulation, the role of social norms, and the importance of programme design in incentivising landholder participation.

Landholder participation in GBR water quality programs can be influenced by the introduction of MBIs such as reverse tenders, specifically aimed at reducing DIN in sugarcane catchments (Eberhard et al. 2021). While these mechanisms can be promising, landholder adoption of the Paddock to Reef (P2R) Water Quality Risk Framework practices has been relatively low, primarily due to factors such as low private benefits, perceived risks to profitability, and climate variability (Gregg and Rolfe 2017; Star et al. 2011, 2019).

The Queensland Government has established regulated minimum standards for agricultural practices in sugarcane, grazing, and horticulture within the GBR catchments. However, regulation alone has not been enough to drive widespread adoption beyond the defined minimum standard, further underscoring the need for complementary incentives and flexible program designs that cater to landholders' diverse needs and circumstances (Coggan et al. 2024).

Social norms and landholder preferences play a critical role in the adoption of wetland conservation practices. Research on graziers in Northern Australia highlights how contract attributes such as conservation actions, payments, and flexible provisions influence participation (Greiner 2023). Furthermore, landholders' aesthetic and practical preferences for land use, such as preferences for tree planting or concerns about long-term contractual commitments, can also impact program uptake (Patrick et al. 2009; Sherren et al. 2012).

Program design and engagement terms are crucial, particularly given the long-term nature of wetland restoration and rehabilitation. Studies from Australia and other global contexts indicate that transaction costs, program flexibility, and secondary benefits like hunting opportunities or aesthetics significantly influence landholder participation (Hansson et al. 2012; Soldo et al. 2022). The availability of suitable land and existing laws can also limit the feasibility of large-scale wetland construction (Byström 2000).

For further government investment aimed at improving water quality specifically from wetlands in the GBR, it is crucial to enhance landholder participation through well-designed programs that address both economic and social drivers. Maximising return on investment requires integrating MBIs with other co-benefits and adapting programs to meet the diverse preferences and concerns of landholders, ensuring long-term engagement and sustainability of wetland restoration efforts.

5.6 Summary of policy, cost and investment considerations

The Australian and Queensland governments have implemented a range of policies and programs to manage wetlands within the GBR catchment, emphasising their ecological importance and the need for sustainable management practices. These initiatives, including the Queensland Wetlands Program and the *Reef 2050 Wetlands Strateg*y, a joint commitment between the Australian and Queensland governments, provide direction for wetland science, planning, coordination and management in the GBR and its catchments. These programs, supported by tools like Wetland*Info*, enable better decision-making and resource allocation for wetland management.

Wetland restoration and management projects can come with significant costs, influenced by factors such as hydrology, location, and design requirements. Understanding these cost drivers is crucial to designing effective and sustainable wetland projects, particularly with regard to incentivising landholder participation and ensuring long-term sustainability. Market-based instruments, such as the Australian Carbon Credit Units, offer financial incentives for landholders to engage in wetland restoration. Options for credits for water quality improvement activities are currently being developed. While these programs highlight the potential for co-benefits like carbon sequestration and biodiversity which can help offset costs, further research is required to quantify the benefits.

The costs of wetland restoration projects vary significantly depending on the project's scope and complexity. Factors such as upfront construction, maintenance, and opportunity costs play a key role in determining overall cost-effectiveness. To ensure long-term project success, it is important to implement well-designed incentive programs that not only meet water quality goals but also account for these varied cost drivers, enabling broader landholder participation and more sustainable outcomes.

Table 22 presents a summary of the key findings identified in this review for policy and project design of wetlands for water quality improvement in the GBR.

Table 22. Key considerations for the policy and project design of wetland programs for water quality outcomes in the GBR.

Key findings	Relevant studies	Application to GBR context	Key cost driver or cost implications in the GBR context
POLICY MECHANISM			
Policy mechanism is a key driver of cost and must consider the long-term nature of wetlands restoration such as agreements over time periods (e.g., 10, 15 and 30 years, noting LRF is 10 years), investments in incentives for remediation, and the implications once these programs change or finished. In studies from GBR catchments the range of costs has been between \$5,629 and \$57,665 AU\$ ha^{-1} yr^{-1} over 15 years at 7%. US incentive programs have considered the cost break down as the following. Pre-construction: 10% of construction cost, 15–20% for largest sized wetlands Construction: timelines based on types of categories defined. Contingencies: 20% Post-construction: Operating and maintenance based on a 50-year life.	Hansen et al. 2015; Kavehei et al. 2021a	Consideration of mechanism and outcome at the end of the program along with the capacity to fund long-term projects.	In the GBR context there have not been project funding commitments beyond five years. LRF offers 5 to 15-year contracts which provides scope for repairs and maintenance. Consideration of costs over long term (50-year) time scales to fully cost the project should then be considered in the initial project funding stages. A long-term project would require significantly higher funding to ensure long term outcomes. If the policy is going to be a market mechanism there will potentially be a higher program cost that must be factored into the program.
The policy time frame, location, contractual obligations, payment schedule and competitive nature of grants are critical to consider for long term outcomes.	Byström 2000; Hansson et al. 2012; Soldo et al. 2022; Trenholm et al. 2017	Consideration of landscape productivity, interaction with agriculture and legal requirements.	Policy design across the landscape considering interaction with agriculture and existing legislation. Along with the timeframes, project contractual obligations and grant allocation are critical to overall costs and long-term outcomes for wetlands.
Where long-term wetland programs have taken place, the average payment (\$ ha^{-1}) required to incentivise voluntary participation in wetland	Graversgaard et al. 2021	Ensure that the allocation of funds in the initial stages is considered in the	The program costs will increase over time so early engagement is critical.

Key findings	Relevant studies	Application to GBR context	Key cost driver or cost implications in the GBR context
construction/restoration schemes has increased substantially through time (even after allowing for inflation).		context of what the future costs may be.	
Although providing co-benefits could increase the overall cost in some instances, securing payments for these additional ecosystem services could help cover the costs of on-ground works and reduce the relative cost for water quality improvement. If the wetland is to achieve other co-benefits, then these must be identified at the design phase of the project. Conversely, stacking benefits in a market-based trading scheme may or may not satisfy additionality. This highlights the importance of determining the mechanism and intent for the wetland outcomes from the planning and design phase.	Canning et al. 2023; Hagger et al. 2022; Hansson et al. 2012; Lentz et al. 2014; Strand and Weisner 2013	Set clear objectives for outcomes and if there are objectives other than water quality. Consideration of combinations of LRF and other environmental and water quality focused programs and what constitutes the level of additionality for stacking.	If additionality is required, the co-benefit and quantity are to be specified before project funding call is made along with specific additional actions that may be required to achieve these additional benefits.
PROJECT COST EFFECTIVENESS			
Initial landscape design accounting for landscape processes, hydrology and topography is critical to achieving cost-effective outcomes. Catchment-collective approaches for edge of field mitigation placement become more cost effective than farm-based approaches when larger nutrient reductions are required.	Cheng et al. 2020; DeBoe et al. 2017; Hassett and Steinman 2022; Roley et al. 2016; Weeber et al. 2022	Consideration of multiple landscape layers, consideration of treatment train processes and catchment area.	Any legal restrictions and existing infrastructure are identified, and the relevant agencies can cooperate. Scale and multi-land use complexity in positioning wetlands in the landscape may increase time to implement but can lead to improved pollutant reductions and cost-effectiveness.
Stacking agronomic and edge of field management practices such as improved timing or reductions to in-field N application, edge of field buffer strips with wetland construction, rehabilitation or treatment systems (Bioreactors) resulted in more cost-efficient outcomes than individual measures.	Balana et al. 2015; Christianson et al. 2018; Geng et al. 2019; López-Ballesteros et al. 2023	Consideration of systems changes across properties to achieve scale through a suite of management actions.	At a property level consideration of a suite of management actions to manage the wetland and the water that enters the wetland with a number of actions across a whole property increasing the cost-effectiveness.
The sought after outcome of reduction of specific pollutant type or other wetland outcomes is a key driver of cost. The outcome will dictate system type, design and maintenance. Canning et al. (2021b) explored the impacts of ponded pastures south of Mackay on biodiversity outcomes and highlighted that a one size approach does not fit all wetland types.	Canning et al. 2021b; Entry and Gottlieb 2014	Project selection may take specialist advice to help fund the most effective approach in the wetland and to ensure that the actions are relevant	Ensure a technical advisory panel is formed to support the selection and design of the proposed wetland projects. This will ensure that for the given location and proposed project there are not further benefits to be

Key findings	Relevant studies	Application to GBR context	Key cost driver or cost implications in the GBR context
		for that particular site.	obtained with slight changes to the design.
The investment in aquatic weed removal and ongoing maintenance is critical for weeds and pest species.	Waltham and Canning 2021; Waltham 2021	Account for weed and feral species maintenance costs in the design phase and consider in project selection the total cost of the project over time.	The Sheep Station Creek maintenance program, which has continued for almost 20 years, provides an example of long-term maintenance for floodplain wetlands.
Maintenance costs are likely to be greater for P reduction as P cycles through the system and accumulates in the sediment requiring regular sediment removal, whereas N can be permanently removed through the process of denitrification.	Byström, 1998, 2000; DESI 2023b	Consideration of the pollutant for reduction and how it reacts in a wetland environment and requirements for removal.	Assessment for projects requires the technical panel to consider the long-term costs for the relevant pollutant removal.

6. GBR evidence base, knowledge gaps and future work

6.1 Characteristics of the GBR evidence base

The global review referenced 238 tropical and subtropical studies in agricultural areas (Waltham et al. 2024b) and 145 studies in non-agricultural areas (Thorburn et al. 2024). In contrast, the GBR specific evidence base is currently limited to 17 studies (published papers and reports listed in Appendix 1: Great Barrier Reef studies included in the review). Most studies in the Great Barrier Reef catchments are recent, with 80% published after 2019, and data collection spanning back to 2017. The few exceptions include McJannet et al. (2012), McKergow et al. (2004) and Alluvium (2016). The synthesis of evidence in the 2017 SCS (Eberhard et al. 2017) also provides an important summary of the knowledge as at 2016 which has been built on in this review.

The majority of GBR studies examine nitrogen (N) removal or N-related processes. Only two studies report sediment removal efficiencies and just one addresses pesticide removal (ametryn). Spatially, most studies are from the Wet and Dry Tropics, with fewer in the Mackay Whitsunday region, a single study from the Burnett Mary region and none from Cape York or the Fitzroy regions.

As discussed in Section 3, many factors influence water quality improvement. To understand how well GBR studies capture these factors, a comprehensive summary of parameters that influence wetland efficacy in removing N, sediments or pesticides was compiled. A simplified table (Table 23) highlights which parameters each GBR study measured or reported. To assess the level of confidence in their monitoring approaches, studies were evaluated against the DES Monitoring Level for Vegetated Drains and Treatment wetlands framework (Manca and Wegscheidl 2024).

Key parameters considered essential for understanding wetland functionality include water flow dynamics, benthic sediment characteristics, N concentration in the water column, vegetation type and density, wetland size and depth, and the relationship between wetlands and catchments. The table further outlines the extent to which each study addresses how these factors influence wetland performance (colour coded).

The concentration of different forms of N (e.g., NO_x-N) in the water column is recognised as crucial in numerous N-related processes. Half of the studies report concentration of N species within the wetland water column.

Conversely, a significant portion of studies conducted in bioreactors lack information concerning water quality in the water column. Carbon concentrations in benthic sediments, deemed a critical parameter for denitrification, is only documented in six studies. All studies contain insights into vegetation types, with the exception of bioreactors that do not have vegetation, yet half do not report vegetation density.

While the inflow and outflow were reported from many studies, this information was missing from several studies particularly those studies measuring processes for nutrient removal such as denitrification in natural wetlands. The absence of this information hinders an accurate assessment of the wetland's actual potential in processing pollutants. Similarly, groundwater interactions, important for hydrology and nutrient dynamics, are rarely considered. While it is recognised that the studies may have had different objectives, it is still relevant to recognise these limitations when seeking to apply the results in the assessment of wetland efficacy.

Hydraulic retention time (HRT) plays a fundamental role in wetland function and is intricately linked to understanding the efficacy value of wetlands. More than half of the studies reported the HRT, but most bioreactor studies do not.

Wetland size and depth was generally provided by studies, and eight provided details on the ratio between wetland size and area of the catchment drained. Similar to the information on water flow, this ratio is important for understanding pollutant load relative to treatment capacity.

Land use is usually reported, though the level of detail varies. Land use is related to the type and concentration of pollutants and it is therefore an important parameter to consider when investigating wetland efficacy. All studies (except Navaratna et al. (2012), a laboratory study) presented at least one climate parameter (temperature or/and air temperature), and few studies also reported evaporation and/or evapotranspiration.

Among the studies measuring wetland efficacy and N processes, few studies considered costs. Other studies that measured costs, but not necessarily wetland efficacy in the GBR, were outlined in Section 5.

Table 23. Summary of the characteristics of the studies conducted in the GBR related to wetland efficacy in removing N, sediments or pesticides. The colour coding shown in the Legend represents the coverage of factors related to each characteristic. Green indicates coverage of the most significant parameters, yellow indicates moderate coverage and red indicates that the data is not provided.

Legend	
Parameter – Data availability	
Water column water quality (WQ)	
Temperature	
Electrical conductivity and pH or Redox potential	
Dissolved oxygen (DO)	
Denitrification species	
Benthic - Sediment	
Soil content carbon	
Vegetation	
Type of vegetation	
Type of vegetation and density	
Wetland configuration	
Size	
Size and depth	
Climate	
Rainfall	
Air temperature	
Rainfall, temperature & evaporation/evapotranspiration	
Inflow/ Outflow Surface Water **Inflow - Outflow nutrients/ TSS/ Pesticides** **Groundwater** **Hydraulic retention time** **Land use** **Ratio wetland: catchment** **Costs**	
Data provided	
Data not provided	

Author (year)	Wetland Type	Region	Study period	Pollutant	Water column - WQ	Benthic sediment	Wetland vegetation	Inflow/ outflow surface water	Inflow - outflow nutrients/ TSS/ Pesticides	Groundwater	Hydraulic retention time	Wetland configuration	Land use	Ratio wetland: catchment	Climate	Costs
Efficacy Literature																
Kavehei et al. (2021a)	Constructed wetland	Wet Tropics, Dry Tropics and Mackay Whitsunday	2015 to 2021	Nitrogen												
Wallace et al. (2022)	Constructed wetland	Mackay Whitsunday	2018 to 2020	Nitrogen												
Mcjannet et al. (2012)	Riverine wetland - natural	Wet Tropics	2007 to 2010	Nitrogen												
Adame et al. (2019)	Natural - few different habitats	Wet Tropics	2018	Nitrogen												
Manca et al. (2021)	Bioreactor	Wet and Dry Tropics	2018 to 2021	Nitrogen		N/A	N/A									
McKergow et al. (2004)	Treatment - Riparian buffer	Wet Tropics	1996 to 1999	Nitrogen & Sediment	N/A											
Navaratna et al. (2012)	Bioreactor	N/A – Lab-scale	N/A	Pesticide -Ametryn	N/A	N/A	N/A			N/A			N/A	N/A	N/A	
Rafiei et al. (2022)	Hypothetical - modelling	Wet Tropics	2012 to 2017	Nitrogen	N/A	N/A	N/A									
Kavehei et al. (2021b)	Wetland treatment systems	Wet Tropics	2019 to 2020	Nitrogen												
Wallace & Waltham (2021)	Constructed wetland	Wet Tropics	2017	Nitrogen & Sediment												

Author (year)	Wetland Type	Region	Study period	Pollutant	Water column - WQ	Benthic sediment	Wetland vegetation	Inflow/ outflow surface water	Inflow - outflow nutrients/ TSS/ Pesticides	Groundwater	Hydraulic retention time	Wetland configuration	Land use	Ratio wetland: catchment	Climate	Costs
Alexander et al. (2023)	Bioreactor	Wet Tropics	2018 to 2019	Nitrogen			N/A									
Alluvium (2016)	Constructed wetlands	Wet and Dry Tropics, and Burnett Mary	2016	Nitrogen												
Wegscheidl et al. (2021)	Bioreactor	Dry Tropics	2018 to 2020	Nitrogen			N/A									
Wegscheidl et al. (2021)	Bioreactor	Wet Tropics	2019 to 2021	Nitrogen												
Wegscheidl et al. (2021)	Bioreactor	Wet Tropics	2017 to 2020	Nitrogen												
Wallace et al. (2020)	Constructed wetland - Off channel facility	Wet Tropics	2017 to 2018	Nitrogen												
N processes Literature																
Adame et al. (2019)	Natural - Forested wetlands	Wet Tropics	2016 to 2017	Nitrogen				N/A	N/A	N/A	N/A					
Adame et al. (2021)	Natural - Floodplain lake	Wet Tropics	2018 to 2019	Nitrogen				N/A	N/A	N/A	N/A					
Canning et al. (2021)	Natural - palustrine wetland	Mackay Whitsunday	2019 to 2020	Nitrogen				N/A	N/A	N/A	N/A					

The 2022 SCS synthesis of evidence included an assessment of the confidence of the evidence base based on an appraisal of the overall relevance (spatial and temporal) and consistency of the evidence (Table 24). The evidence for the non-agricultural and stormwater treatment systems was included within Question 4.6 regarding nutrient management options and is therefore not summarised in the same way as the other questions. The studies added as part of this report have not been added to the appraisal but were limited in number and unlikely to change the ratings presented below. Using this assessment approach, it is clear that the confidence in the evidence for all questions was Moderate, with the exception of urban stormwater which was High, most likely due to the longevity of the evidence base.

The assessment indicates that further evidence is required to increase the confidence in the potential pollutant removal efficiencies of wetland treatment systems in agricultural landscapes in the GBR catchment area, particularly in relation to natural and near-natural wetlands.

Table 24. Summary of the evidence appraisal indicators and confidence ratings in the evidence base for the wetland questions in the 2022 SCS (Q4.7, Q4.8, Q4.9). The Confidence rating was determined by the overall relevance of studies to the question and the consistency of the body of evidence. Note: In Diversity of items: Experimental (E), Meta-analysis (MA), Mixed (X), Modelling or Remote sensing (M), Observational (O), Reviews (R), Theoretical or Conceptual (T).

Question	Quantity of items	Diversity of items	Overall relevance	Consistency	Confidence
[4.7] What is the efficacy of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality (nutrients, fine sediments and pesticides)? What are the key factors that affect the efficacy of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality and how can these be addressed at scale to maximise water quality improvement?	High (238)	High *(45% E, 28% O, 14% M, 7% T, 3% X, 3% R)*	Moderate	Moderate	Moderate
Additional items considered	*5 (see App 1)*				
[4.8] What are the measured costs, and cost drivers associated with the use of natural/near-natural wetlands, restored, treatment (constructed) wetlands and other treatment systems in GBR catchments in improving water quality?	Low to Moderate (56)	High *(41% M, 39% R, 20% O)*	Moderate	High	Moderate
Additional items considered	*19*				

Question	Quantity of items	Diversity of items	Overall relevance	Consistency	Confidence
[4.9] What role do natural/near-natural wetlands play in the provision of ecosystem services and how is the service of water quality treatment compatible or at odds with other services (e.g., habitat, carbon sequestration)?	High (125)	High *(31% O, 18% R, 18% M, 16% T, 12% E, 5% X)*	Moderate	High	Moderate
Non-agricultural	119		Moderate	High	Moderate
Stormwater runoff – wetlands and biofilters	26		High	High	High
Additional items considered	*1*				

6.2 Knowledge gaps

The 2017 SCS identified a number of priority knowledge gaps related to the role of wetland and treatment systems in water quality improvement in the GBR. These were captured and prioritised in the Reef 2050 Water Quality Research, Development and Innovation Strategy 2017-2022 (Australian and Queensland governments, 2018). The priority questions most relevant to this review include:

- What are the impacts of poor water quality on wetland coastal ecosystems?
- How does poor water quality affect the ecological services wetlands provide to the GBR?
- What is the capacity of wetlands to improve water quality from the catchment to the reef?
- What is the conceptual understanding of pesticides in natural, near-natural and artificial wetlands including their transport, fate and retention?
- What is the spatial distribution of groundwater dependent ecosystems and what are their ecological function?
- What is the impact of weed mat infestations on coastal and marine ecosystems of the GBR?
- What is the effectiveness, efficiency and cost of treatment systems for removing nutrients?

Over the last five years there has been some directed research funding to address these knowledge needs and the findings are captured in this review. The research, in conjunction with the development of the Whole-of-System, Values-Based Framework (DESI 2022c), provides a positive foundation for understanding the values and ecological function of wetlands, and increases confidence in understanding pollutant removal efficiencies.

Despite this recent work, the number of studies relating to wetlands and water quality improvement efficacy for different wetland systems in the GBR is still limited, particularly when considering their values, and when compared to other wetlands globally (e.g., the Everglades and Mississippi delta and catchment in the US and the Yellow River and floodplain, alongside regional and coastal areas of China). While there are an increasing number of studies on constructed or treatment wetlands, there is a moderate level of uncertainty in the understanding of the effectiveness of natural and near-natural wetlands in the GBR in N removal and a higher level of uncertainty for sediment, P and pesticides. Variation in research questions, methods, equipment, monitored variables, and sampling frequency/duration also hinders our ability to draw strong conclusions or compare results across studies. There are also limitations to monitoring and evaluation programs, primarily because of resource limitations.

To date, there has been limited support for <u>ongoing</u> assessment or monitoring and evaluation of natural and near-natural wetlands prior to, or following, the completion of a restoration project or activity in the GBR catchment area. There are a wide range of approaches to monitoring water quality outcomes from wetland systems, including a range of sampling equipment (e.g., flow gauges, auto-samplers, loggers, grab samples and piezometers), sampling frequencies, study duration, time of year (i.e., wet or dry season), wetland size, and additional important information (e.g., vegetation cover or hydroperiod). The approach of collecting water quality samples at the defined inlet and outlet to wetlands, and building a water balance model for wetland sites, is not very common and is often carried out inconsistently.

Studies that do not include relevant methodological details or model the water balance over a reasonable period (several years) make interpretation of the water quality improvement efficacy difficult. Further, consideration and evaluation of additional water sources in wetlands, particularly groundwater, are rarely considered or included, which further limits the interpretation of water quality data and the ability to assess the full water and nutrient balance. The inclusion of all these details in publications, as supplementary material, would also assist with comparisons and provide greater context for managers to consider when planning projects.

Details on wetland maturity at the time of sampling are often missing, which can be crucial since older wetlands may have accumulated more carbon to support denitrification (Martínez et al. 2018; Mitsch and Gosselink 1993). Only one study explicitly noted water sampling in a wetland that was less than 1 year since construction (Ham et al. 2010). Providing these details in future studies would be a useful contextual addition for managers.

DESI have developed information for monitoring and evaluation programs within the *Monitoring guidelines to quantify nitrogen removal in vegetated water treatment systems - Treatment system technologies to improve water quality* (Manca and Wegscheidl 2024) which provides guidance on designing water monitoring programs for field-based vegetated water treatment systems, aiming for a more accurate estimation of N reduction. By promoting consistency in monitoring approaches, this framework facilitates comparisons across different vegetated water treatment systems. Within the framework, a tiered approach to monitoring is introduced—gold, silver, and bronze—specifically designed for diverse flow conditions (including baseflow, high discharge, and water-year) in vegetated drains and treatment wetlands. These tiers signify varying degrees of cost and confidence levels, with the gold tier representing the highest confidence level in results, albeit requiring a higher investment. Applying these standards retroactively suggests that none of the existing GBR studies would meet the confidence criteria of even the bronze tier.

While reporting of wetland extent and classification dates back to 2001, the P2R Wetland Condition Monitoring commenced in 2018. Continuation of this program to provide long-term datasets is critical to understand the current status of, and threats to, GBR wetland ecosystems and will also provide important information for understanding wetland hydrology and model validation. Further knowledge of the impacts of sediment accumulation, nutrient enrichment and pesticides on wetland ecosystems is also understudied in the GBR (Collier et al. 2024; Diaz-Pulido et al. 2024; Negri et al. 2024).

There are only a small number of studies that quantify the key costs and cost drivers for wetland treatment systems. The limited evidence, in conjunction with the diversity of the study sites and wetland systems, means that a comparison between costs of different wetland systems is difficult. The limited studies where this information has been reported do however provide an insight into the potential range of costs that may be experienced through wetland construction, rehabilitation or the installation of bioreactors.

Measured costs of wetland projects need to be captured over a consistent timeframe and discount rate to evaluate the effectiveness of programs. This includes costs during the pre-construction phase (e.g., conceptualisation, design, planning, landholder engagements, approvals), construction phase (e.g., earthworks, planting), and post-construction phase (e.g., monitoring, maintenance, repair). Long-term opportunity costs and ongoing maintenance must be considered in assessing the cost effectiveness of wetland projects. These are also important considerations in defining the length of funding programs and monitoring requirements, potentially (and most likely) extending beyond the life of the initial funding program.

Co-benefits (e.g., biodiversity outcomes) are well documented from wetland restoration projects, particularly in large landscape-scale wetlands, but water quality outcomes are often poorly quantified. Clearer articulation of the desired co-benefits must be included from the initial project design as well as the policy and program design. These may also require different monitoring and reporting, and potentially be influenced by different cost drivers that must be considered. There is increasing interest in the application of environmental markets to initiate, incentivise, and fund restoration of wetland projects for a range of ecosystem services. Studies have also mapped and estimated the potential economic return for landholders to transition from farming to wetlands for water treatment or ecosystem services (e.g., blue carbon or biodiversity). However, with the rapid development of environmental markets, the need for a values-based approach is emphasised to ensure that trade-offs are considered, perverse outcomes are avoided, and that monitoring and evaluation programs are in place to capture the learnings and successes.

The potential implications of future climate change projections, such as sea level rise and more severe weather events (e.g., cyclones), also require careful consideration when locating and designing wetland projects and activities in the GBR. This also highlights the need for a co-design process early in the project cycle where all stakeholders and beneficiaries are involved in setting the ecosystem service goals.

Finally, there is a need for policies and planning to ensure the long-term protection and conservation of the remaining natural and near-natural wetlands in the GBR catchment area. To support this, the establishment of long-term and values-based whole-of-system management plans are essential and must include adequately resourced and regular monitoring on the performance, health and function of the wetlands and associated flora and fauna, and long-term maintenance plans.

Table 25 summarises major knowledge gaps identified through the SCS questions and this review. Addressing these gaps requires a comprehensive research strategy tailored to the GBR's diverse landscapes and climate conditions. Improved data will support model development to predict water quality outcomes from wetland treatment and/or restoration projects, inform management decisions and enhance policy and funding strategies. This is also recognised in the Reef 2050 Wetlands Strategy, which has a Theme related to improving wetlands information for decision making and action.

Table 25. Summary of knowledge gaps related to understanding the effectiveness of wetlands in pollutant processing. Modified from Waltham et al. (2024a, 2024b) and Star et al. (2024a).

Gap in knowledge	Possible research or Monitoring & Evaluation question to be addressed	Potential outcome or impact for management if addressed
Effectiveness of wetlands in pollutant processing		
Wetland hydrological dynamics over multiple years.	What is the residence time of water in different wetland systems and how does this vary over time?	This information would allow catchment-wide wetland efficacy to be assessed and assist in the design of treatment wetlands and restoration of natural and near-natural wetlands.
Quantification of pollutant processing in wetland systems under a range of conditions and	What is the effectiveness of wetlands in pollutant processing in a diverse range of land uses,	This information would allow catchment-wide wetland efficacy to be assessed and assist in the

Gap in knowledge	Possible research or Monitoring & Evaluation question to be addressed	Potential outcome or impact for management if addressed
different landscapes through targeted observational studies.	hydrology, climate and soil conditions in 1) single systems and 2) different configurations of multiple systems in the landscape.	design of treatment wetlands and restoration of natural and near-natural wetlands.
The role of natural and near-natural wetlands in water quality improvement, including an understanding of the quantification of the changes to catchment hydrology as a result of land use change, the implications of infrastructure including drainage and barriers to flow, and widescale landscape modification including floodplain development.	How do natural and near-natural wetlands contribute to pollutant processing across a range of conditions, historic and current landscape modifications?	The impacts of these changes on biological processes and water quality improvement in wetlands will provide a better understanding of the optimal placement of wetland treatment systems in the landscape.
Water quality conditions under different hydrograph periods.	What does the shape of the nutrient and sediment concentration and load graphs look like for different land uses, catchment areas and rainfall event sizes?	These data would assist the design of treatment wetlands and restoration of natural and near-natural wetlands.
Dissolved oxygen cycling in wetlands.	What is the optimal range of dissolved oxygen concentrations necessary to maximise nutrient processing in wetlands? What are the wetland requirements to optimise these desirable dissolved oxygen concentrations?	Designing treatment wetlands and restoration of natural and near-natural wetlands.
Sediment particle size distribution in wetlands.	To what extent do wetlands capture and retain sediments from flow events? What is the distribution of sediment (and particulates) particle sizes stored in, and passing through wetlands and how does this affect them over time, including the role of vegetation?	Improved understanding of the sediment accumulation rates in wetlands.
Vegetation specific nutrient and sediment processing potential.	To what extent do native wetland vegetation species provide water quality improvement? What density and/or % cover of vegetation is most effective for water quality improvements?	Knowledge on the specific role each wetland plant species has in improving water quality. These data would be used in modelling efficacy and for cost-effective assessments.
Groundwater contribution to water balance and nutrient processing in wetlands.	What is the degree of interaction between groundwater and surface waters in wetlands, and how does this interaction change over spatial-temporal scales in the GBR? What are the drivers of groundwater	Groundwater contribution to wetlands is poorly understood, even overlooked, in studies in the GBR, but also more broadly. Modelling the contribution of groundwater to wetlands is

Gap in knowledge	Possible research or Monitoring & Evaluation question to be addressed	Potential outcome or impact for management if addressed
	contribution to wetlands and how does this change with land use, land use change and in the restoration of wetland ecosystems?	complex and can vary over complex spatial-temporal scales.
Climate change with respect to changing rainfall and flow through natural and near-natural wetlands.	What is the response of wetland ecosystem services under more variable hydrology (i.e., increase erosion susceptibility or sedimentation accumulation).	Understanding the sediment characteristics, processes, and dynamics in wetlands (levels of sediment accretion). These data would assist with informing maintenance needs in the wetlands and impacts of climate change.
Effectiveness of engineered treatment wetlands.	What is the optimal sized wetland to treat particular catchment sizes, for each dominant land use in the wet tropics? What is the optimal sized wetland to treat catchments, for each dominant land use in the dry tropics? What role would bioreactors have in the landscape in providing water improvement in agricultural areas?	Model development for effective design and construction for land use and environmental conditions in GBR catchments. This model design could be used in catchment-scale monitoring to back calculate how many wetlands are needed, among other land use strategies, to achieve water quality targets. Cost benefit analysis could be completed.
Effectiveness of floating treatment wetlands (FTW).	How does the removal efficiency of FTWs compare to *in situ* constructed/treatment wetlands? What is the potential for FTW use in the tropics? How is the water quality improvement efficiency of FTWs maximised? How are these systems designed to maximise resilience in tropical climates?	There is evidence elsewhere that FTW can treat nutrients. Pilot studies are needed to determine their utility and application in the tropics.
Effectiveness of epibenthic algal mats.	Do epibenthic algal mats improve the pollutant removal efficiency of natural/restored/ treatment/constructed wetlands?	Epibenthic algal mats are highly productive and may contribute to the water quality improvement efficiency of wetlands.
Longevity, maintenance and renewal requirements over time.	What is the performance of systems in the longer-term, what maintenance and renewal actions are required, and what are the associated costs and mechanisms needed to ensure how these can be best addressed over the life span of the system?	Greater confidence in the design of wetland systems for ongoing pollutant removal, ongoing maintenance requirements and associated costs.
Cost drivers and cost-effectiveness of wetland systems		
Measured costs across all cost categories and measured water	What are the measured costs across all cost categories and measured	Understanding the cost effectiveness and the potential

Gap in knowledge	Possible research or Monitoring & Evaluation question to be addressed	Potential outcome or impact for management if addressed
quality improvements in GBR locations in a comparable way (standard metric).	water quality improvements in GBR locations?	total cost for implementing a wetlands strategy.
Understanding at a landscape level of where wetlands could be situated to achieve efficient pollutant reductions and the subsequent actions and costs to achieve the reductions.	Where in the landscape could a series or different systems of wetlands be most efficient at achieving pollutant reductions and what is the subsequent cost?	Optimisations of funds and actions long term.
Impacts of climate change on the construction and post-construction phase costs.	What was the weather sequence which resulted in making amendments to the wetland at either construction or post-construction phase?	Targeting across the landscape to limit risk due to extreme weather outcomes.
Consideration of policy and legislative mechanisms and approaches over time to achieve the targeted reductions i.e., incentives or trading scheme.	What policies over time are required to achieve the outcomes based on costs and adoption?	Optimisations of funds and actions long term.
Stacked actions such as paddock-scale management, drains buffers and then wetland management.	What are the ongoing combinations of different wetland management actions that achieve the best outcomes in the shortest timeframes?	Understanding the cost effectiveness of wetlands in the GBR catchment area.
Capacity to achieve co-benefits and the mechanism to achieve them.	Would we like to see co-benefits achieved with water quality improvements? If so, what are they? And what are the design modifications that are required to achieve these outcomes over time? What policies will be applied and will these be stacked?	Co-benefits realised.

6.3 Implications for the design and on-ground delivery of wetland projects in the GBR

Drawing on the evidence captured in previous documents (2017 SCS, Wetland*Info*, Reef 2050 Wetlands Strategy) and this review, there are several factors associated with on-ground implementation of projects that influence the performance of wetland systems in water quality improvement. Many of these factors have been demonstrated to some extent in the GBR context, and are clearly recognised in the resources provided in Wetland*Info*, including the Whole-of-System, Values-Based Framework (DESI 2022c) and Aquatic Ecosystems Rehabilitation Process (DESI 2021),. These factors are also important considerations in the development of a model to represent pollutant processing in wetland systems and are outlined below.

The following learnings draw on the evidence base to identify key considerations for wetland design, establishment and maintenance in the GBR catchments, with examples provided where relevant. It also highlights some of the limitations to existing studies in assessing treatment performance, recognising though that projects might not have been specifically designed for this objective.

1. **Clearly defined objectives:** Successful projects start with well-defined goals linked to a values-based approach. This clearly aligns with the Whole-of-System, Values-Based Framework (DESI 2022c) that focuses on the components and processes in wetlands that can maximise provision of intrinsic values and ecosystem services for beneficiaries and therefore the overall project success. The upfront definition of goals is also recognised as a major step in many planning frameworks including the Aquatic Ecosystem Rehabilitation Process (DESI 2021).

   The primary benefits and the policy and program design are critical to project success. If co-benefits are sought, the characteristics of the specific benefits, the capacity to stack multiple benefits, the framework that is applied to measure and achieve the primary and additional benefits, the timescale expected to achieve benefits, and the monitoring and maintenance frameworks required to demonstrate outcomes must be documented.

   There also remains a challenge around what the **role of existing wetland systems** should be if they are being restored, but using the values-based approach as an essential starting point can help to identify primary and secondary goals for the wetland. For example:

   - Degraded systems might be restored by focusing on the primary goal of re-establishing their intrinsic values around habitat, ecosystem function and overall biodiversity, with any potential water quality improvement being a secondary goal.
   - Newly constructed wetlands may have a primary goal of water quality treatment, but secondary goals around habitat or biodiversity may influence key design choices such as plant species, wetland shape and/or bathymetry.
   - Wetlands that have become isolated from the hydrologic regime may have a primary goal of restoring their hydrologic function to ensure they can become sustainable wetland systems, with water quality treatment being a secondary goal.

2. **Wetland position in the landscape**: The location of the proposed or existing wetland needs to be viable for pollutant processing in the overall landscape. This needs to consider:

   a. Contributing catchment areas, e.g., can the wetland receive flows that are within a suitable hydrologic regime for the system (i.e., are there sufficient volumes of inflows to maintain a wetting and drying regime that will provide long-term sustainability for wetland vegetation). If the contributing catchment area is too small, the system may dry out for too long and vegetation will not be able to recover, or if the catchment is too large, water levels or flow rates may drown or scour plants and prevent their future recovery.

   b. Relationship to downstream conditions, e.g., can the wetland drain when required or are the downstream conditions unsuitable to allow for the wetting and drying regimes to be maintained.

   c. Connectivity and habitat fragmentation, e.g., does the wetland need to provide habitat or biodiversity linkages, or does it interrupt existing connectivity.

   d. Adjacent land uses e.g., can the wetland integrate with adjacent land uses successfully. For example, it may not be appropriate to put a constructed wetland immediately adjacent to built-up areas without considering a suitable buffer zone.

3. **Regulatory landscape:** Wetlands are part of the hydrologic network and are often constructed within or immediately adjacent to other waterways. There are a range of regulatory requirements that may need to be considered around fish passage, fish habitat, vegetation clearing, dredging etc., some of which can have significant cost implications. Two examples of this are wetlands developed in Bakers Creek in the Mackay Whitsunday region, and the Tully Landscape Wetland developed as part of the Wet Tropics Major Integrated Project. In both cases, the requirement to facilitate fish passage resulted in the need to construct large fishways which significantly increased the costs and delivery timeframes of both projects.

4. **Hydraulics**: Understanding how water moves through the wetland including via groundwater is essential to maximising treatment effectiveness. Wetland water quality treatment effectiveness has a strong dependency on the period of time that a “parcel” of water can interact with the wetland processes that result in improvement, such as enhanced sedimentation, denitrification, biofilm interaction and plant uptake. If water is able to move through a wetland too quickly, such as short-circuiting of the flow path because of insufficient vegetation density or high flow, then there will be insufficient time for the wetland processes to improve water quality. Understanding the hydraulic efficiency of the system can therefore be critical to enhancing the overall treatment effectiveness of a system, including understanding flow paths, mixing processes, retention times, seasonality effects and the discharge regime.
5. **Establishment phase**: Palustrine wetlands are vegetated systems, but also interact with the hydrologic regime. Just as plants take time to grow, it takes time for wetlands and the processes within them to establish and become fully functional. This establishment period, as indicated by the literature reviewed in this synthesis, can range from a few months to several years, but this work also shows that many systems continue to improve over time.

   There is also a need to ensure that what has been planned and/or designed is delivered on-ground. Many projects have failed because they have diverted from delivering the designed system (and the values associated with delivering that design), to simply delivering “the project” without considering that it is a functioning ecosystem that needs to have all of the intrinsic design elements properly integrated.
6. **Maintenance:** It is clear from the evidence base, and locally in the GBR, that lack of maintenance has a major influence on project success. This is often because maintenance was not considered in the project design and delivery, or the resourcing of maintenance was not allowed for or not specifically incorporated into the cost/funding model to continue maintenance into the future. Key aspects such as plant survival and recruitment, weed control, pest management, sediment removal, litter removal, bird predation, management of inlet and outlet structures and overall water level control will need to be considered and both routine and rectification activities planned for. These will have recurrent and sometimes capital expenditure costs associated with them over the long-term (20 years).
7. **Disturbance events**: Major floods, cyclones, extended droughts and bushfire may all cause significant impacts to both natural and constructed wetland systems. In most cases, if the damage is not extensive, wetland systems and processes can recover quickly, but often some rectification may be needed such as additional planting, water level management or debris removal. These are likely to be short term actions but can help reduce long-term costs and prevent wholesale failure or regime shifts in system processes. For example, a local wetland in an urban area experienced a major flood with build-up of debris at the flow outlet. This resulted in very high water levels being maintained for long periods, resulting in drowning of vegetation and potentially complete destruction of the wetland plants. The plants and associated nutrient processes did not re-establish quickly, and algal species became dominant with the wetlands shifting from vegetated systems to open water bodies experiencing regular algal blooms.
8. **Monitoring and evaluation**: As described above in Section 6.2, monitoring and evaluation of the performance of wetland systems in terms of the effectiveness of pollutant processing and water quality outcomes is essential, in addition to regular checking of wetland design and integrity of any supporting infrastructure. This is important to understand the performance of the system, ensure the expected effectiveness is maintained, but also to provide information necessary for any future decision-making tools. For example, development and refinement of models requires relevant monitoring data to calibrate and validate models. Without these local data, the development of these models is dependent on published literature or expert judgment. Monitoring and evaluation can be expensive and time consuming, but could become more cost-effective with new technologies, shared learnings and data. Appropriate training, experience and application of quality assurance and quality control procedures is

required to ensure that the data is of sufficient quality for a range of applications. Data should also be freely available, which means it needs to be stored on an open access portal that is maintained and quality checked for any errors, and any limitations or caveats to its quality or use are flagged.

In addition to these practical considerations, there are several aspects of policy and planning requirements that need to be considered in on-ground delivery of wetlands projects. These were identified in Section 5 and relate to planning, policies and regulatory requirements, and the consideration of increasing interest in delivering multiple benefits. Additional information is regularly updated on these aspects on Wetland*Info* under 'Programs, policy and legislation' (DESI 2023).

6.4 Summary of the GBR evidence base, knowledge gaps and future work

The global review draws on 238 tropical and subtropical studies in agricultural areas and 145 studies in non-agricultural areas. In the GBR catchment area, the review identified a limited number (17) of studies, primarily focusing on nitrogen removal, with few studies on sediment and pesticide processing. Most studies were conducted after 2019, and they are concentrated in the Wet Tropics and Dry Tropics, with few studies from other regions. The studies show significant variability in methodologies, monitoring approaches, and hydrological conditions, which limits the ability to draw conclusions about wetland efficacy. Despite recent advancements, knowledge gaps remain, particularly regarding the role of natural and near-natural wetlands in pollutant processing and the need for long-term data collection.

Future work should focus on addressing these knowledge gaps by exploring the hydrodynamics of wetlands, quantifying pollutant processing under different conditions and understanding the influence of groundwater and landscape changes on wetland function. Additionally, more research is needed on the cost-effectiveness of wetland treatment systems and how these systems can be optimised to improve water quality. Integrating long-term monitoring and evaluation programs into wetland projects will help establish baselines, inform management decisions, and support the development of policies for protecting and enhancing wetlands in the GBR catchment area and understanding their role in water quality improvement.

In terms of on-ground implementation, it is clear that successful projects start with clearly defined goals, taking into account frameworks that take a holistic approach such as the Queensland Wetland Program's Whole-of-System, Values-Based Framework. Effective wetland design requires careful attention to the landscape's hydrology, vegetation density, and hydraulic efficiency, as well as the wetland's ability to receive and process flows. Regulatory considerations, including fish passage and environmental laws, can also impact project costs and timelines, as seen in projects in the Mackay Whitsunday and Wet Tropics regions. Wetlands take time to establish, and their maintenance is crucial for long-term functionality. Restoration after disturbance events may also be required. Baseline monitoring and ongoing monitoring and evaluation of performance is essential to inform optimisation of the wetland's effectiveness. Finally, policies, planning, and considerations for co-benefits, like habitat restoration, are important aspects to be integrated into project design and implementation.

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Appendix 1: Great Barrier Reef studies included in the review

Table A1.1. Summary of studies that report wetland efficacy or nitrogen processes in the GBR. Cells/Lines in grey refers to non-peer reviewed literature.

Author	Year	Title
Effectiveness Literature		
McKergow L.A., Prosser I.P., Grayson R.B., Heiner D.	2004	Performance of grass and rainforest riparian buffers in the wet tropics, Far North Queensland. 2. Water quality
McJannet D., Wallace J., Keen R., Hawdon A., Kemei J.	2012	The filtering capacity of a tropical riverine wetland: I. Water balance and II Sediment and nutrient balances
Navaratna D., Shu L., Baskaran K., Jegatheesan V.	2012	Treatment of ametryn in wastewater by a hybrid MBR system: A lab-scale study
Adame M.F., Roberts M.E., Hamilton D.P., Ndehedehe C.E., Reis V., Lu J., Griffiths M., Curwen G., Ronan M.	2019	Tropical Coastal Wetlands Ameliorate Nitrogen Export During Floods
Pfumayaramba, T., Wegscheidl, C., Nothard, B.	2020	A preliminary cost-effectiveness analysis of denitrifying bioreactors in Lower Burdekin
Kavehei E., Hasan S., Wegscheidl C., Griffiths M., Smart J.C.R., Bueno C., Owen L., Akrami K., Shepherd M., Lowe S., Adame M.F.	2021a	Cost-effectiveness of treatment wetlands for nitrogen removal in tropical and subtropical Australia
Manca F., Wegscheidl C., Robinson R., Argent S., Algar C., De Rosa D., Griffiths M., George F., Rowlings D., Schipper L., Grace P.	2021	Nitrate removal performance of denitrifying woodchip bioreactors in tropical climates
Kavehei E., Roberts M.E., Cadier C., Griffiths M., Argent S., Hamilton D.P., Lu J., Bayley M., Adame M.F.	2021b	Nitrogen processing by treatment wetlands in a tropical catchment dominated by agricultural landuse
Wallace J., Waltham N.J.	2021	On the potential for improving water quality entering the Great Barrier Reef lagoon using constructed wetlands
Wallace J., Bueno C., Waltham N.J.	2022	Modelling the removal of nitrogen and sediment by a constructed wetland system in north Queensland, Australia
Rafiei, V; Nejadhashemi, AP; Mushtaq, S; Bailey, RT; An-Vo, DA	2022	Groundwater-surface water interactions at wetland interface: Advancement in catchment system modelling
Alexander W. Cheesman, Shannon Todd, Liz Owen, Dennis AhKee, Han She Lim, Maureen Masson, Paul N. Nelson	2023	In-drain denitrifying woodchip bioreactors for reducing nitrogen runoff from sugarcane
Alluvium	2016	Costs of achieving the water quality targets for the Great Barrier Reef
Wallace, J., Adame, M.F., Waltham N.J.	2020	A treatment wetland near Babinda, north Queensland: a case study of potential water quality benefits in an agricultural tropical catchment
Wegscheidl, C, Robinson, R & Manca, F	2021	Using denitrifying bioreactors to improve water quality on Queensland farms: Case study 3
		Using denitrifying bioreactors to improve water quality on Queensland farms: Case study 6

Author	Year	Title
		Using denitrifying bioreactors to improve water quality on Queensland farms: Case study 7
N processes Literature		
Adame M.F., Franklin H., Waltham N.J., Rodriguez S., Kavehei E., Turschwell M.P., Balcombe S.R., Kaniewska P., Burford M.A., Ronan M.	2019	Nitrogen removal by tropical floodplain wetlands through denitrification
Adame M.F., Waltham N.J., Iram N., Farahani B.S., Salinas C., Burford M., Ronan M.	2021	Denitrification within the sediments and epiphyton of tropical macrophyte stands
Canning, A., Adame, F.A. and Waltham, N.	2021	Evaluating services provided by ponded pasture wetlands in Great Barrier Reef catchments – Tedlands case study
Adame, M. F., Vilas, M. P., Franklin, H., Garzon-Garcia, A., Hamilton, D., Ronan, M., Griffiths, M.	2021	A conceptual model of nitrogen dynamics for the Great Barrier Reef catchments

Appendix 2: Wetland processes and related components

Table A2.1: Characteristics of wetland processes and related components, and the significance of the process when assessing pollutant processing.

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
Open water		
Nitrification/nitrogen fixing	Temperature	H
	Sediment composition (particle size)	M
	Inflow volume	H
	Denitrification species	H
	Organic carbon	M
	Dissolved oxygen	M
	Time (duration)	M
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	L
	Depth	M
	Redox	H
Macroinvertebrate grazing/predation	Temperature	M
	Algal species	M
	Organic carbon	M
	Dissolved oxygen	M
	Time (duration)	H
	Time (seasonality)	M
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	L
	Depth	M
Algal growth and decay	Temperature	H
	Inflow volume	H
	Algal species	H
	Nitrogen concentration	M
	Time (frequency)	M
	Time (duration)	H
	Time (seasonality)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	M
	Depth	M
Stratification	Temperature	H
	Inflow volume	H
	Time (frequency)	M

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
	Time (duration)	H
	Time (seasonality)	M
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	H
	Depth	H
Adsorption/desorption	Temperature	M
	Sediment composition (particle size)	H
	Time (duration)	H
Dissolution/flocculation	Temperature	M
	pH	H
Benthic zone		
Sedimentation/resuspension	Sediment composition (particle size)	H
	Inflow volume	H
	Time (duration)	H
	Shape	M
	Depth	M
	Inflow configuration	H
	Surface area	M
Sediment/water column nutrient flux	Temperature	H
	Sediment composition (particle size)	H
	Inflow volume	M
	Biofilm species	M
	Denitrification species	M
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	M
	Time (duration)	H
	Depth	H
	pH	H
	Redox	H
Nitrification/denitrification	Temperature	H
	Sediment composition (particle size)	H
	Inflow volume	M
	Biofilm species	M
	Denitrification species	M
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	M
	Time (duration)	H

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
	Depth	H
	pH	H
	Redox	H
Nitrogen assimilation/ annamox/ ammonification	Temperature	H
	Sediment composition (particle size)	H
	Inflow volume	M
	Biofilm species	M
	Denitrification species	M
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	M
	Time (duration)	H
	Depth	H
	pH	H
	Redox	H
Litterfall/organic matter accumulation	Temperature	M
	Inflow volume	H
	Plant species	H
	Plant density	H
	Organic carbon	L
	Time (duration)	H
	Time (seasonality)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	M
	Shape	L
	Depth	M
Biological uptake at sediment/water column interface	Temperature	H
	Sediment composition (particle size)	H
	Biofilm species	H
	Denitrification species	H
	Organic carbon	M
	Dissolved oxygen	M
	Nitrogen concentration	M
	Time (duration)	H

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	L
	pH	M
	Redox	H
Oxygenation/deoxygenation	Temperature	H
	Sediment composition (particle size)	M
	Inflow volume	M
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	L
	Time (duration)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	M
	Redox	H
Vegetated zone		
Nitrification/denitrification	Temperature	H
	Sediment composition (particle size)	H
	Inflow volume	M
	Biofilm species	M
	Denitrification species	M
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	M
	Time (duration)	H
	Depth	H
	pH	H
	Redox	H
Nitrogen assimilation/ annamox/ ammonification	Temperature	H
	Sediment composition (particle size)	H
	Inflow volume	M
	Biofilm species	M
	Denitrification species	M

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
	Organic carbon	M
	Dissolved oxygen	H
	Nitrogen concentration	M
	Time (duration)	H
	Depth	H
	pH	H
	Redox	H
Litterfall/organic matter accumulation	Temperature	M
	Inflow volume	H
	Plant species	H
	Plant density	H
	Organic carbon	L
	Time (duration)	H
	Time (seasonality)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	M
	Shape	L
	Depth	M
Biological uptake at plant/water column interface	Temperature	H
	Inflow volume	M
	Plant species	H
	Organic carbon	M
	Nitrogen concentration	M
	Time (duration)	H
	Time (seasonality)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	L
Plant uptake (root zone)	Temperature	H
	Plant species	H
	Plant density	H
	Time (duration)	H
	Time (seasonality)	H
	Perturbing events (cyclone, flood, feral animals, desilting, weeds, vegetation failure)	L
Wetting/drying	Temperature	H
	Inflow volume	H
	Time (duration)	H
	Shape	L
	Depth	H

Process	Component/driver	Significance for process H-High, M-Medium, L-Low
	Inflow configuration	M
	Outflow configuration	H
Adsorption/desorption	Temperature	M
	Sediment composition (particle size)	H
	Time (duration)	H
Photosynthesis/respiration	Temperature	H
	Plant species	M
	Plant density	M
	Time (duration)	H
	Time (seasonality)	M