WATER SERVICES
ASSOCIATION OF AUSTRALIA

How a nutrient trading regime can deliver environmental outcomes

March 2023

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The Water Services Association of Australia (WSAA) is the peak body that supports the Australian urban water industry. Our members provide water and sewerage services to over 24 million customers in Australia and New Zealand and many of Australia's largest industrial and commercial enterprises. WSAA facilitates collaboration, knowledge sharing, networking and cooperation within the urban water industry. The collegiate approach of its members has led to industry-wide advances to national water issues.

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EXECUTIVE SUMMARY

Novel approaches are increasingly being used to fund restoration of degraded catchments across Australia and internationally. Nutrient offsetting is one such mechanism where buyers, typically point source polluters such as the wastewater industry, i.e., utilities, pay sellers for restoration works in the rural and urban landscapes to reduce nutrients as a way of offsetting nutrient discharge from community wastewater treatment plants.

The water industry can use nutrient offsetting as a tool to meet regulatory requirements for nutrient discharge, especially as regulations become more stringent, treatment technology limits are reached or output from plants increase with correlated population demand. It may also be used along with upgrades to treatment processes, and/or on-site nutrient mitigation strategies. Interviews with a range of people involved in nutrient offsetting, either within the water industry or in state government agencies throughout Australia, have identified a range of projects in various stages of development.

State government policies (in some states) and agreements provide a mechanism for approval of offsetting projects, and there are several examples of success where modelled nutrient offsetting loads from catchment restoration have equaled or exceeded nutrient loads from wastewater treatment plants. However, this success has been achieved in a piecemeal way, project by project, resulting in considerable long-term overheads for the water industry to set up and administer the project, for which costs are, in turn, borne by customers. There have been many benefits from offsetting projects, not just to the water industry and its customers, but also for improved water quality and ecosystem health, landholders, local communities, catchment management groups and consultants. However, these benefits are typically not quantified. Additionally, there are environmental co-benefits, such as carbon capture and flood mitigation from tree planting, a reduction in erosion and hence sediment and land loss to waterways, and biodiversity benefits in the riparian and aquatic environments.

Whilst the interviews and literature review suggest that nutrient offsetting has provided many benefits, it is also clear that there is considerable scope for improvement in the scheme, on many levels. Key challenges identified by interviewees were significant costs in coordinating projects, lack of good quality data on return on investment (ROI), time taken to realise benefits, and risks of not meeting government compliance requirements (at a state level). Additionally, with respect to the involvement of government regulators, it was clear that the water industry felt that regulators were not sharing the risk in terms of approving schemes. There was a lack of coordination and consistency in policy, and lack of information on government targets for the environment more broadly, e.g., sustainable loads for waterways. Conversely, regulators within state government felt constrained by the regulatory role that they need to play, which prevents risk sharing.

A review of the international literature identified examples of regionally coordinated schemes which serve as exemplars for a more coordinated and efficient mechanism for linking investment by utilities with targeted outcomes from catchment restoration. This approach reduces the overhead costs and risks for wastewater industry to engage in nutrient offsetting for overall improvements in catchment water quality. Possible models of types of nutrient offsetting projects and schemes, focused on those involving catchment restoration are shown below (Figure 1):

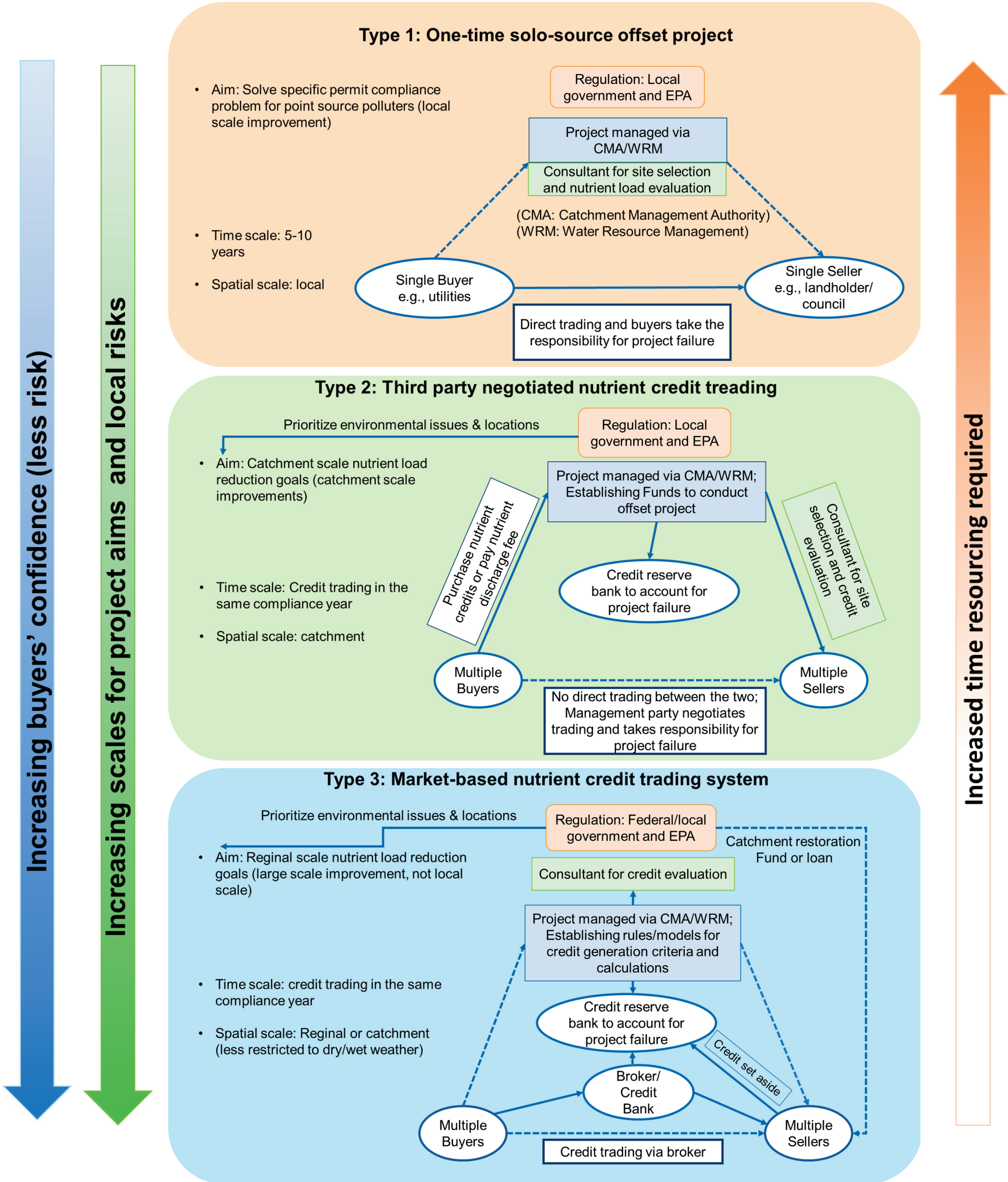


Figure 1. Three nutrient offset strategies/frameworks involving non-point source offsetting that can be adopted with different aims and risks. Solid arrows indicate direct interaction between two components in the diagram. Dashed arrows indicate indirect interaction (or facilitation) between the two components.

1

Type 1 is typical of the approach currently taken in Australia by water utilities, i.e., one-off, solo-source offsetting project. However, this approach results in a complex and time-consuming process for water utilities, and typically focused on localised areas.

2

Type 2 is a third party negotiated nutrient credit trading that has been adopted in international examples, such as Chesapeake Bay, Virginia. In this case, there are multiple buyers and sellers within the program, with a credit reserve bank to account for project failure. The benefit of this approach for water utilities is that it is simpler to set up and manage, with more flexibility in site selection for offsetting works. The environmental benefits may also be broader but can also have local outcomes.

3

Type 3 is the market based nutrient credit trading scheme. This scheme also has multiple buyers and sellers, with a broker or a credit bank facilitating credit trading. An example of this is the Nutrient Credit Trading Scheme in Pennsylvania, targeting Chesapeake Bay. This is a more sophisticated scheme which can deliver broader scale environmental benefits but requires leadership and coordination.

Based on interviews with water utilities across Australia, it is clear that the Type 1 model, which is the norm, has a number of deficiencies. Therefore, progress towards a Type 2 model is likely to create considerable benefits for the water industry, most notably greater efficiency and less transactional costs for utilities. There is also demonstrated success as measured by reduced nutrient loads to waterways with the Type 2 model internationally. However, it requires leadership from the water industry to champion this approach and a willingness of various levels of government to engage.

Another component of this study was to identify scientific knowledge gaps, both nationally and internationally, which are currently increasing the uncertainty and risk around water industry engagement in nutrient offsetting. This includes:

- The need to improve calibration/validation data for models used to determine nutrient load reduction from catchment mitigation.
- More robust methods to improve comparability of point and non-point nutrient sources, including equivalency and delivery ratios.
- Improved data to determine the assimilative capacity/sustainable loads of receiving water environments.
- More robust science concerning the efficacy of different catchment mitigation options.
- Methods for incorporating co-benefits, including biodiversity, carbon, and sediment, into nutrient offsetting programs.
- Analysis of the costs versus benefits of nutrient offsetting over the long term, including co-benefits.

The recommendations outlined below are designed to establish a nutrient trading regime that promotes collaboration, efficiency, and environmental sustainability, addressing both short- and long-term objectives. Drawing on valuable insights from utilities and government stakeholders, as well as best practices derived from successful international nutrient offsetting programs, these recommendations emphasize a coordinated approach, expert engagement, and a commitment to research and development. We believe that implementing these recommendations will lead to the creation of an effective nutrient trading framework, ultimately benefiting the water industry, the environment, and all relevant stakeholders:

1. Create a Maturity Framework: Develop a maturity framework for nutrient offsetting that scales from local to regional levels, focusing on cost-effectiveness and minimal bureaucracy. This framework should establish clear objectives and provide guidance for all stakeholders involved in nutrient trading.

2. Foster Knowledge Sharing: Invite national and international experts experienced in nutrient trading to give presentations to the water industry and government stakeholders. This will encourage knowledge sharing, collaboration, and the development of best practices for nutrient offsetting.

3. Establish an Industry Task Force: Consider creating a dedicated task force within the water industry to advise on strategies and inform advocacy for an improved nutrient trading scheme at the state and/or federal government level. This group should consist of experts from various sectors, ensuring diverse perspectives and well-informed recommendations.

4. Effective Monitoring and Evaluation: Implement a well-defined monitoring and evaluation system that assesses and ensures the presence of key success factors for nutrient offsetting programs. This system should focus on securing buy-in from local government, regulated point-source polluters, and other stakeholders within the catchment, enabling the continuous tracking of progress and outcomes in the industry

5. Standardize Estimation Methodologies: Develop consistent and standardized estimation methodologies for nonpoint source actions in Australia, drawing from successful nutrient offsetting programs internationally. This will promote uniformity and trust in the nutrient trading process.

6. Support Research Projects: Invest in research projects that address key scientific knowledge gaps, reduce risk, and increase confidence in nutrient offsetting. This should include improving data and models, optimizing sites for remediation, and effectively quantifying co-benefits.

7. Integrate Green Solutions: Incorporate green solutions into nutrient trading programs to deliver a range of benefits, such as biodiversity enhancement, sediment reduction, and carbon capture. This holistic approach will ensure the best possible outcomes for both the water industry and the environment.

INTRODUCTION– DRIVERS AND BENEFITS OF NUTRIENT OFFSETTING

The intent of this paper is to outline benefits, hazards for implementation and management, and a range of governance framework approaches for catchment nutrient offsets based on existing market mechanism experiences across Australia and internationally. While nutrients are the primary focus of this paper, the associated benefits and implications for sediments will likely be covered to some extent.

The audience for this paper is water utilities, environmental regulators, and other stakeholders and policy-makers. It is also desired to present a compelling case that existing offset regimes are effective and do deliver beneficial and social environmental outcomes. It will need to tease out that the regulatory contexts in different states of Australia vary, and the way in which nutrient offsetting approaches may interact with those regulatory frameworks.

There can be overlap between nutrient offset regimes and trading regimes. The main focus of this paper is frameworks for offsetting. To acknowledge the relationship between the two, this paper starts with a definition of offsetting.

Background

Eutrophication, primarily resulting from excessive nutrient inputs, has been an important environmental issue worldwide, causing environmental impacts, e.g., toxic algal blooms, fish kills (Diaz and Rosenberg, 2008; Malone and Newton, 2020). A global study on the safe operating space for humanity identified excessive nitrogen pollution as a major risk, being outside of a safe operating load (Rockström et al. 2009). Excessive nutrient inputs to waterways are delivered from point- (e.g., wastewater treatment plants) and non-point sources (e.g., agricultural sources from degraded catchments). Within Australia, recent regulations on point-source discharge have demanded net zero discharge of dissolved inorganic nitrogen from new or upgraded facilities in the Great Barrier Reef catchments (https://environment.des.qld.gov.au/__data/assets/pdf_file/0019/238132/era-gl-reef-discharge-standards-industrial-activities.pdf).

Diffuse catchment discharge of nutrients is not regulated but can also play an important role in increasing eutrophication in the downstream aquatic ecosystems. Catchment degradation is a major issue throughout Australia and can be caused by a range of activities, mostly associated with human-induced land use impacts, such as soil erosion (Kometa, 2019; Renaud et al., 2021). Catchment degradation can also be exacerbated by severe weather conditions (Azimi Sardari et al., 2019; Behera et al., 2020; Pruski and Nearing, 2002).

Nutrient offsetting provides the opportunity for a win-win mechanism to tackle catchment and aquatic ecosystem degradation. It allows nutrient source dischargers with high abatement costs on-site to purchase nutrient load reductions from sources off-site that have lower abatement costs (Selman et al., 2009). For example, to offset the expansion of nutrient discharge facilities to accommodate a growing human population, utilities can directly invest in catchment restoration or nutrient-reducing best management practices (BMPs) that reduce nutrient runoff from catchments. In some regions of the world, utilities can pay a point source discharge fee to agencies that coordinate non-point source reduction activities. Therefore, nutrient offsetting offers greater flexibility on the timing, location, cost, and level of technology needed for nutrient management than a nutrient source polluter might need by upgrading existing plants or building plants with higher levels of treatment (Hall, 2012).

For the purposes of this review, nutrient offsetting is defined as a mechanism for buyers to quantitatively offset their nutrient discharge to waterways by paying sellers to ensure nutrient remediation works occur offsite. It should be acknowledged that at times a utility can be both a buyer but play a key and ongoing role as a facilitator for the seller.

Utilities typically have nutrient discharge load licenses issued by Australian state and territory governments to comply with in order to continue to operate. In contrast, there is no regulation associated with nutrient offsetting, but rather state governments have a range of policies to facilitate offsetting. For example, the Queensland Government developed a policy in 2019 (https://environment.des.qld.gov.au/__data/assets/pdf_file/0033/97845/point-source-wq-offsets-policy-2019.pdf). In this policy total nitrogen is used as the comparison factor for offsetting and the offset ratio between point and non-point sources is 1.5:1. This ratio takes into account a 1:1 equivalency ratio (ecosystem effects of each ton of total nitrogen from point and non-point sources) and a 1.5:1 delivery ratio (accounting for the spatial separation of point and non-point sources with non-point sources typically being upstream of point sources). The policy is set to be reviewed in 2023.

In **Victoria**, a discussion paper was produced: EPA Publication 2002.3 (June 2008): Discussion Paper – Environmental Offsets. A 'safety factor' of 1.5:1 total nitrogen is used for offsetting projects (https://waterportal.com.au/swf/images/swf-files/10tr16-001-swf-alluvium-water-quality-offsets-framework-final.pdf). In 2011, the Victorian water industry and EPA produced a paper proposing a framework to fill the existing gap between the legislative ability of EPA to authorise the use offsets and the ability of water corporations to successfully make cases for the use of environmental offsets during EPA processes (EPA and Victorian Water Industry 2011). During 2014/15 Western Water and the Smart Water Fund engaged Alluvium to support the further development of a water quality offsets framework (https://www.waterportal.com.au/swf/images/swf-files/10tr16-001-swf-alluvium-water-quality-offsets-framework-final.pdf).

In **New South Wales (NSW)**, the EPA produced a concept paper in 2002 on green offsets for sustainable development (https://www.environment.nsw.gov.au/resources/greenoffsets/greenoffsets.pdf). A key focus for nutrient offsetting has been the greater Sydney region, including the Hawkesbury/Nepean River system. In 2010, EPA produced a nutrient management strategy for this river (https://www.environment.nsw.gov.au/-/media/OEH/Corporate-Site/Documents/Water/Water-quality/lower-hawkesbury-nepean-river-nutrient-management-strategy-10225.pdf). There is a regulatory framework with nutrient load caps in zones throughout the river system. Polluters must undertake works to demonstrate that nutrient loads do not exceed caps, and nutrient offsetting or trading is a mechanism for this. These caps vary and unlike Queensland there is no single offsetting ratio. In some states there can be other policy drivers, such as the need for catchment improvement programs and need to maintain environmental flow.

South Australia has no policy trigger, but the environmental protection act has environmental performance agreements, which are contracts involving polluters such as utilities.
Across the states, projects are typically reviewed on 5-year timelines. The rationale for this is that it takes some years for engineering works to bed down, and trees to grow to a sufficient size to be playing a significant role in bank stabilisation. Additionally, 5 years allows time for one or more larger rainfall events to occur to test the stability of the restoration area.

Key findings from interviews

This component of the study involved interviews with people involved in nutrient offsetting across Australia. It involved a range of questions, developed in consultation with the WSAA steering committee for the project (for details see Appendix 1).

- A total of nine people employed at utilities, three people employed at state government, three people employed as researchers and people at a range of other relevant organisations, e.g., Port of Brisbane environmental managers, consultants, who have been involved in nutrient offsetting were interviewed about their experiences, including what worked and what didn't.

- Key findings include that there is no coordinated scheme for nutrient offsetting, either at the Federal or State level, with projects occurring on an ad hoc basis. Based on the interviews, offsetting projects that have been implemented were judged to be successful in terms of being able to achieve the nutrient reduction goal, based on modelled catchment nutrient load reductions, and in terms of costs relative to hard infrastructure alternatives. These projects have also driven innovative thinking about solutions, which is critical to a future green economy.

- There was general consensus amongst utilities and government that a coordinated framework with an independent entity taking a role in coordinating nutrient offsetting at a regional scale would be beneficial, provided this was done cost-effectively and with minimal bureaucracy.

- Nutrient offsetting is typically considered by utilities when other on-site options for nutrient treatment, particulatly nutrient loads, are insufficient or unfeasible. This can include both compliance requirements, and corporate environmental performance targets. As such there is a sequence of steps typically undertaken by utilities (see Figure 2 below) commencing with:
    - Optimising the water treatment processes, e.g., enhanced control mechanisms.
    - Then identifying options for on-site treatment, e.g., irrigation with wastewater, use of constructed wetlands.
    - If required, catchment mitigation strategies are then examined.
    - In some international examples, nutrient trading has progressed to a more coordinated regional approach.

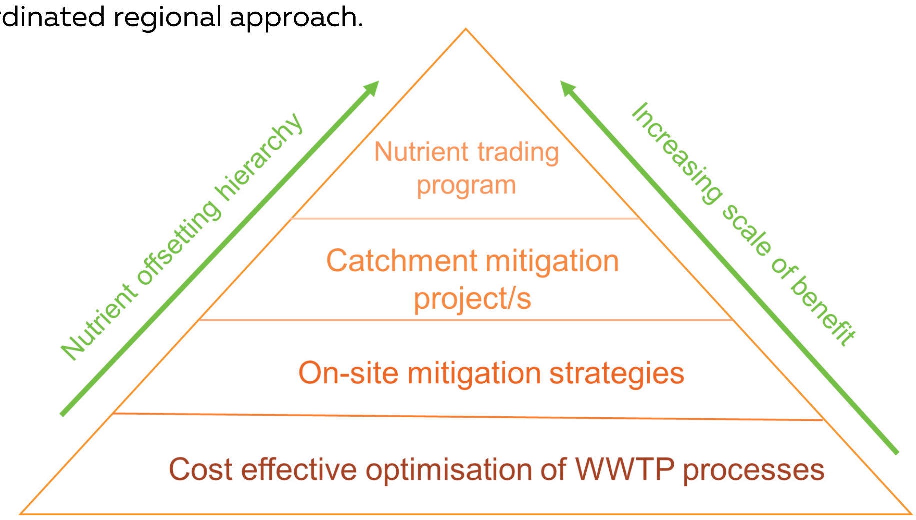


Figure 2: Pyramid of approaches to nutrient reduction into waterways

- Utilities typically take the risks and outlay the funds for offsetting projects, both to get projects up and running, and for ongoing maintenance and monitoring costs. State governments take little of the risk, instead focussing on their regulatory role. Utilities have raised concerns about the complexity of navigating nutrient offsetting, especially in their interactions with state governments.

- The catchment mitigation component of nutrient offsetting projects in Australia typically takes one of two approaches:
    1. Streambank restoration, primarily in rural land, which is focussed on streambank stabilisation, tree planting and exclusion fencing;
    2. Urban stormwater mitigation strategies, such as bioretention basins, swales, and channel naturalisation.

  For approach 1, effective engagement with landholders is essential to gain access to land and ensure the ongoing maintenance of sites, while approach 2 is typically managed at a local government level to ensure that there is an effective strategy for implementation and ongoing maintenance of infrastructure.

- Streambank restoration involves greater risk as it relies on ongoing effective relationships with landholders to ensure ongoing benefits from the projects. Interviewees have identified a lack of clarity around who owns the offsetting works, and there is legal ambiguity as to who 'owns' the mitigation works, given that the landholder owns the land. As a result, there can be risks of non-compliance if the relationship between landholder and water utility is not effectively maintained. Often catchment management groups are engaged to facilitate the relationships with landholders. There are also examples where grants have been awarded to landholders so they can control the remediation process, creating greater ownership of the project and benefits.

- The benefits of nutrient offsetting projects are typically assessed using modelling approaches, both quantifying the efficacy of bank stabilisation and other remediation measures in the riparian zone and assessing the water quality improvements in rivers/creeks. Usually, consultants are engaged to undertake the modelling. There are very few examples where long-term monitoring has also been done. This is typically for two reasons:

  1. It is more expensive than modelling.
  2. It can be difficult to measure an attributable change in water quality when there is so much natural variability in aquatic systems.

- Due to the paucity of data before and after offsetting, there has been limited calibration of models for local conditions, resulting in a significant level of uncertainty in the model outputs. Therefore, there is a concern from regulators that models may be used inappropriately in assessing the effectiveness of an offsetting project. This is an area that warrants further investigation.

- In Australia, nutrient offsetting may give the greatest benefit in catchments where there are significant areas of erosion, which is typically rural areas.

- Often state governments specify that there is the need to do offsetting works in the same catchment and upstream of the discharge point from wastewater treatment plants (WWTPs). At times this limits the effectiveness of offsetting, e.g., where there is limited scope for offsetting upstream or where other stressors confound the ability to achieve benefits from offsetting.

- The assimilative capacity of receiving water environments for nutrient inputs is poorly understood making it difficult to set targets for ecological health, and hence assess nutrient offsetting projects against these targets. Additionally, there is currently a lack of clarity in comparing the effect of wet (catchment runoff) vs. dry season (constant inflows from wastewater) nutrient inputs on water quality and ecosystem health. This includes the need to take a long-term view of the cumulative effects of wet season inputs on waterways. The nutrient equivalency and delivery ratios need to be more accurately determined across catchments.

- The transaction costs, particularly for utilities with implementing nutrient offsetting, are typically not measured and are likely to be quite high. Additionally, these costs occur over relatively long timeframes, due to the need to maintain the offsetting sites and undertake modelling or other requirements. A robust analysis of the cost effectiveness of nutrient offsetting should be determined.

- The need for an investment rolling plan was identified so markets can be established with a long-term plan.

- There are many examples of co-benefits when undertaking nutrient offsetting which include: benefits for local communities, e.g., engaged landholders who want to do more catchment restoration; reduced water and sewage treatment costs for consumers; improved recreation; improved real estate values; job creation via undertaking mitigation works. Societal benefits also create social capital for utilities. Typically, these societal benefits have not been costed in a formal way and this is an area that warrants further investigation to fully value the co-benefits.

- Offsetting also results in co-benefits for the environment, both in the immediate area, and downstream. This includes carbon sequestration from increased vegetation, improved biodiversity of species, and numbers of animals and plants using riparian and river ecosystems, flood mitigation, and reduced sediment loss and associated silting up of rivers, reservoirs and coastal environments. Utilities have flagged many examples, typically anecdotal, where restoration of riparian habitats has had significant benefits for ecosystem health and biodiversity. Typically, these have not been costed in a formal way and this is an area that warrants further attention.

Two examples of successful nutrient offsetting projects include Urban Utilities (southeast Queensland) which have taken a lead role in promoting nutrient offsetting in Queensland. They undertook two pilot projects to locally offset their WWTP riverine discharges in Beaudesert (commenced 2013) and Laidley (commenced 2018) in southeast Queensland.

They undertook streambank restoration nutrient management actions with rural landholders. Project environmental approvals were issued from the Environmental Regulator (DES) in accordance with the point source water quality offsets policy. The DES policy requires offset projects to be located upstream of the WWTP outfall and a minimum 1.5:1 total nitrogen ratio to ensure the project delivers an overall improvement in water quality at the WWTP outfall location. Based on the ongoing monitoring and evaluation program, the green streambank assets are performing according to upfront modelling estimates with nutrient offset credits being validated every year. The offset projects lowered regulatory risks and allowed Urban Utilities to defer major upgrades at existing WWTPs, providing more time to undertake robust planning to optimise WWTP processes and cater for future population growth. Urban Utilities paid all the costs associated with offsetting projects.

Another example of a successful offsetting project is Coliban Water's WWTP at Kyneton, Victoria, (the Kyneton Water Reclamation Plant (WRP)). It had non-compliance issues with its EPA-issued discharge licence, primarily nutrients. The compliance issues were exacerbated by a reduction in the volume of passing flows in the receiving water (the Campaspe River) and increasing inflows into the WRP. A proposal was submitted to EPA focussed on riverbank remediation works at several riverfront properties upstream of the discharge point, with predicted 150-200% nutrient offset. A 5-year river health monitoring program was also initiated. EPA was initially supportive but then decided not to support the inclusion of environmental offsets as part of an amended discharge licence. Despite this, Coliban Water chose to complete the riverbank remediation works. Since the completion of the works, and the delivery of the reports for the river health monitoring program, EPA have become more supportive of potentially recognising these offsets under any amended discharge licence for the WRP.

A summary of the processes that we suggest are required in nutrient offsetting projects is outlined in Figure 3. It identifies key steps and feedback loops within each step and provides information on key considerations and knowledge gaps to fill. It is clear that undertaking offsetting projects is complex and mechanisms for streamlining are needed.Challenges are identified in Table 1.

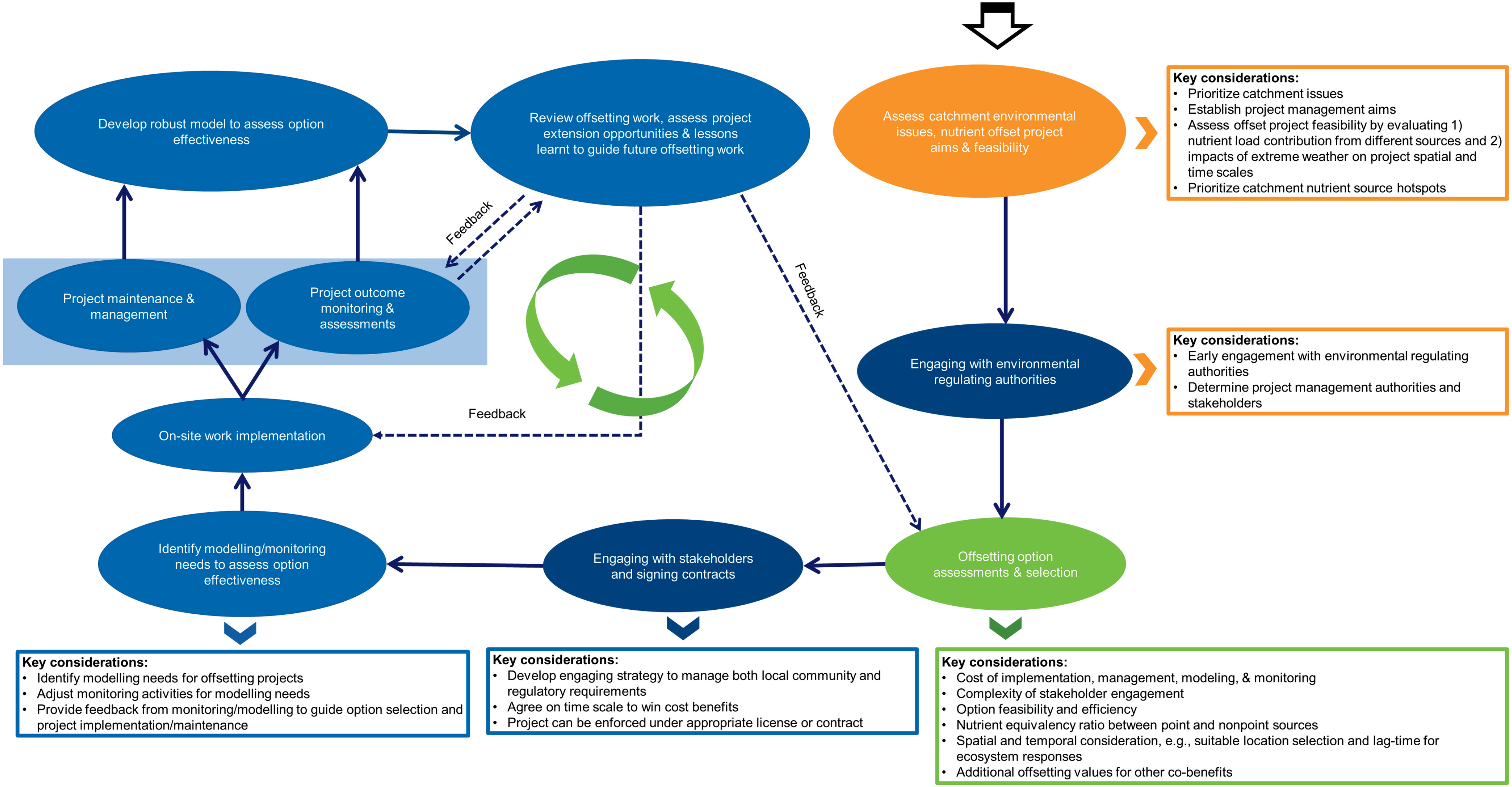


Figure 3: A summary of the processes involved in nutrient offsetting projects

Table 1: Most frequently identified challenges with undertaking offsetting programs

SCIENTIFIC KNOWLEDGE GAPS	CHALLENGES WITH GOVERNMENT OVERSIGHT	CHALLENGES FOR WATER INDUSTRY INVESTMENT
Optimising mitigation for maximum benefit, including identification of optimal sites and strategies for maximum ROI	Government role is regulatory rather than working collaboratively to make the scheme work, meaning that all the risk is with utilities	Significant costs in coordinating projects with multiple stakeholders without clear information on the maximum 'bang for buck'
More and better data to calibrate/validate models estimating nutrient load reduction from mitigation	Lack of an offset framework and guidelines. Lack coordination and consistency of policy within government, creating complexity	Lack of good quality data on return on investment, and risk to water industry from uncertainty in model outputs
Scientific basis needed for more robust equivalency and delivery ratios	Little engagement by local governments (except where they manage wastewater facilities) and lack of coordinated approach by local governments	Uncertainty in ratios may result in inequities and lack of confidence in investment
Methods for assessing the costs and benefits of offsetting, including co-benefits	Lack of inclusion of stormwater in many schemes	Riskier for investment compared with grey infrastructure
Lack of assimilative capacity/sustainable loads for most catchments	Insufficient information on government targets, e.g., river assimilative capacities, wet season vs dry season targets, identification of where in the catchment-to-sea continuum should environmental benefits occur	Lack of offset project feasibility assessment to achieve the nutrient load management goals

Literature review – key findings

Another component of the nutrient offsetting project was a review of the international literature.

- There are excellent international examples of how nutrient offsetting could work as a market where there is an overarching organisation facilitating offsetting., e.g., the South Nation Conservation Association that managed the Ontario South Nation River trading program in Canada (Figure 4), Pennsylvania Infrastructure Investment Authority and Pennsylvania Department of Environmental Protection that managed the Pennsylvania nutrient credit trading program in Chesapeake Bay, USA (Figure 5, Boleman and Jacobson, 2021). In some cases, a fund is set up by an independent entity that can be managed by multiple stakeholders, including point and non-point source representatives (Figures 4, 6 & 7). In other cases, local governments provide all the key information and a website where people can trade freely (Figure 5). There are also examples of the use of a credit bank which buyers pay into which protects them against mitigation failures, e.g., floods, droughts. In some cases, the government puts up the funds/loans initially for mitigation which allows buyers and sellers to apply for funds, and in the case of buyers they pay the government back at a later stage. It provides capital when it might not otherwise be available in the short term to kickstart markets. Additionally, carbon credits can be included in the scheme, e.g., when planting trees, providing a credit against the costs of nutrient offsetting. This all assumes that credits have a tradeable value.

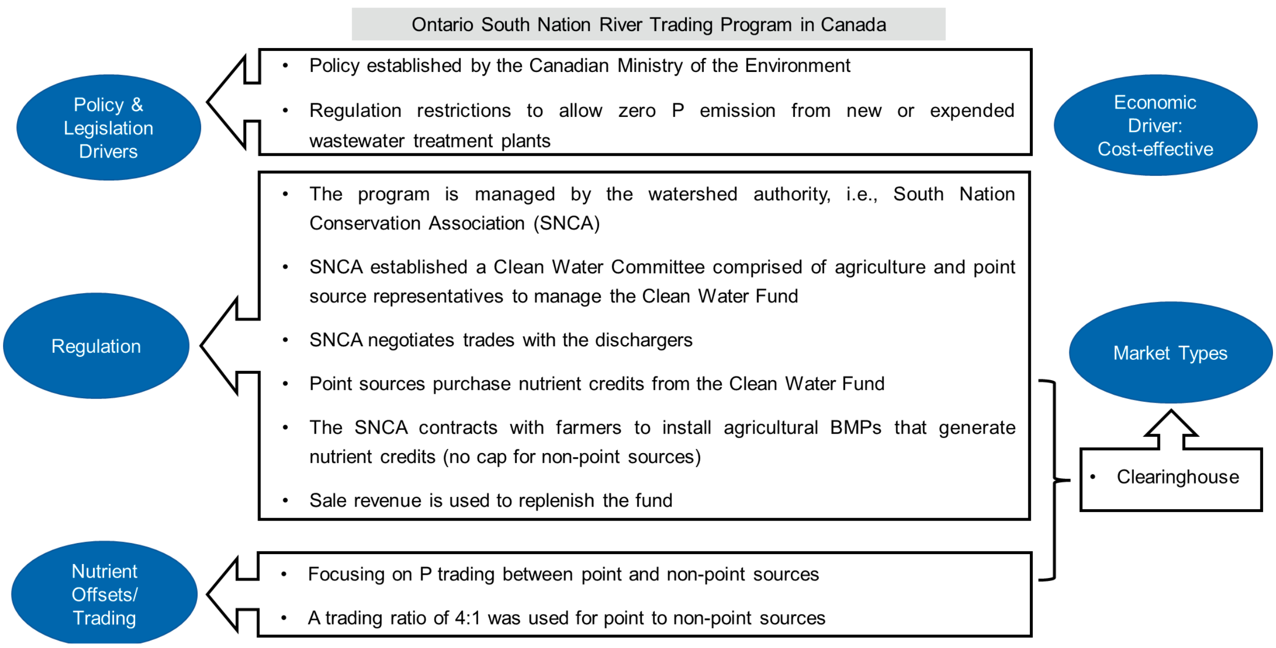


Figure 4. Case study: Ontario South Nation River trading program in Canada

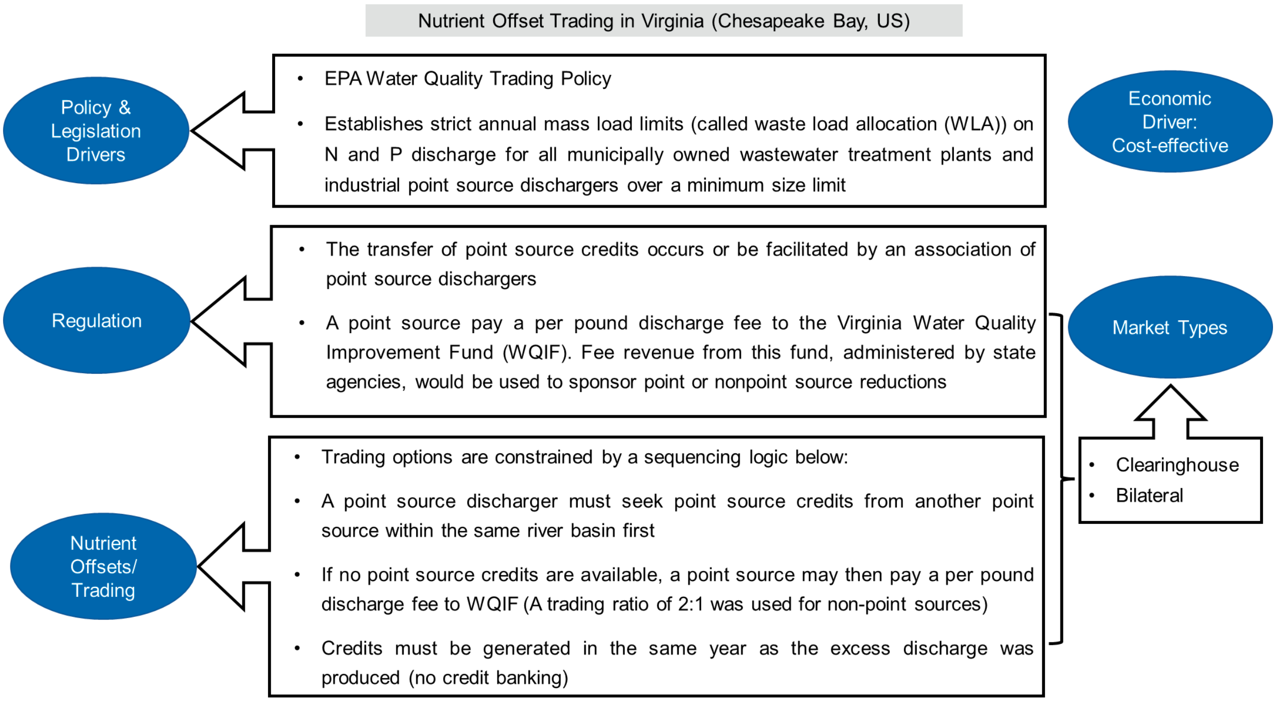


Figure 5. Case study: Nutrient offset trading program in Virginia (Chesapeake Bay, USA)

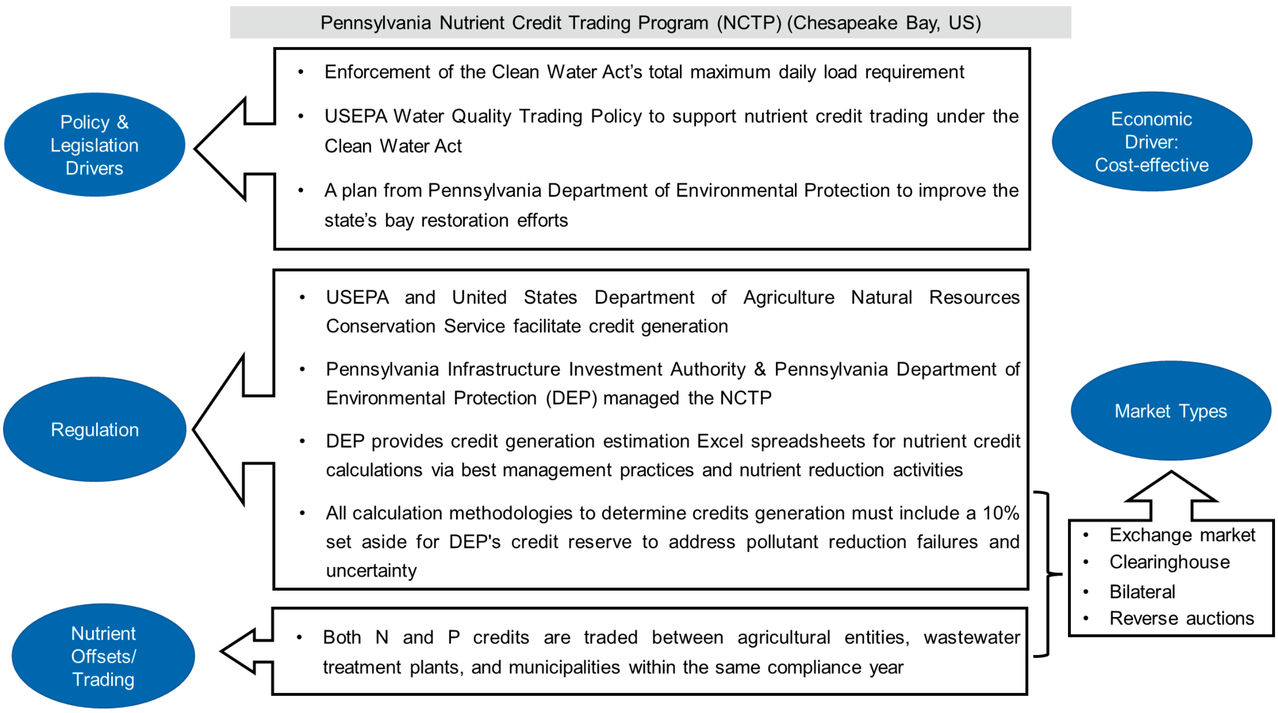


Figure 6. Case study: Pennsylvania nutrient credit trading program in Chesapeake Bay, USA

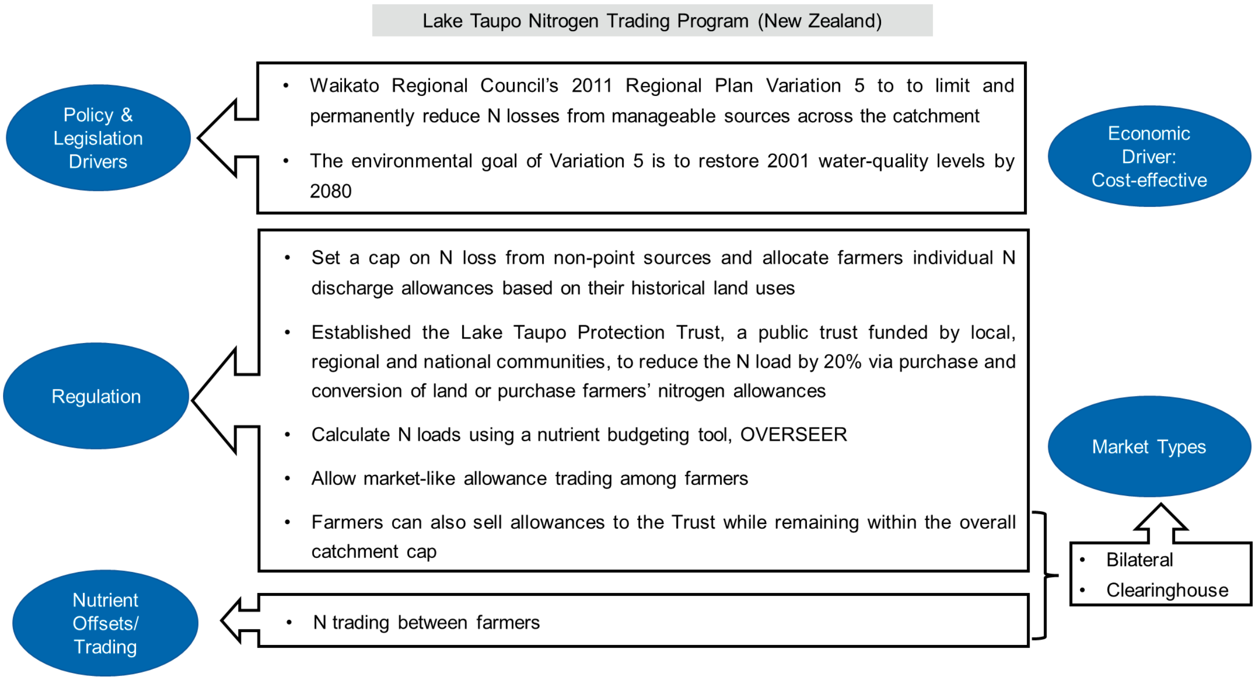


Figure 7. Case study: Lake Taupo nitrogen trading program in New Zealand

- The analysis of the literature shows that successful international nutrient offsetting examples (measured by nutrient load reduction to waterways) have sufficient drivers for both nutrient credit buyers and sellers to be involved in the offsetting market. Drivers for buyers are both from government policy/regulation agencies that set nutrient loads capping, as well as ensuring the cost-effectiveness of implementing catchment approaches to meet the nutrient discharge limit. Incentives for sellers are typically about getting more revenue from implementing nutrient reduction interventions, compared to their business as usual. International studies have shown that nutrient offsetting programs should have buy-in from local government, the regulated point-source polluters, and other stakeholders within the catchment. Before going to the expense of developing a water quality trading program, it is recommended that the relevant bodies, either governmental or nongovernmental, ensure these factors are in place.

- Cost estimates have been ascertained for different nutrient load reduction options within catchments (US EPA, 2001), e.g., WWTP upgrades, stormwater treatments, and catchment restoration. Some studies suggest that the cost to reduce the same amount (e.g., per kg) of point source and stormwater nutrient load was much more expensive than non-point source nutrient load reductions (Hall, 2012; Lötjönen et al., 2021; Shortle, 2012). Allowing nutrient offsets between point and non-point sources has been estimated to reduce the cost of implementing water quality standards in the US by $140–235 million USD annually (US EPA, 2001). Some nutrient offset programs demonstrated an overall nutrient cap goal for a catchment at a reduced cost for a point source offset, e.g., nutrient offset projects in North Carolina in the US (US EPA, 2008).

- Studies have also shown co-benefits outside the offset scheme, such as sediment reduction, carbon storage, groundwater recharge, flood retention, and habitat and biodiversity protection, which further improves the health of the terrestrial and aquatic ecosystems (Cole et al., 2020; Mueller et al., 2019). For example, studies have shown that revegetating 30 m width of a riparian buffer not only removed 50–70% of total nitrogen (TN) in the runoff, but also removed approximately 80% of sediment and total phosphorus (TP) loads (Hoffmann et al., 2009; Mayer et al., 2007; Sweeney and Newbold, 2014). However, some co-benefits, such as increased aquatic biodiversity and functionality might need longer response times, compared to the timeframes for nutrients and sediment reduction (Rinne, 1999; Roni et al., 2002).

- Internationally, it is acknowledged that the models used to assess the effectiveness of nutrient reduction from mitigation works have a significant level of uncertainty. However, countries such as USA have the advantage of already having significant background information on soil, water quality and hydrological characteristics to reduce model uncertainties. Therefore, successful nutrient offsetting programs typically have consistent and standardized estimation methodologies developed for nonpoint source actions. In Australia, there is less information to calibrate and/or validate models. Typically, models are calibrated with soil erosion information from LIDAR or satellite images, and it is assumed that nutrient loss to waterways is associated with soil erosion. However, field studies have shown that soil erosion rates do not necessarily equate to nutrient loss rates. Therefore, soil nutrients from mitigation sites should be measured. Additionally, validation or calibration with existing/background water quality data is important to reduce uncertainty for nutrient load estimation from catchments.

- Another uncertainty for the industry is the offset or trading ratio between point and non-point sources of nitrogen or phosphorus, which is typically based on both the equivalency and delivery ratios. It is acknowledged in the literature that this is a major knowledge gap. We could not find any evidence in the literature that the effect of different sources on ecosystem health, for the same TN or TP load, had been validated. A current ARC Linkage grant (CI: Burford, Griffith University) is currently examining equivalency ratios. More research is needed on delivery ratios.

Recommendations

Overall, this study has identified many positive elements of nutrient offsetting in Australia but also many impediments to further expansion of the approach. Therefore, there are several recommendations from this study:

- There was consensus amongst utilities and government that a coordinated framework with an independent entity taking a role in coordinating nutrient offsetting at a regional scale would be beneficial, provided this was done cost-effectively and with minimal bureaucracy.

- It is suggested that both national and international experts who have experience with nutrient trading give presentations to the water industry and government stakeholders.

- Following on from this, greater coordination is needed within the water industry to create an improved scheme and advocate within state and/or federal governments. This could include setting up a committee to advise on strategies as well as advocating on behalf of the water industry with government.

- International studies have shown that successful factors of nutrient offsetting programs include buy-in from local government, the regulated point-source polluters, and other stakeholders within the catchment. Before going to the expense of developing a nutrient trading program, it is recommended that the relevant bodies, either governmental or nongovernmental, ensure these factors are in place. Therefore, the advocacy and shared learning processes are critical.

- Successful nutrient offsetting programs internationally typically have consistent and standardized estimation methodologies developed for nonpoint source actions, and this needs to be examined and developed in Australia.

- There is a need to support projects that fill key scientific knowledge gaps to reduce risk and increase confidence in nutrient offsetting. This would include improved data and models, optimising sites for remediation and effective quantification of co-benefits. The range of benefits would cover biodiversity, sediment reduction and carbon capture, as a first step to integrate green solutions to deliver the best outcomes for the water industry.

Appendix 1 – Results of interviews with Australian participants in nutrient offsetting

A1.1 Interview process

In this Appendix, a more detailed explanation of the interviews undertaken with individuals in the water industry and government regulators in Australia, as well as individuals within the university sector and consultants, is given (Table A1). The questions outlined below were shared with the Steering Committee prior to interviews commencing.

The questions asked of the interviewees were:

1. Have you had/or are being involved with some aspect of nutrient (and/or carbon) offsetting?
2. If you were or are currently involved in a project or planned project, give details.
3. When did planning for the project take place?
4. Do you think the project had or is having or likely to have significant benefits?
5. Have there been key learnings of the project so far?
6. If you were doing this again, what would you have done differently?
7. What do you think are the key knowledge gaps or uncertainties that need to be addressed for offsetting to be more effective?

The following people were interviewed:

Table A1-1: List of people interviewed and status of implementation. M&E = monitoring and evaluation

Organisation	Contact person	Relevant scheme/s	Status
Logan Water (Water utility)	Kam Akrami	Cedar Grove	Implemented In M&E phase
Port of Brisbane	Craig Wilson	Mulgowie (Laidley Creek), Downfall Creek	Implemented In M&E phase
Urban Utilities (Water utility)	Cameron Jackson	Beaudesert Mulgowie (Laidley Creek)	Implemented In M&E phase
Unitywater (Water utility)	Luisa Magalhaes	Pine River Mooloola River Caboolture River (Bellmere) Yandina wetlands,	Pine and Mooloolah being investigated Others are implemented in M&E phase
Goulburn Valley Water (Water utility)	Kirsten Hogan	Kilmore project (Kilmore Creek)	Implemented In M&E phase
Melbourne Water (Water utility)	Sharyn Rossrakesh	Urban upgrades	Implemented In M&E phase
Sydney Water (Water utility)	Jenny Rogers	Hawkesbury/Nepean Blue Mountains	Still being implemented
Hunter Water (Water utility)	Louise McKenzie	Conjuway Creek, Paxton	Implemented In M&E phase
Townsville City Council (Water utility)	David Manning (previous employee)	Bohle Creek, urban stormwater upgrades	Implemented In M&E phase
Coliban Water (Water utility)	David Sheehan	Kyneton Water plant	Implemented In M&E phase
Cairns City Council	Lynne Powell	Potential applications in urban areas of Cairns	No projects yet

Organisation	Contact person	Relevant scheme/s	Status
SA EPA (Regulator)	Shaun Thomas	Urban stormwater upgrades	Implemented In M&E phase
Victoria EPA (Regulator)	David Robinson	Whole of Victoria perspective	
NSW EPA (Regulator)	Matthew Hart	Whole of NSW perspective	
Qld DES (Regulator)	Ian Ramsay	Whole of Qld perspective	
Griffith University (Researcher)	Jim Smart	GBR research projects on nutrient trading and environmental accounting	
Aurecon Consulting (Researcher)	Abel Immaraj	QWMN project on nutrient offsetting twinning	
UNSW (Researcher)	William Glamore	Restoration projects with utilities	
Green Collar	Carole Sweatman	Voluntary reef credicts scheme, Great Barrier Reef catchments	Implementation commenced, ongoing

A number of other people were contacted, and interviews were requested. There was either no response to repeated requests, or they felt they had insufficient experience with nutrient offsetting to contribute meaningfully.

The interviews identified a wide range of approaches to undertaking nutrient offsetting, as well as identifying successes and failures, both at a project level and a state level. Key constraints and opportunities were also identified which are key to ensuring that nutrient offsetting will be a viable option and will expand significantly as an activity in the future. There were some excellent examples of projects that have been very successful, e.g., projects with Urban Utility, Unitywater, Goulburn Water, Hunter Water, but the slow rate of implementation of new schemes across Australia suggests some key impediments that still need to be addressed and possibly other constraints.

A1.2 Nutrient offsetting projects in Australia

There are a range of projects throughout Australia using nutrient offsetting to offset WWTP nutrient discharge. This includes projects initiated by Urban Utilities (Beaudesert, Laidley Creek), Logan Water (Logan River), Unitywater (Caboolture River), Goulburn Valley Water (Kilmore) and Hunter Water (Paxton). These projects undertook restoration work in rural or semi-rural catchments which focussed on bank stabilisation, riparian tree planting and fencing. There were several urban projects as well, e.g., Melbourne Water. Other projects are still in the scoping or early construction phase, e.g., Sydney Water. The sections below integrate the findings of all these studies to provide information on the range of approaches taken, and their effectiveness. A range of studies in various phases of implementation throughout Australia, focussed primarily on those undertaken by utilities.

Logan Water (Utility)
Logan Water undertook an offsetting project in the Logan River, southeast Queensland to offset discharge from their WWTP at Cedar Grove. This was located on agricultural land (cattle grazing) 25 km upstream of the WWTP, in collaboration with landholders. This involved stabilisation and revegetation. There was an agreement with Queensland Department of Environment and Science (DES) within the nutrient offsetting policy. The estimated offsetting benefit was 2.89 tonnes TN per year and 2.89 tonnes TP offset. This compared with 700 kg TN allowable WWTP discharge. Logan Water paid all the costs associated with offsetting. Offsetting was done in concert with on site nutrient reduction works, i.e. constructed wetland.

Port of Brisbane (Port operator)
This was not technically an offsetting project but was driven by a need to treat stormwater at the Port in order to comply with the Queensland State Planning Policy (State Interest Water Quality Supplementary Implementation Guideline (February 2021)) regarding stormwater treatment, i.e., reduce sediment (80%), phosphorus (60%) and nitrogen (45%).
Historically the Port used traditional onsite treatment mechanisms such as bioretention basins, swales etc. Given this was a very expensive option they explored alternatives to treating sediment, nitrogen, and phosphorus in the catchment. Sites for remediation were examined and the Lockyer Valley region, southeast Queensland, was identified as a hotspot for erosion. They ran a pilot study in Laidley Creek at Mulgowie in 2017 and demonstrated a significantly better sediment and nutrient reduction outcome than onsite treatment. They have now run three projects in Laidley Ck, and another project in partnership with Brisbane City Council and Council of Mayors (SEQ) at Downfall Creek in 7th Brigade Park, Chermside. Port of Brisbane generate investment through a stormwater treatment levy on developers, however stormwater is still treated onsite using passive structures and gross pollutant traps.

Urban Utilities (Utility)
Urban Utilities (southeast Queensland) have taken a lead role in promoting nutrient offsetting in Queensland. They undertook two pilot projects to locally offset their WWTP riverine discharges in Beaudesert (commenced 2013) and Laidley (commenced 2018) in southeast Queensland. They undertook streambank restoration nutrient management actions with rural landholders. Project environmental approvals were issued from the Environmental Regulator (DES) in accordance with the point source water quality offsets policy. The DES policy requires offset projects to be located upstream of the WWTP outfall and a minimum 1.5:1 TN ratio to ensure the project delivers an overall improvement in water quality at the WWTP outfall location. Based on the ongoing monitoring and evaluation program, the green streambank assets are performing according to upfront modelling estimates with nutrient offset credits being validated every year. The offset projects lowered regulatory risks and allowed Urban Utilities to defer major upgrades at existing WWTPs, providing more time to undertake robust planning to optimise WWTP processes and cater for future population growth. Urban Utilities paid all the costs associated with offsetting projects.

Unity Water (Utility)
Two offsetting projects for WWTPs are underway in Caboolture River and Yandina Wetlands, north of Brisbane. For the Caboolture River, they chose streambank restoration at nine sites along the river. There were significant modelled benefits from erosion reduction. Unitywater paid all the costs associated with offsetting. There was an agreement with DES within the nutrient offsetting policy. For Yandina, Unitywater purchased two lots of former cane farming land and transformed it into a wetland opened to the public.

Goulburn Valley Water (Utility)
WWTP discharge at Kilmore, Victoria was offset by remediating the riparian zone of a rural creek by fully funding fencing, weed (gorse) control, revegetation. This was conducted on private land. It was done on top of some upgrades of a WWTP. A total of 945 kg phosphorus p.a. needed to be offset and they achieved 1042 kg P. If upgrades to the WWTP had been done it was estimated that it would cost $50M, while the offsetting project cost around $15M.

Melbourne Water (Utility)
Conditions have been put on new subdivisions to ensure nutrient loads from stormwater are managed in the Melbourne urban area. Mitigation works can be done on site or off site. Offsetting funds go to Melbourne Water for use in catchments that flow into Port Philip Bay. The focus is on water quality benefits in Port Philip Bay, rather than on local catchments. Offsets are primarily in urban land. Nutrient offsetting is not, however, in the Victoria planning provisions.

Sydney Water (Utility)

Technological improvements have seen a dramatic reduction in nitrogen and phosphorus discharge from WWTPs but the increasing population in greater Sydney means that further upgrades will require advanced technology that is prohibitively expensive. The NSW EPA has developed a framework for regulating nutrient discharges into the Hawkesbury-Nepean River from wastewater treatment plants that includes offsetting and trading schemes. The proposed framework, which comes into effect on 1 July 2024, includes equivalency/delivery ratios between 3:1 and 12:1 depending on the type, reliability and location of the offset. Sydney Water is constructing pilot offset projects to test the viability and effectiveness of the new framework. The pilot projects include a bank stabilisation at Camden and a raingarden biofilter at Glenbrook.

Hunter Water (Utility)

Hunter Water, NSW needed to tackle their Paxton WWTP nutrient discharge, so they developed an effluent management strategy. This identified offsetting as an option for the ephemeral stream, i.e., Conjuway Creek. The project was not licensed as a nutrient offset scheme because the loads were not high enough to require offsetting. Offsetting was done on rural properties, focussed on riparian and erosion works. Hunter Water set up a series of grant rounds which landholders applied for, and they were responsible for implementing the works on their properties. The model estimate of the benefit was 632 kg nitrogen retained p.a., and 89 kg phosphorus retained p.a.

Townsville City Council (Utility)

The Townsville City Council, north Queensland, were having difficulties complying with requirements for nutrient discharges from their WWTPs. Queensland DES and the Office of the Great Barrier Reef put in funding to examine offsetting options using stormwater. A program of works has commenced including channel naturalisation and water sensitive urban design.

Coliban Water (Utility)

Coliban Water's WWTP at Kyneton, Victoria, (the Kyneton Water Reclamation Plant (WRP)) has had non-compliance issues with its EPA-issued discharge licence, primarily nutrients. The compliance issues have been exacerbated by a reduction in the volume of passing flows in the receiving water (the Campaspe River) and increasing inflows into the WRP. A proposal was submitted to EPA focussed on riverbank remediation works at several riverfront properties upstream of the discharge point, with predicted 150-200% nutrient offset. A 5-year river health monitoring program was also initiated. EPA was initially supportive but then decided not to support the inclusion of environmental offsets as part of an amended discharge licence. Despite this, Coliban Water chose to complete the riverbank remediation works. Since the completion of the works, and the delivery of the reports for the river health monitoring program, EPA have become more supportive of potentially recognising these offsets under any amended discharge licence for the WRP.

Reef Credit Scheme (water quality trading scheme)
Reef Credits are tradable water quality credits generated from sediment and nutrient pollution prevented from reaching the Great Barrier Reef. GreenCollar (a consultant company) partnered with the Qld government, Terrain NRM, NQ Dry Tropics to develop and set up Reef Credits. In 2017, Eco Markets Australia was established, Australia's first and only independent markets administrator. The Scheme has:

- Buyers (purchasers of Reef Credits, including government, corporates, philanthropic organisations)
- Sellers – the credits are generated through actions by farmers in the reef catchments
- Project Developers –Partnering with landholders to create projects, and determine technical requirements of registering, auditing and completing verification and generation of Reef Credits through Eco Markets Australia.

The scheme has already generated 42,000 Reef Credits – which is 42,000 kg dissolved inorganic nitrogen prevented from flowing to the GBR.

A1.3 Implementation strategies

A1.3.1 Sites and strategies chosen for offsite mitigation

The process for developing nutrient offsetting projects has occurred on an ad hoc basis with individual utilities typically taking a lead role throughout the lifetime of the projects.
The mitigation works for nutrient offsetting projects typically fall into two categories: urban and rural. Projects in rural areas rely on engagement with landholders to get their buy-in to allow restoration works on their river and creek fronts. Landholders can range from 'tree-changers' with acreage property, primarily for lifestyle reasons, through to extensive and intensive agricultural industries. There is a significant time cost in engaging these landholders throughout the relationship establishment phase as well as when mitigation, maintenance and monitoring activities occur. Additionally, as activities are occurring on private land, ensuring continuing access to that land is critical. The benefits of mitigation may be diluted by fragmentation, i.e., some landholders agreeing to be involved and other adjacent landholders declining. In some cases, all costs are borne by the industry partner, in other cases there is co-contribution from the landholders, e.g., awarding grants.

Mitigation activities undertaken in urban areas often use land adjacent to waterways that is not privately owned. This land may be owned by local councils, e.g., reserves. In large urban centres the focus for mitigation may be broader than individual creeks or rivers. In Melbourne and Adelaide, mitigation has focussed on safeguarding water quality and environmental values in coastal environments, e.g., Port Philip Bay. In Sydney, much of the focus has been the Hawkesbury/Nepean River system.

The most common mitigation actions identified were:

- Riparian tree planting.
- Riverbank stabilisation with engineering works, e.g., battering.
- Fencing to stabilise banks, protect vegetation and exclude stock.
- Other activities currently being undertaken include:
    - o Seaweed and oyster cultivation, both of which are in the trial phase to determine their efficacy in nutrient removal. Seaweed is used for dissolved nutrient removal, while oysters remove particulate nutrients.
    - o Offsite wetlands are also used in some cases, but some interviewees felt that they are expensive to maintain.
    - o Offsite rain gardens area also being trialled.
    - o In urban areas in some states, stormwater management may also be used. This involves approaches such as bioretention basins, swales, rain gardens, rainwater tanks, passive diffusion of runoff, and street tree watering.

A1.3.2 Collaborative arrangements

Nutrient offsetting projects involve a range of organisations and individuals. It is clear from the interviews that managing these collaborations is critical to the success of all projects and requires a considerable time investment. These collaborations include catchment management organisations such as Landcare and Healthy Land and Water. These organisations often have good relationships with landholders to facilitate negotiations. They may also have the expertise to undertake the mitigation works, e.g., bank stabilisation, tree planting, and may also undertake the maintenance works, such as weeding and watering to maximise the success of the mitigation. However, mitigation and maintenance activities are not always done by an external organisation, and sometimes landholders themselves may undertake these works. Another important role for catchment management organisations may be identification of suitable sites for offsetting schemes, based on assessment of areas where rehabilitation is most likely to be effective.

As mentioned in the section above, landholder collaborations are also critical. Interviewees identified that many landholders have been very positive about the benefits (and co-benefits, see below) of the projects, and this has facilitated greater engagement from other landholders.

Consultants may also be collaborators or contracted on projects, facilitating landholder engagement, undertaking on ground works and/or running catchment models. Researchers at universities and other research agencies may also be involved, undertaking trials of the effectiveness of different mitigation options, examining mechanisms for establishing nutrient markets, and providing data on equivalency ratios for nutrients when WWTP discharge and catchment runoff are compared.

The Reef Credits scheme appears to be the only system in Australia where a market has been set up. Consultants, the Queensland government, and natural resource management organisations partnered to set it up and Eco Markets Australia was set up as Australia's first and only independent market administrator. It is focussed on reducing sediment and nutrient loss from agricultural land with the purchasers of Reef credits being government, corporations and philanthropic orgnisations. The scheme only commenced a couple of years ago but already has already generated 42,000 Reef credits which is equal to 42,000 kg reduction in dissolved inorganic nitrogen flowing to the Great Barrier Reef.

A1.3.3 Monitoring and evaluation

An important component of offsetting projects is determining if they are working successfully. WWTP operators are usually required to maintain sites, and pay the costs associated with this. Typically, the buyers of nutrient offsetting must ensure efficacy of the sites for between 5 and 10 years. In some cases, sites are reviewed on a 5-year basis, then licenses may be renewed.

The main method used for assessing the benefits of mitigation is via modelling approaches. This involves models such as SOURCE (hydrological model) and BSTEM (Bank stability and toe erosion model) to both quantify the likely benefits of mitigation works ahead of time, and if done in concert with monitoring, to quantify the actual benefits after implementation. Technologies such as LIDAR (remote sensing with lasers) are used to estimate erosion control before, and in the years after implementation of mitigation works. On-ground assessments are also done to make estimates of factors such as tree cover, integrity of riverbanks, etc.

Monitoring of a reduction in nutrient concentrations in waterways, or measurements of improvements in other measures of aquatic ecosystem health as a result of estimated nutrient reductions, are rarely done. This is because monitoring is expensive and it can be difficult to detect change after mitigation due to other confounding perturbations, e.g., flooding, droughts, extreme temperatures. Therefore, there are risks that environmental improvements are not actually achieved, given the limitations of using models if they are not effectively calibrated.

In the Reef Credits scheme, there is an administrator that undertakes auditing of projects to ensure a standardised protocol and delivery of benefits.

A1.3.4 Other mitigation activities done in concert with offsite mitigation

Often offsite mitigation works are done in concert with other nature-based solutions on or adjacent to the site of the WWTP. There are a range of approaches including:

- Wetlands to reduce nutrients discharged from WWTP.
- Upgrading and optimising nutrient reduction strategies within existing plants, e.g., use of aeration on lagoon systems, nitrification enhancement including floating structures, alum dosing, advanced control systems.
- Irrigation of pastures with discharge water on adjacent land.

A1.3.5 Benefits of schemes (including co-benefits)

Many interviewees identified a range of co-benefits from their nutrient offsetting project. However, these co-benefits are often not assessed quantitatively. Qualitatively they appear to be an important component of demonstrating the success of nutrient offsetting projects. The co-benefits include:

- Creating high quality habitat for enhancing biodiversity. This includes the riparian zone, wetlands, and in-stream habitats. Benefits including an increase in bird and terrestrial plant numbers and diversity. Within waterways, aquatic plant and fish numbers and diversity may also increase. Additionally, riparian areas may have greater ground cover and structural diversity. Wildlife corridors may also be created.

- Reduction in erosion processes which not only reduce nutrient loss to waterways but also result in greater sediment stabilisation. Preventing this mobilisation is beneficial in reducing sand slugs in rivers, decreasing smothering of seagrass beds in estuaries and the nearshore, and reducing the frequency of dredging channels for navigation. In one proposed study in Adelaide, the timescale over which nutrient reduction from the WWTPs would be increased, so that these facilities could instead invest in reducing sediment from urban stormwater. This approach is being considered as sediment smothering of seagrass was considered a more serious environmental impact than the effect of nutrients in urban waterways.

- Planting trees and encouraging natural germination of existing seedstock increases carbon capture and may contribute to carbon offsetting.

- Nutrient offset projects may also provide a mechanism to reduce nutrient concentrations without reducing flow rates. This can be beneficial in creeks where the WWTP discharge is the main, and important contributor to flow in the creek. This ensures that creeks continue to support fish passage and habitat for multiple aquatic species.

- Additional benefits are the improved wellbeing of local communities who access waterways, as well as improving the social license of WWTP operators. In situations where the costs of mitigation were less than the estimated costs, or the costs of upgrading plants, local communities also benefit from reduced rates, or less of an increase in rates, for treatment of wastewater.

- State and local governments may also benefit from less erosion of landscapes, including roads, which require less funds expended to undertake repairs, as well as improvements in water quality for surface drinking water supplies, reducing human health risks and treatment costs.

As part of the analysis of the information gained from the interviews, a SWOT analysis was conducted to examine the strengths and weaknesses of nutrient offsetting, then to identify opportunities and threats going forward.

A1.4 Strengths and weaknesses

A1.4.1 Strengths of the current approach to the nutrient offsetting concept

- Multiple sites in Australia have shown a quantifiable net reduction in nutrients from catchment mitigation, typically with less cost expenditure than for WWTP upgrades. For example, the Kilmore project determined that the ratio of benefit was higher, i.e., 2.2:1 total nitrogen ratio, compared with the 1.5:1 required.

- There were many beneficiaries of nutrient offsetting:
    - o WWTP operators and their customers gain the maximum benefits as it allows them to continue to operate.
    - o Other beneficiaries are local councils, landholders, catchment management and land care groups, researchers, special interest groups, the public accessing the enhanced amenities, and state and local governments.
    - o Most importantly, in terms of the intent of nutrient offsetting, the environment benefits via measures such as increased biodiversity and improved water quality.

- Nutrient offsetting provides investment for catchment restoration that wouldn't otherwise be done primarily because it is too expensive.

- Reducing nutrients in a catchment, via offsetting, without reducing flow from wastewater discharge can provide low flow at times when there is no flow to assist fish passage and spawning, and as well as ensuring refugia and habitat are maintained in rivers and creeks.

- However, it should also be acknowledged that nutrient offsetting works best when WWTPs are not the dominant nutrient source for a catchment or subcatchment, otherwise it can be difficult to find mitigation sites where sufficient nutrients can be offset upstream.

- Nutrient offsets work well in those rural catchments where there are significant areas of erosion.

- Some studies have shown evidence that despite floods after mitigation has occurred, restored riparian zones can be largely maintained, rather than being washed away.

- Nutrient offsetting projects drive innovative thinking about green infrastructure and co-benefits. This innovative thinking is critical to a future green economy and valuing natural capital, including development of nutrient treatment approaches, e.g., seaweeds, oyster beds.

- Landholder engagement in offsetting has resulted in their desire to do more streambank restoration.

- Offset sites can provide a showcase for the public highlighting the benefits of natural capital and positive feedback has been received from public regarding project benefits, e.g., Hunter Water.

- In some offsetting approaches the science is quite well established, reducing uncertainty in determining the benefits, e.g., urban wetlands.

A1.4.2 Weaknesses of current approach to nutrient offsetting

Despite the benefits of nutrient offsetting outlined above, there has been limited uptake by the water industry across Australia. This is due to risks (discussed below) and weaknesses in the current approach. Some of these generic weaknesses are outlined below:

- There is a lack of a state government or national framework. Many interviewees felt that state governments were too risk averse. The focus was on strict compliance rather than working collaboratively to solve problems. There is also tension between the desire for development of urban areas within one sector of government vs the desire to regulate nutrient outputs from another government sector. Additionally, there is also a lack of coordinated approach by local councils, and between councils and state governments, making it difficult to develop generic protocols.

- In some areas thresholds for offsets are too flexible. Additionally, the lack of reviews of some projects mean that mitigation strategies were not being enforced.

- Lack of political will to embrace nutrient offsetting in some states means that there is more potential to set up carbon offsetting, with nutrient offsetting becoming the co-benefit, rather than the focus.

- Scientific uncertainties remain including issues around: nutrient equivalency between catchment runoff and wastewater treatment discharge; ability to monitor effectiveness of mitigation; insufficient monitoring and mitigation data to validate models for specific sites.

- There is a significant time investment and cost in setting up, maintaining and monitoring offsetting projects. Typically, this is borne by individual WWTP operators.

- It can take a long time to have measurable benefits from mitigation, particularly planting trees which take years to fully establish.

- The need to do nutrient offsetting in the same catchment as that for point source water input limits the ability to undertake works, especially if the main nutrient input is from point sources, or conversely if the main nutrient input is from non-point sources. This is also true if the catchment goes across multiple jurisdictions.

- Sometimes implementation and maintenance are done poorly – high quality providers need to be available to undertake the work.

- Offsetting may not work well in sites where there are significant environmental multi-stressors such that the benefits of reduced nutrients are not realised.

A1.5 Risks and opportunities

A1.5.1 Opportunities for improvements

Restoration

In many rural catchments there is plenty of scope for restoration/mitigation but sites need to be identified, ideally using a framework. Icon Water, for example, have a strategy for prioritising erosion hotspots for restoration in their three drinking water catchments to improve water quality (https://www.iconwater.com.au/water-education/sustainability-and-environment/sustainability-and-environment-programs/protecting-water-supply.aspx)

- Co-benefits can be significant if planned properly. In a green economy, nutrient offsetting should be integrated with co-benefits to maximise return on investment.
- Inaccuracies in estimating bank stabilisation need to be addressed.

Markets

- The voluntary reef credits scheme in the Great Barrier Reef catchments is a market-based example that is up and running and appears to be working to date. It may be a useful starting point for discussions about how the water industry could engage with nutrient offsetting in a more coordinated way.
- Non polluters with an interest in a social license to operate and/or developing green credentials could also be involved if a trading scheme is developed. However, this requires leadership and a framework.
- There is the need for a better approach to determining equivalency of point- and non-point source nutrients to improve the value proposition and certainty for buyers.
- Improved economic analysis and frameworks would assist buyers in assessing the value proposition. The current benefit estimation is not good enough.
- Having more cost-effective ways to manage restoration sites would improve the value proposition for buyers.
- More holistic assessments are needed for identification of whether mitigation should involve green vs grey infrastructure.
- Nutrient offsetting should include stormwater to maximise the benefits. This is already being done in some locations.

Research gaps

- Better engagement with researchers would assist in improving the uncertainties around the benefits of offsetting. Greater engagement by the Commonwealth government in promoting research in this area would assist. This approach would be particularly effective in areas adjacent to areas of key importance to the Commonwealth government, e.g., marine parks such as the Great Barrier Reef.

- More robust science needed for reconciling wet weather runoff vs dry weather runoff.

- Need to consider the longer-term impacts of nutrients from wet weather runoff, e.g., nutrient remineralisation in Moreton Bay.

- Much of the catchment mitigation involves stabilising banks to control sediment, with the assumption that this will also deal with nutrients. However, other research has shown that different soil types have different nutrient stabilisation characteristics. Better characterisation of soils and their impact would provide more robust measures of how much bank stabilisation is needed.

- There is a need for assimilative capacity data for rivers and creeks, not just loads and concentrations.

- Lack of information on what happens to nutrients as they travel through the aquatic system.

- Interplay between nutrient impacts and climate change needs to be examined.

Engagement

- There is an opportunity to improve projects with early engagement with all stakeholders. A generic strategy for this could be developed.

- Greater inter-agency collaboration needed.

- Better education of public is needed, especially ratepayers, of the benefits of offsetting, catchment restoration, improved ecosystem health, etc.

Monitoring and evaluation

- There is evidence that some wetlands may not give the nutrient reduction benefits that they were modelled to give, but there is insufficient monitoring to substantiate this.

- Studies should start with a good baseline of the background water quality condition of the creek/river where mitigation will occur, or a space-for-time comparison.

- Drones provide a useful tool to measure the success of bank stabilisation and riparian revegetation.

- Expert panels would assist in developing nutrient offsetting frameworks and identifying knowledge gaps.

- There is a lack of sufficient data to inform models.

A1.5.2 Threats to future success of offsetting

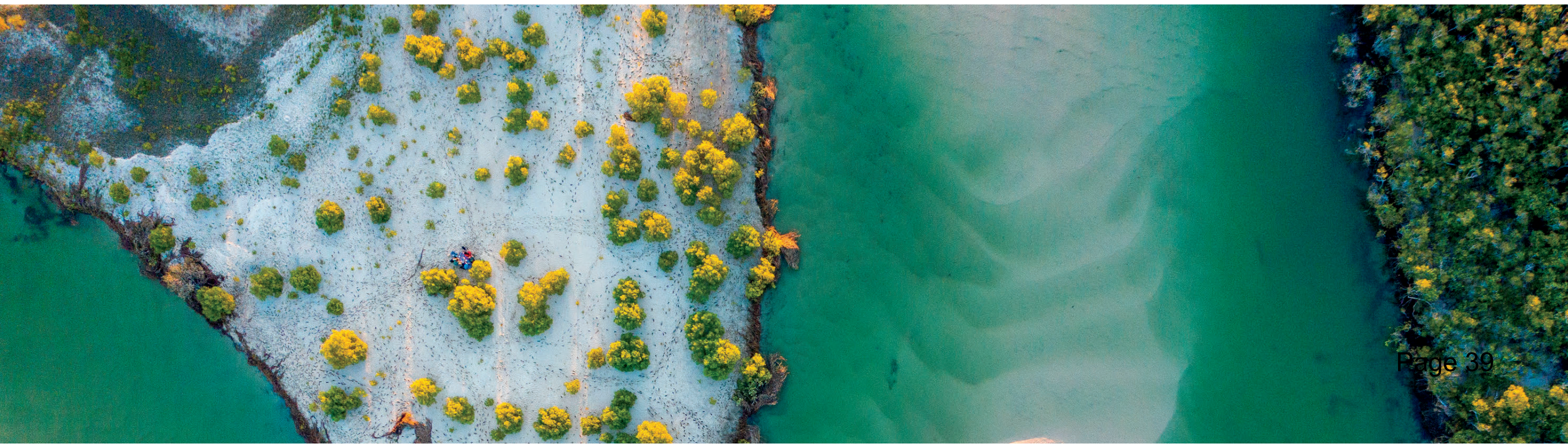
A wide range of threats and/or challenges were identified by interviewees. A summary of challenges most frequently mentioned are given in Table 2.

Policy framework

- The complexity of the nutrient offsetting approach, including too many government departments to deal with, limits investment. There is also irrationality in some government approvals as they are not set up for offsetting.
- State governments do not see the program as a partnership but prefer regulatory role which limits the effectiveness of the program.
- Governments unclear in their expectations of nutrient offsetting and its benefits, creating a risk for utilities.
- Lack of knowledge of targets for nutrient management in catchments.
- Clustered nature of mitigation, i.e., in catchments near WWTPs, increases risks for investors due to climatic issues, e.g., floods.
- Clarity needed around who owns the offset works.
- If offsetting aims to achieve nearshore or end of river benefits, it may come at the expense of headwater creeks.

Cost effectiveness

- Complexity of setting up offsetting projects, i.e., organising partners, quantifying benefits, monitoring and maintenance ongoing needs and managing those, means that new projects are not coming through the pipeline as might be expected.
- Timeframes to demonstrate benefits mean risks for businesses.
- Significant time commitment to work with state governments, consultants, councils, landholders. People like instant results, not having to wait many years to see benefits.
- An investment rolling plan is needed so markets can be established with a long-term plan and employment, e.g., traditional owner rangers and utilisation of surplus offsets generated given the precautionary approach applied to meet minimum requirements with contingency.



Scientific challenges

- Expertise for effectively maintaining restoration sites can be limited.
- Lack of knowledge of the effects of climate change in flow conditions, e.g., larger flow events damaging mitigation sites.
- Assessment of nutrient impacts stops at river mouth at some sites, e.g., Hawkesbury/Nepean, and in other situations focusses only on the nearshore at the expense of rivers.
- Concern about whether some current pilot scale projects will work at the larger scale.
- Questions about whether sewage bypass during wet weather is a confounding environmental issue.
- There is a need for better engagement with researchers to ensure scientific underpinning is available.
- More robust science needed around reconciling wet weather runoff vs dry weather runoff.
- Need to consider the longer-term impacts of nutrients from wet weather runoff, e.g., nutrient remineralisation in Moreton Bay.
- Much of the catchment mitigation involves stabilising banks to control sediment, with the assumption that this will also deal with nutrients. However, other research has shown that the impact of nutrients from different soil types on ecosystems varies considerably. Better characterisation of soils and their impact would provide more robust measures of how much bank stabilisation is needed.
- Need assimilative capacity data, not just loads and concentrations.
- Lack of information on what happens to nutrients as they travel through the system.
- Interplay between nutrient impacts and climate change need to be examined.

Appendix 2 – Literature review of nutrient offsetting approaches globally

This section involves surveying and summarising information from scientific papers, the grey literature and relevant websites. The focus would be on strategies used, benefits/effectiveness and learnings/challenges.

A2.1 Nutrient offsetting strategies and benefits

There are different approaches used to implement nutrient offsetting to address local water quality issues, such as one-time solo-source offset projects, third party regulated offsetting projects, and developed voluntary market-based trading (Figure 1). Those approaches differ in many ways, including participants involved (e.g., point sources and non-point sources), funding agencies, regulations, and organizations that are responsible for offset monitoring and effectiveness assessment. In this section, we will provide some case studies to highlight the different approaches to achieve local environmental goals via nutrient offsets.

A2.1.1 Nutrient offsets between point sources only

The simplest nutrient offsetting approach is the offsets between regulated point sources only. In this case, stormwater can be classified as a point source when it is discharged to a water body via piping or conveyances (https://hcb-1.itrcweb.org/strategies/). This point source nutrient offset approach establishes a cap on nutrient discharge from regulated point source dischargers but allows them to buy extra nutrient discharge credits from other point source polluters that are not using their established cap to achieve the same nutrient reduction goal for aquatic ecosystems. The motivation for this approach is more flexibility and cost-saving in achieving the same environmental goal. For example, nutrient credit trading between WWTP within the Albemarle-Pamlico Sound watershed, the second largest estuary in the US, has resulted in substantial cost-saving compared to the traditional command-and-control approaches (Doyle et al., 2014). The study of phosphorus discharge trading among 22 WWTPs in the Passaic River watershed, USA, was estimated to cost a modest 2–3% of the cost relative to a command-and-control approach (Sado et al., 2010). It is also clear from the research that the cost-efficiency assessment needs to be done for specific cases, rather than simply relying on information from other studies (Eheart et al., 1987).

There are also case studies where nutrient discharge permits were combined ("bubble licensing") for multiple point source dischargers so that they can work together more flexibly to meet nutrient discharge requirements and improve water quality. For example, the NEW Water Silver Creek pilot project in Green Bay, Wisconsin, USA, and the South Creek Bubble Licensing Scheme in the South Creek area of the Hawkesbury-Nepean River in the Sydney metropolitan area, Australia. (https://www.environment.nsw.gov.au/resources /licensing/lbl/lbl_module4.pdf).

A2.1.2 Nutrient offsets that engage non-point sources

When non-point sources were included in nutrient offset schemes, they are typically sellers of nutrient credits, not buyers. This is because non-point sources are normally under no regulatory obligation to reduce their discharge. The simplest approach to offset point source nutrient discharge from non-point sources might be one-time sole-source offsets which allow regulated point source polluters to solve their specific permit compliance problems (Shortle, 2012). They may choose to fund offsite catchment mitigations that can generate nutrient credits for up to 10 years. This type of nutrient offset project has nutrient credits over the project life. For example, the Minnesota Pollution Control Agency in the US allowed two industrial point sources (Rahr Malting Company and the Southern Minnesota Beet Sugar Cooperative) on the Minnesota River to use agricultural or other non-point source nutrient reduction BMPs to meet their nutrient cap requirements. In those cases, the point source polluters are responsible for identifying nonpoint source offsetting partners and ensuring the continuing performance of the mitigation projects from non-point sources (Fang et al., 2005). There are a few nutrient offsetting projects in Australia that applied a similar approach, e.g., the riverbank stabilization and revegetation work at Laidley Creek that have been implemented by Urban Utilities, the riparian restoration at Logan River to offset the Beaudesert WWTP discharge, and the gully rehabilitation and riparian fencing and revegetation work to offset Kilmore WWTP discharge.

To address a catchment scale water quality issue, many case studies implemented nutrient offsets or trading via trusted third parties that act as a broker, credit bank or clearinghouse to facilitate the nutrient credit exchange between nutrient sources and assure that more participants get involved, mitigations perform adequately, and offsetting/trading activities go smoothly, etc. Credit exchange might also include a reserve of credits to deal with failed trades. However, these approaches don't necessarily develop a market-based trading system to encourage multiple trading activities in the project. For example, the Ontario South Nation River Trading Program in Canada is managed by the watershed authority, i.e., South Nation Conservation Association (SNCA) to offset point source phosphorus discharge from non-point source (primarily farmers) BMPs (Figure A2-1). This program was driven by a zero-phosphorus emission regulation from new or expanded WWTPs. It also resulted in cost-savings compared with an on-site WWTP. The SNCA established a Clean Water Fund that was run by a multi-stakeholder committee which contracted farmers to implement agricultural BMPs that generated the nutrient credits (O'Grady, 2008). Point source dischargers can purchase nutrient credits from the Clean Water Fund, and the SNCA oversees negotiating trade with them. Sales revenue is used to replenish the fund. This scheme implemented 269 phosphorus-reducing projects through the watershed's Clean Water Fund from 2000 to 2009 and reduced an estimated 11,000 kg of phosphorus loads.

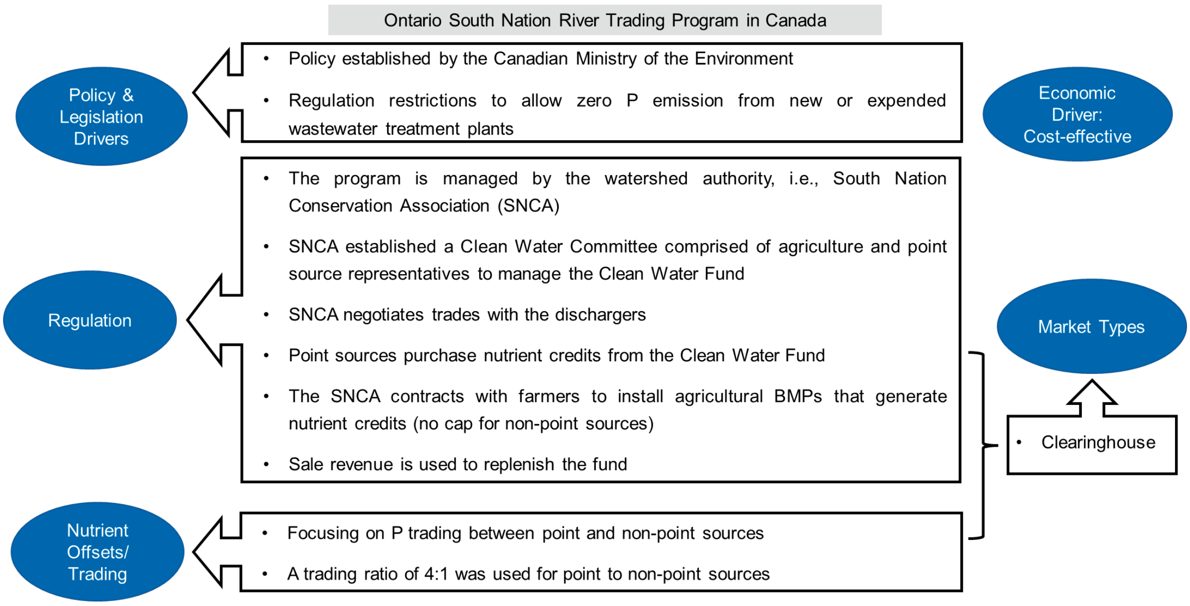


Figure A2-1. Case study: Ontario South Nation River trading program in Canada.

Another nutrient trading scheme developed for Chesapeake Bay catchments, USA, also used a third-party provider (Figure A2-2). This scheme allows point source dischargers to seek nutrient credits (both nitrogen and phosphorus) from both point and non-point sources, but with a sequencing logic of credits from point source dischargers first which commences in the same river basin. An association of point source dischargers facilitates the transfer of point source credits. For non-point source nutrient credits, the point source discharger pays a per unit nutrient discharge fee to the Virginia Water Quality Improvement Fund (WQIF) administered by state agencies, but only when no point source credits are available. Fee revenue from WQIF is used to sponsor point or nonpoint source reductions. There is also a requirement that the nutrient offset credits must be generated from the same year as the excess discharge was produced, thus, no credit banking is allowed.

A few studies have shown that the use of a trusted third party for active engagement and mediation with the agriculture industry was the main reason for the success of nutrient offset programs in Canada and the US (Breetz et al., 2005; Shortle, 2012).

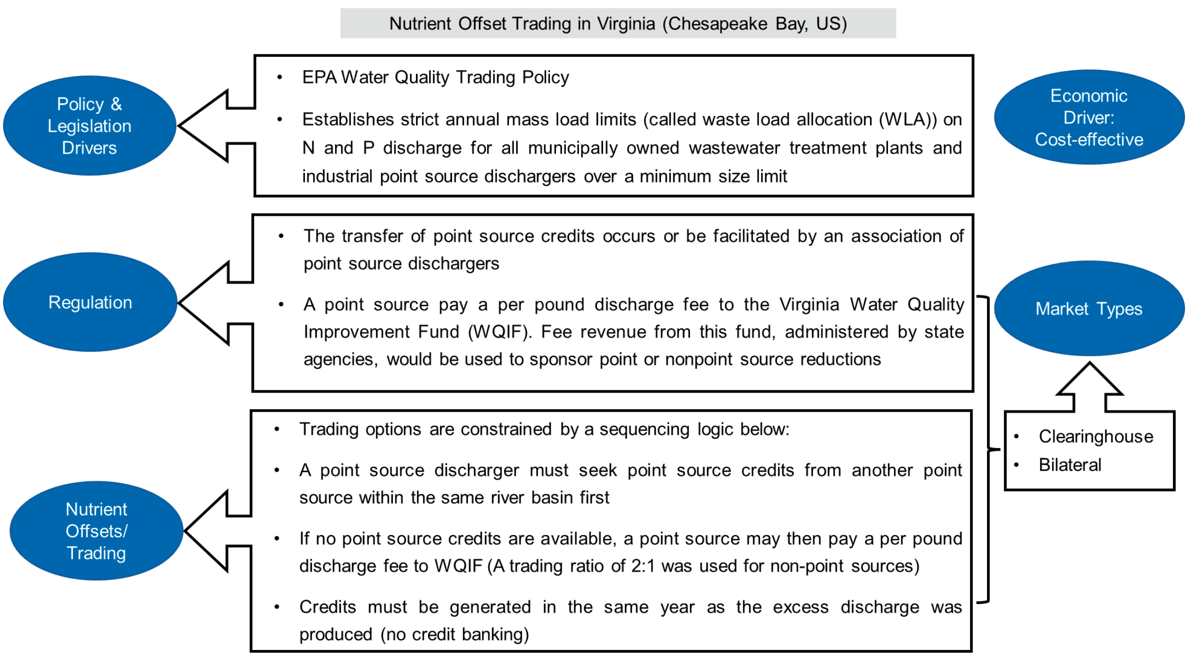


Figure A2-2. Case study: Nutrient offset trading program in Virginia (Chesapeake Bay, USA)

A more sophisticated nutrient offsetting approach involves a market-based nutrient credit trading system, so credit buyers and sellers can negotiate amongst themselves, rather than mostly relying on a third party to facilitate the trading. However, this approach requires a well-developed nutrient credit calculation framework for non-point sources. The most famous and active market-based nutrient credit trading program was developed in Pennsylvania, Chesapeake Bay, USA (Figure A2-3) (Boleman and Jacobson, 2021). This program is managed by the Pennsylvania Infrastructure Investment Authority and the Pennsylvania Department of Environmental Protection (DEP). Both nitrogen and phosphorus credits are traded between agricultural entities, WWTPs, and municipalities within the same compliance year (October 1st to September 30th of each year). Pennsylvania DEP provides Excel spreadsheets for estimating nitrogen and phosphorus credit generation for nutrient credit calculations using best management practices and nutrient reduction activities occurring for non-point sources. Agricultural producers from the catchment need to meet specific baseline requirements before extra nutrient credits can be generated, including meeting existing nutrient management and soil erosion control laws, manure and fertilizer application limits, and maintaining a 35 ft. (10.7 m) minimum vegetation buffer between fields in production and streams (Boleman and Jacobson, 2021). Additionally, all calculation methodologies to determine credit generation must include 10% set aside for PDEP's credit reserve to address pollutant reduction failures and uncertainty.

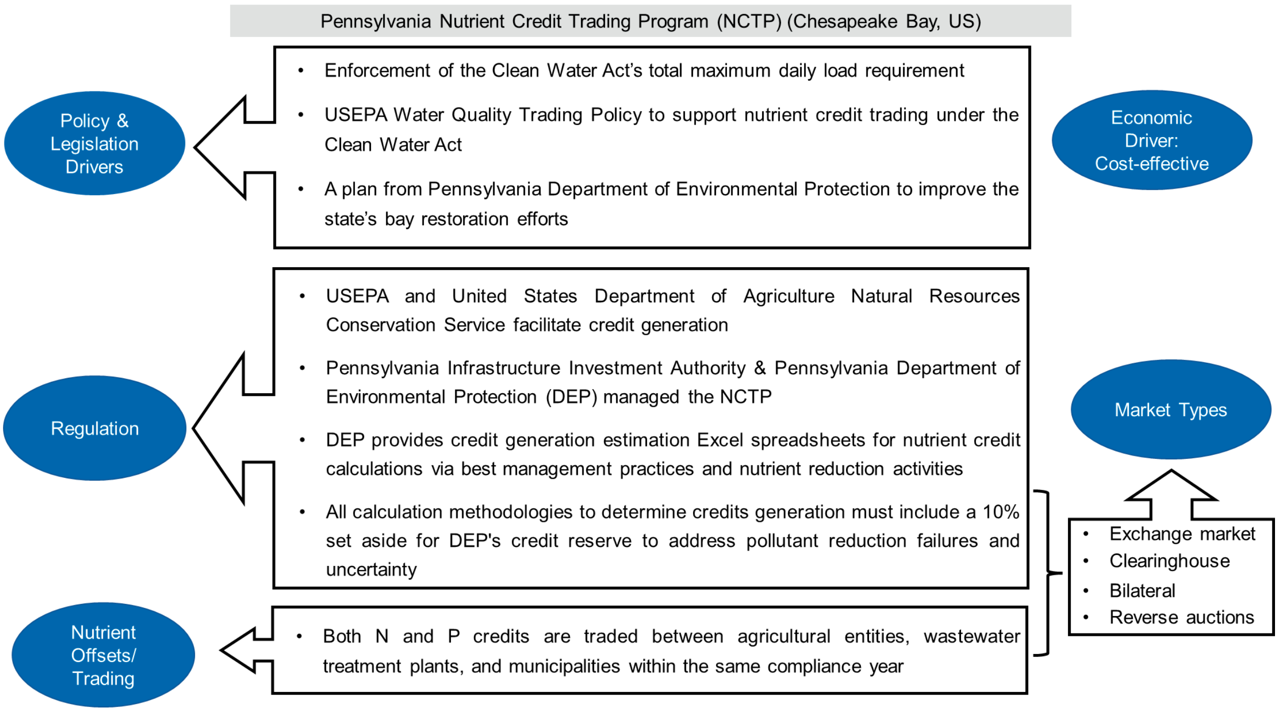


Figure A2-3. Case study: Pennsylvania nutrient credit trading program in Chesapeake Bay, USA

A2.1.3 Nutrient offsets between non-point sources

Nutrient offsetting can also include offsetting between non-point sources. Some case studies explored the efficacy of this approach for catchments where non-point sources are the dominant nutrient source dischargers. In these instances, one or both of the non-point sources involved in the offset needs to be regulated (Selman et al., 2009). For example, Lake Taupo catchment in New Zealand established the first agricultural non-point source water-quality cap and trade scheme in the world (Figure A2-4) (Duhon et al., 2015). This scheme caps discharges from farmers and foresters and allows trading amongst these participants to achieve nutrient load reduction to Lake Taupo. The Lake Taupo Protection Trust was set up, a public trust funded by local, regional, and national communities, to reduce the nitrogen load by 20% via the purchase and conversion of land or the purchase of farmers' nitrogen allowances. Farmers can also sell allowances to the Trust while remaining within the overall catchment cap. The nitrogen loads from farms were calculated using a nutrient budgeting tool known as OVERSEER. Even though this scheme was only implemented a couple of years ago at the time of publication of the paper (Duhon et al., 2015), there was evidence of improved cost-effectiveness compared with traditional command-and-control approaches in achieving environmental goals of a 20% nutrient load reduction to Lake Taupo (Duhon et al., 2015).

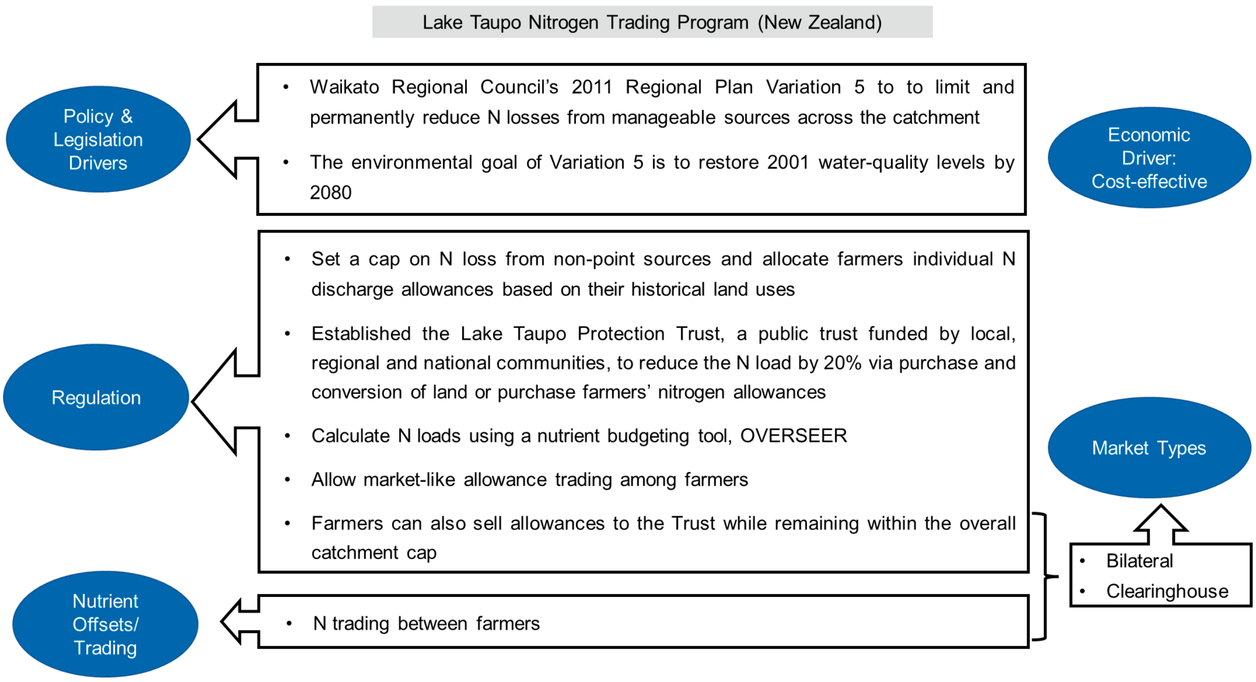


Figure A2-4. Case study: Lake Taupo nitrogen trading program in New Zealand.

Hasan et al. (2022) used a modelling approach to simulate a potential cap-and-trade scheme between non-point sources for nitrogen loads in Limfjorden catchment, Denmark. Their study showed that using a trading scheme to reduce the nitrogen load by 21% below the baseline at Limfjorden could reduce costs by 56% compared to the traditional command-and-control regulation (Hasan et al., 2022). There have also been studies exploring the use of mussel farming to filter and trap nutrients from agricultural runoff. Their results showed that mussel farming can be considerably cost-effective in nutrient reduction compared to the traditional command-and-control regulation (Ferreira and Bricker, 2016; Gren et al., 2009). However, mussels can also suffer mortality if catchment inputs have a high sediment load, so background studies are needed on individual catchments to determine their effectiveness.

A2.1.4 Nutrient neutralizing for new activities in the catchment

Instead of using the term 'nutrient offsets', the term, 'nutrient neutralizing' has been proposed in Europe. This involves the use of legislation or legislative proposals to improve and protect water quality in the United Kingdom (UK) and some countries in Europe, such as Sweden, Denmark, and Finland. Under the concept of 'nutrient neutralizing', any new activity/project at the catchment needs to earn permission to conduct the activity/project whilst demonstrating no increase in the release of nutrients into the environment, i.e., neutralizing its net impacts on water bodies by implementing some nutrient abatement measures. For example, in order to neutralize the nutrient runoff during earth works from one of the largest home construction companies in the UK (Taylor Wimpey), the effectiveness of a constructed wetland outline design was assessed by a consultant. The firm responsible, i.e., RPS Group plc, also assessed and demonstrated that a large number of nutrient credits could be generated by adapting the design of the wetlands and reedbeds to remove additional phosphate (and nitrate) load from the river. These two functions can be designed to be compatible (RPS group website on the 15th August 2022: https://www.rpsgroup.com/services/environment/ecology/expertise/achieving-nutrient-neutrality/).

A2.2 Challenges for implementing nutrient offsetting

Even though nutrient offsetting can provide multiple potential benefits as mentioned in previous sections, offsetting may not be applicable to all situations in every catchment. The feasibility of applying nutrient offsetting to achieve nutrient load management goals might need to be assessed for specific catchments considering the uncertainties related to non-point source nutrient loads (Section 2.2.1), nutrient characteristics and loads differences between nutrient sources and their impacts on ecosystem responses (Sections 2.2.2, 2.2.3, and 2.2.4), catchment rehabilitation impacts (Sections 2.2.5 and 2.2.6), and the impacts of nutrient offset projects with different spatial and time scales to waterways (Section 2.2.7), e.g., local- vs catchment- scale impacts, and dry and wet weather impacts.

It is clear from the literature review that there are a range of uncertainties that is a major barrier limiting the implementation of nutrient offsetting, and its various variations globally (Horan and Shortle, 2017). These uncertainties may include difficulty in accurately measuring the nutrient load, lack of a mechanism to determine offset ratios between different nutrient sources, insufficient data on the evaluation of the effectiveness and cost-effectiveness of mitigation strategies, and lack of information on optimal sites for maximum nutrient reduction in the catchment, etc. In this section, we will summarise what are those uncertainties and provide some strategies to address these uncertainties.

A2.2.1. Uncertainty of non-point source nutrient contribution and baseline establishment

Non-point source nutrient pollution within catchments, especially from agricultural land use, has been found to be a key contributor to eutrophication worldwide in many catchments (Diaz and Rosenberg, 2008; EPA, 2015), but, unlike point sources, non-point sources are typically not regulated. One reason why limits do not exist for non-point sources is the lack of quantitative information on the loads and proportion of nutrients contributed by non-point sources within many catchments. There are few long-term monitoring studies with sufficient temporal resolution to capture high river flow events when most of the nutrient loads travel downstream from non-point sources. As a result, in the case of catchments that are dominated by nutrient discharges from non-point sources, capping or limiting the point source discharge is unlikely to achieve sufficient nutrient load reductions to result in demonstratable improvements in water quality parameters linked to nutrient inputs. Similarly, if non-point sources only contribute a small proportion to the nutrient loads within a catchment, restoration of non-point source pollution may be inadequate to achieve a significant water quality improvement (Gaylard, 2005).

The quantification of nutrient loads from non-point sources is also critical to establish a baseline for nutrient credit calculation for non-point sources via nutrient offsetting (Selman et al., 2009). The identification of acceptable baselines from non-point sources has been an unresolved issue for many catchments (Stephenson et al., 2010). This is because nutrient loads from non-point sources are often not monitored as it can be logistically challenging during high river flow events, and costly (King and Kuch, 2003; Srinivas et al., 2020). Non-point source nutrient loads are generally estimated via modelling approaches, based on mass balances with empirical equations for nutrient processes (Fu et al., 2019), or conversely based on land use types and management practices that might change nutrient discharge (Hermoso et al., 2015; Pantus et al., 2011).

Due to the lack of sufficient monitoring data to calibrate or validate models, there is a high level of uncertainty regarding both the baseline output of nutrient discharges from non-point sources, but also the efficiency of BMPs for nutrient discharge reductions. Applying general models to specific catchments can also fail to represent local conditions (Wainger and King, 2007). Therefore, better monitoring design and robust catchment water quality models for specific catchments are needed to provide more certainty in the benefits of offsetting.
In one example, non-point source discharges have been converted to measurable point source discharges to reduce the uncertainty of pollution load estimation. In the California Grassland Areas Program, technology for drainage water management in irrigation districts was used to convert nonpoint emissions from farms into point source emissions that can be more readily monitored and measured (Shortle, 2012). This site had issues with excessive selenium impacting water quality and therefore a trading program was developed that allowed the participating districts to meet district-specific selenium limits through trading.

A2.2.2 Temporal and spatial variation of nutrient sources

Non-point source discharges from catchments are naturally stochastic as their discharge is largely driven by climate conditions and dry/wet cycles (Pekárová et al., 2003; Romero et al., 2013), thus may show more seasonal and interannual variation than point sources. This makes it challenging to directly compare point- and non-point discharges. For example, Donohue et. al. (2005) found that the extent of highly productive grasslands and urban areas in an Irish catchment was positively correlated with nutrient loads, higher nutrient concentrations and algal bloom risk during high flow in summer and autumn, not in spring and winter. There are examples of nutrient offset projects failing (lack of trading) due to negligible non-point source nutrient load generation during dry weather conditions, with nutrient generation only being substantial in the wet season, hence limiting the ability to offset point sources (Morgan and Wolverton, 2005). This highlights the need to reconcile wet vs. dry season inputs from point- and nonpoint sources in any offsetting scheme.

In drought and flood-prone catchments, a single storm event may dominate nutrient loads for a couple of years (Nash and Murdoch, 1997; Fleming and Cox, 2001; Drewry et al., 2006), especially for catchments with intermittent rivers. Drying and rewetting cycles can significantly increase the liberation of carbon, nitrogen, and phosphorus from soils into water bodies (Baldwin and Mitchell, 2000). Other factors, such as land use types, soil types, agricultural BMPs, wetlands, reservoirs, hydrologic connectivity, and hydrologic residence time, can also significantly change the nutrient loads and composition of non-point source discharges (Gelbrecht et al., 2005; Newcomer-Johnson et al., 2016; Hansen et al., 2018; Shi et al., 2019 Lampman et al., 1999; Laznik et al., 1999; Sileika et al., 2006).

Point source nutrient discharges, such as from WWTPs, show less temporal and spatial variation in nutrient composition and loads than point sources (Wainger and King, 2007). The relative contribution of point source nutrient load to the total nutrient load in receiving waters can also be significantly affected by river flow conditions. Depending on the location, there is also scope for point source discharge to be stored and released to maintain river flow conditions during dry seasons (Bhargava, 1985) or to better mimic natural flow regimes of streams in arid and semi-arid regions (Brown et al., 2011; Eppehimer et al., 2021).

A2.2.3. Determining appropriate environmental equivalence ratios between nutrient sources

Point source and non-point source nutrients enter waterways at different concentrations and proportions of nutrient forms, which will be assimilated into the environment in different ways. The frequency and location of discharge in the catchment will also vary. The impact of point and non-point nutrient sources on water quality measures in receiving waters is, therefore, unlikely to be directly comparable. Providing the scientific basis for determining offset ratios between the two is a fundamental step to implementing a fair and transparent nutrient offset program (Puzyreva et al., 2019). Some studies have used relatively high offset ratios as a precautionary response, but they may also reduce the incentive for point-source polluters to engage in offsetting because the scale of mitigation required is greater and hence more expensive. It might be more cost-effective for point-source polluters to treat waste on-site rather than trade with other polluters (King and Kuch, 2003). High offset ratios have been shown to reduce the willingness of buyers to engage in trade (Horan and Shortle, 2017).

Commonly used nutrient offset ratios between point source to non-point sources have been shown to range, for example, from 1.5:1 to 4:1 in the USA, with 2:1 commonly used to manage uncertainties associated with cross-pollutant source offsetting (Hoag et al., 2017; Morgan and Wolverton, 2005). However, according to our review, there is limited scientific evidence to validate the ratios currently used, especially in terms of the aquatic ecosystem response to different nutrient sources (environmental equivalency ratios).

In most cases, only TN and/or TP loads from individual sources are measured for calculating nutrient ratios. However, it is well known that individual sources have fractions of different forms of nitrogen and phosphorus that have differing effects on aquatic ecosystem health (Gaylard, 2005). For example, discharge from WWTPs with tertiary treatment typically has a higher proportion of dissolved inorganic nitrogen while catchment runoff has a higher proportion of particulate nitrogen. Studies of the responses of marine and freshwater microalgae, using algal bioassays, to sediments generated from catchment runoff identified that specific forms of nitrogen, as well as the organic carbon to nitrogen ratios, best explained the algal responses (Franklin et al., 2018; Garzon-Garcia et al., 2018a). This suggests that nutrient availability is not just dictated by dissolved inorganic nutrient concentrations, but bacterial mediation, where organic forms of nutrients become bioavailable to algae, may also be important. Therefore, TN and TP loads are unlikely to be the most meaningful measures for the comparison of point and non-point source discharge.

Recent studies in small agricultural streams in Germany also indicated that the changes in bioavailable carbon to reactive nutrient ratios may control nutrient assimilation by heterotrophic bacteria (Graeber et al., 2021). Another study showed that the addition of labile carbon to a eutrophic lake in Denmark changed the algae biomass and community structure in a 20-day mesocosm experiment (Fonseca et al., 2022).

A2.2.4 Variation of ecosystem responses to nutrient loads

The variability of aquatic ecosystem responses can be affected by the assimilative capacity of the aquatic ecosystem to different nutrient concentrations and loads, the contact time of nutrients within each system, and the interaction of nutrients with other pollutants (Farhadian et al., 2015). The assimilative capacity of an aquatic ecosystem can be interpreted as the ability of aquatic ecosystems to process, dilute, and disperse pollutants without unacceptable harm to the aquatic environment, such as the loss of biodiversity or the degradation of environmental services (Butler et al., 2013; GESAMP, 1986). The assimilative capacity can be assessed by monitoring, experimentation, and modelling approaches(Butler et al., 2013; Peternel-Staggs et al., 2008). River and coastal water assimilative capacity is often empirically defined in different regions, e.g., total maximum daily load (Landis, 2008) or the annual pollution cap (maximum amount of a pollutant) applied in different catchments. Both these measures only infer assimilative capacity, rather than directly measuring it. The assimilative capacity has also been estimated by a range of threshold indicator values such as chlorophyll-a concentrations, dissolved oxygen concentrations, macrophyte/seagrass/mangrove extent and species presence, and nutrient processing rates, such as nitrification and denitrification rates (Butler et al., 2013). These thresholds can be estimated by testing scenarios via ecohydrological models (Butler et al., 2013; Wild-Allen et al., 2011, 2010; Xu et al., 2016).

Many studies have examined nutrient thresholds, where there is an abrupt system change in water quality, or some other phenomenon (Groffman et al., 2006; Stevenson et al., 2012; Xu et al., 2015). However, there has been an emphasis on lakes rather than the river and coastal systems. For example, a TN threshold of 0.8 mg L-1 and TP threshold of 0.03–0.05 mg L-1 were identified for eutrophic Lake Taihu for cyanobacterial blooms (Xu et al., 2015) and from the Ozark highlands ecoregion for nuisance filamentous green algae bloom in streams (Stevenson et al., 2012). A higher threshold of 0.59–1.79 mg L-1 for TN and 0.03–0.28 mg L-1 for TP was identified for agriculture steams in the western US, based on a range of physical, biological, and chemical factors (Black et al., 2011). According to the concept of alternative stable states in ecosystems (Scheffer, 1990; Scheffer and Carpenter, 2003), aquatic ecosystems under certain nutrient thresholds have better ecosystem health, e.g., low trophic status, low disturbance, and high biodiversity, than that above the nutrient thresholds. They may also have a higher tolerance/resistance and higher nutrient removal capacity to excess nutrient input, as well as higher resilience in maintaining the current status than aquatic ecosystems that have nutrient levels above the nutrient thresholds (Folke et al., 2004). However, once the system has shifted to a new stable state with reduced ecosystem health, the recovery to the previous state does not readily occur requiring a significant change in the stress level (Bellwood et al., 2004; Scheffer and Carpenter, 2003).

Seasonal differences may affect the assimilative capacity. For example, outputs from water quality modelling for Gallinas River, Mexico, indicated that dry and wet seasons resulted in different nutrient load reduction rates for the same proportion of nutrient input increase in the river from agriculture and industrial WWTP discharges (Villota-López et al., 2021). In the model output, nitrate concentrations were three times higher in the dry season than the wet season because the river could not assimilate less nitrate in the dry season compared to the wet season. The findings were also similar in other river systems (Chiejine et al., 2014; Obin et al., 2021; Torres-Bejarano et al., 2023) which showed that the dry season has lower river assimilative capacity, and it is proposed that this is due to less mixing and dilution processes compared to the wet season. This is despite the fact that the total nutrient loads from catchments into rivers and coastal waters in wet seasons can be much higher than in dry seasons (Cardoso-Mohedano et al., 2022; McPherson et al., 2005, 2002). Although the impacts of nutrient inputs in the dry season may be greater closer to the source of nutrients, the impacts of wet season inputs are likely to be greater further from the site of input, e.g., seagrass and coral reef loss (Fabricius et al., 2016; McCulloch et al., 2003). Therefore, reconciling the location and scale of impacts of dry and wet season inputs on the environmental equivalency ratio between nutrient sources is critical for effective nutrient offsetting programs.

The proportion of different nutrient forms and the interaction of nutrients with other pollutants can also significantly change ecosystem responses to nutrient inputs from catchments (Wagenhoff et al., 2011). Primary production in receiving waters can be limited by either nitrogen or phosphorus individually or co-limited by both (Elser et al., 2007), so the impacts of nitrogen on primary production may depend on phosphorus levels in receiving waters that were co-limited by both nutrients (Guildford et al., 2022). The presence of fine sediments from runoff events may counteract nutrient inputs and limit light availability to primary producers (Wagenhoff et al., 2011)(Hunt et al., 2012). Additionally, the presence of fine sediment combined with a small increase in nutrient input, as a second stressor, may lead to significant river condition deterioration (Wagenhoff et al., 2011).

Water residence time (pollutant contact time) can significantly affect nutrient transformation and attenuation in receiving waters. Longer contact time of nutrients in the river system may show either a positive or negative effect due to: 1) an increase in the capacity for removal of nutrients in the river because of the longer contact time; and 2) a reduction in the dilution rate of the nutrients from less mixing. Modelling of nutrient processing in rivers has shown that the river assimilative capacity can be improved by up to 80% or even 97% by slowing down the river flow rate, i.e., increasing the residence time (Farhadian et al., 2015; Hashemi Monfared et al., 2017). However, this may come at the expense of ecosystem health in the river. Therefore, for nutrient offsetting schemes it is important to be clear about whether the benefits should be near the point of discharge or in vulnerable habitats further removed from the discharge point.

A2.2.5. Uncertainty of catchment mitigation effectiveness and cost-efficiency

Uncertainties related to calculating the credits from nutrient reduction have been listed as one of the biggest challenges to implementing nutrient offsetting (Morgan and Wolverton, 2005). There are significant uncertainties and variability in the effectiveness of mitigation strategies for nutrient removal for non-point sources. For example, catchment revegetation, streambank rehabilitation and gully remediation, which may include engineering-based mitigations (e.g., rock chute or check dam), can be more effective for reducing particulate nutrients than dissolved nutrient reduction (Garzon-Garcia et al., 2019). In contrast, wetland construction and some agricultural BMPs (e.g., fertilizer and irrigation management, stock exclusion from the waterways), may be more effective for reducing dissolved nutrient delivery into waterways. In this section, we review the nutrient and sediment reduction efficiencies from a range of commonly used catchment mitigations and explore multiple confounding factors that reduce their effectiveness.

Revegetation
Hillslope and riparian buffer revegetation have frequently been used to reduce sediment and nutrient delivery from catchment non-point sources. The scale of vegetation cover in riparian areas, and the travel distance of on-site pollutants to the closest river channel are regarded as the most critical factors to consider when estimating the sediment and nutrient delivery ratio from catchments into waterways (Kinsey-Henderson et al., 2005). Catchment sediment delivery ratios can vary dramatically from close to zero on floodplains (due to sediment deposition) up to 0.7 in the uplands in the Murray Darling Basin, Australia (Lu et al., 2006), meaning that 70% of the sediment in the uplands is transported to the water.
Before sediment and nutrients reach the river channel, riparian vegetation can act as an efficient buffer to trap sediment and particulate nutrients to prevent them from getting into waterways (Mayer et al., 2007; Sweeney and Newbold, 2014). Studies have shown that it is important to have continuous riparian buffers on both sides of the river to maximize the benefits of protecting or improving water quality (Karssies and Prosser, 1999; Prosser and Karssies, 2001). However, the riparian buffer is typically less effective at removing dissolved nutrients as sediment-attached nutrients, especially in catchments, and at times when there are large rainfall events (Dosskey et al., 2006; Karssies and Prosser, 1999). Studies have shown that a vegetated buffer width of 30 m is capable of removing approximately 80% of sediment attached nutrients and 68% of nitrate in the runoff, with the removal efficiency varying from 40–99% for sediment and 20–95% for nitrate (Mayer et al., 2007; Sweeney and Newbold, 2014). However, most studies have been conducted in climatic regions that do not have large rainfall/flooding events. The effectiveness of riparian buffers in trapping nutrients is likely to be substantially diminished during these events. As such, riparian buffers may be less cost-effective than other means of remediation in these climatic regions.

Sediment and nutrient reduction efficiencies from the riparian buffer also depend on a range of catchment and sub-catchment-specific factors. This includes soil types, vegetation types, vegetation root volume ratios in the riverbank, vegetation canopy management, river discharge volumes (stream order) and intensity, slopes of the riparian buffer areas, land use, and microclimates of the drainage area above the riparian buffer areas (Laceby et al., 2017; Liu et al., 2008; Sweeney and Newbold, 2014). Site maintenance after revegetation, such as weed control, watering, and replanting, can also be critical to improving the revegetation success, and thus their effectiveness in nutrient and sediment removal. Further research on these factors is needed to improve the certainty of estimating riparian buffer sediment and nutrient removal efficiencies for specific catchments.

Riparian fencing for stock exclusion
Fencing off the riparian areas for stock exclusion can reduce the direct impact of these animals grazing and trampling riparian vegetation, disturbance to the riverbank, and deposition of animal faeces and urine into the water. A review by Grudzinski et al. (2020) identified a positive impact of riparian fencing on river water quality in cattle-grazed land with a buffer width >5–10 m. The greatest improvements were for fecal indicator bacteria and sediment parameters, followed by nutrients. However, the effectiveness of riparian fencing has been shown to vary dramatically (e.g., from 0–96% of water quality improvement for different indicators, such as sediments, nutrients, and fecal indicator bacteria) across studies and at different flow conditions (Kay et al., 2018; Sunohara et al., 2012). There is also a lack of studies in semi-arid and tropical environments that have different runoff and flow characteristics and greater erosion risks (Grudzinski et al., 2020). Therefore, a high level of uncertainty remains on the effectiveness of riparian fencing in improving river water quality (Muirhead, 2019).

Gully remediation

Gullies in the landscape can be sites of considerable erosion during rainfall events resulting in sediment and nutrients associated with sediment being transported into waterways. Gully remediation may include stabilizing gully heads, walls, and floors via fencing off livestock, revegetation, managing runoff, and/or engineering-based earthwork, etc. Bartley et al. (2020) reviewed the magnitude and the timeframe over which sediment load reductions occurred following the remediation of gullied landscapes globally. They found only a limited number of studies (37 studies found worldwide) that quantitatively documented the effectiveness of gully remediation efforts. Percentage sediment erosion reduction ranged from 12–94%, and the timeframe to achieve this reduction varied considerably, i.e., from 2–80 years (Bartley et al., 2020). This is likely due to the variation in gully types, remediation methods applied, and differences in hydrological conditions. They also showed that the effectiveness of gully remediation for nutrient load reduction is not necessarily aligned with sediment load reduction, especially for dissolved nutrients. Review studies also suggested that longer timescale (e.g., semi-decadal to decadal) monitoring is needed to more accurately determine the effectiveness of gully remediation on sediment and nutrient reductions (Bartley et al., 2019; Doriean et al., 2021).

It has been proposed that low-cost remediation strategies, such as runoff diversion, pasture renovation, and livestock exclusion are typically less efficient at rapidly reducing gully erosion compared to engineering-based remediation, however, in the long term, they are easier to manage and maintain (Koci et al., 2021). Vegetation restoration in the drainage areas above the gully head cut, which reduces hillslope runoff, can also be effective at controlling gully expansion (Chen and Cai, 2006; Gomez et al., 2003). However, a high percentage (>60%) of combined vegetation cover of trees and grasses in the drainage area, needs to be restored and maintained to ensure that this vegetation remains an effective control mechanism (Li et al., 2015).

Wetlands

Wetlands have been generally regarded as effective to remove nutrients from wastewaters and provide additional services like stormwater retention to reduce the flooding and erosion risk, carbon storage, and supporting biodiversity and recreation activities. A global review of nutrient retention in restored streams and rivers which had some wetland reconnection/construction showed that nitrate needs to travel an average of 10 times further to be assimilated in degraded river systems than that in restored systems (Newcomer-Johnson et al., 2016). Many studies have found that the nutrient removal efficiency of wetlands can vary widely (ranging from negative values to close to 100%) (Land et al., 2016). This is affected by a range of factors, including water temperature, the connectivity of wetlands with waterways and water residence time (hydraulic conditions), inflow water quality (e.g., turbidity and nutrient concentrations), wetland internal conditions (e.g., soil carbon content, macrophyte coverage, and redox potentials), and post-construction management for wetland vegetation and sediment (Adame et al., 2019; Hansen et al., 2018; Jesus et al., 2018; Land et al., 2016). Wetlands are particularly effective at removing nitrate via denitrification. Denitrification is the dominant permanent nitrogen removal process from wetlands, which has often been found to be enhanced by higher nitrate concentration, higher organic carbon supply, and appropriate redox conditions (Adame et al., 2019; Sirivedhin and Gray, 2006). In contrast, ammonium, phosphate, and dissolved organic nitrogen can be released from wetland sediments, depending on mineralization and nutrient desorption conditions (Kavehei et al., 2021; Roberts et al., 2021). Wetlands might also work as settling pools for sediments, but the phosphorus loading reduction from wetlands might be diminished over time with the accumulation of sediments and the development of hypoxia that may result in internal phosphorus release (Oldenborg and Steinman, 2019). Wetlands may even become a nutrient source during high flow events due to the flushing of accumulated internal nutrient loads within the sediment (Fisher et al., 2004). Therefore, wetland maintenance and management, including sediment dredging and maintaining healthy vegetation, is critical to ensuring optimal wetland nutrient removal efficiencies. Additionally, there is also a lack of evidence for long-term (e.g., >20 years) wetland performance in the literature (Land et al., 2016). As such, understanding nutrient removal processes and the development of complete nutrient budgets in a wider range of wetlands is needed.

Additionally, the cost of construction and maintenance has been a key concern to implement wetlands as a mitigation. There are a few studies that investigated the cost-effectiveness of applying wetlands to manage catchment nutrients via hypothetical scenarios. For example, a catchment-scale analysis of re-establishing wetland buffer zones along rivers showed their high cost-effectiveness as a mitigation to reduce non-point nutrient pollution from agriculture in North-Eastern Poland (Jabłońska et al., 2020). Their findings showed that hypothetically catchment-scale polygonal wetland buffer zones can remove 11%–30% nitrogen and 14%–42% phosphorus load from the catchment, and 33%–82% nitrogen and 41%–87% phosphorus removed by linear wetland buffer zones with the cost comparable to the costs of building 20 km of a major highway (Jabłońska et al., 2020). A cost-effectiveness comparison analysis between semi-natural wetlands and activated sludge WWTP systems in Italy showed that they are equally effective in their capacity to remove nutrients from the wastewater, but overall semi-natural wetlands would improve the service cost related to nutrient removal by 2-8 fold, compared to the activated sludge WWTP systems (Mannino et al., 2008). However, this study also found that semi-natural wetlands can have higher development costs, such as designing, planning, and implementation costs. In contrast, they typically have significantly lower maintenance costs due to their relative self-regulating systems, low artificial energy inputs and no waste disposal needs, compared to the activated sludge WWTP systems (Mannino et al., 2008).

Agricultural and urban BMPs

Best management practices (BMPs) are another strategy for the reduction of pollution loads from agricultural and urban areas. These are typically implemented by farmers in agricultural areas, or local government authorities, in the case of urban areas. Agricultural BMPs may include: recycling treated water for irrigation; controlling and reducing irrigation and fertilizer use (nutrient management) activities; reducing soil erosion by ensuring constant crop cover and crop rotation; using low fertilizer-use crops such as nitrogen fixers; no-till farming; and constructing vegetated buffer strips and wetlands. The effectiveness of agricultural BMPs is typically assessed using models as it is expensive and challenging to use monitoring programs to test effectiveness. The ability of BMPs to reduce nutrient loss from the land depends on many factors including the type of BPM being used, the timing of the BMP application, soil infiltration rates for nutrients, and rainfall patterns (Lichtenberg, 2004; Qiu, 2013; Rittenburg et al., 2015). There is a need to develop catchment-specific water quality models with representative local conditions to provide better accuracy on the effectiveness of agriculture BMPs.

Urban BMPs can be used to treat stormwater runoff quality and quantity by applying a range of practices. For example, bioretention systems can be used to reduce surface runoff, increase groundwater recharge, and treat pollutants. Other examples include green roofs (vegetated roof systems) to retain stormwater runoff; and the construction of wet and dry retention ponds to settle sediments and retain stormwater (Dietz, 2007). Water-sensitive urban design is a strategy that combines a range of approaches and can be integrated into urban planning by designing more water-efficient urban environments with less impervious surfaces (Sharma et al., 2016; Wong, 2006). It has the benefit of reducing water and nutrient losses from the land. Despite the use of a range of these systems, there are few long-term studies that quantify the effectiveness of urban BMPs in reducing nutrient losses over time (Liu et al., 2017). As with all mitigation strategies, the effectiveness varies depending on factors such as the design, local land use, and climate conditions (Ahiablame et al., 2012; Liu et al., 2017). Reducing the uncertainties to better predict the effectiveness of both agricultural and urban BMPs would also improve the load estimation from non-point sources in catchment water quality models, and increase confidence in the use of these methods (Arabi et al., 2007).

A2.2.6. Lack of environmental assessment of offset applications

There is a lack of sufficient monitoring data on the effectiveness of nutrient offsetting (Angelopoulos et al., 2017; Brooks and Lake, 2007), with many studies relying on models that may not be effectively calibrated or validated. The associated monitoring work is generally costly, weather-dependent, and lacks historical data as a benchmark to compare the post-mitigation results (Roni et al., 2008). Existing water quality monitoring programs in catchments often lack the appropriate temporal and spatial scales for robust assessment of impact and effectiveness. Additionally, this monitoring may not include dissolved and particulate components. As a result, there may be a lack of confidence to invest in offsetting. Mitigation and monitoring projects, therefore, need to be designed to be representative and response sensitive to provide robust data on their effectiveness (Angelopoulos et al., 2017). Selecting appropriate reference sites for monitoring projects or starting to monitor the mitigation site before the mitigation is implemented can be critical to assessing the effectiveness of mitigations. The use of the space-for-time concept (selecting appropriate reference sites for data collection to represent the pre-mitigation data) for monitoring can be useful when sampling cannot commence before mitigations are made (Laceby et al., 2017; Pickett, 1989).

In some conditions, due to the small contribution of an offset project to water quality improvement in the degraded river systems, it may not be possible to measure the water quality change directly (O'Mara et al., 2014). As a result, catchment water quality modelling provides an alternative to assess the effectiveness of mitigations on water quality improvement but relies on a robust catchment water quality model.

A2.2.7 Identification of critical source areas for mitigation within catchments

To maximise the benefits of catchment restoration works, it is important to identify sites of maximum benefits from restoration in the landscape. Catchment non-point nutrient loads can be generated from a range of processes, including riverbank, gully, and hillslope erosion, as well as agricultural activities such as fertilizing and tilling. For example, nitrogen loads delivered to the Gulf of Mexico, were greatest from corn and soybean cultivation, whilst phosphorus loads were greatest from animal manure (Alexander et al., 2008). In Australia, cultivated agriculture is typically less intensive, so gully and riverbank erosion are a more important contributor to nutrient loss from catchments (Olley et al., 1996; Wasson et al., 1998). Differences in soil type may also significantly affect the sediment and nutrient generation for the same nutrient source (Garzon-Garcia et al., 2018a). Therefore, prioritizing critical source areas for implementing management actions is needed to achieve catchment scale nutrient load reduction goals.

Remediation of hotspots where more than one pollutant can be tackled, e.g., sediment and nutrient, should also take higher priority than single-pollutant type hotspots, due to the interaction between pollutants. For example, in New Zealand streams, dissolved inorganic nutrient impacts have been shown to interact with fine sediment to result in larger changes in macroinvertebrate communities compared with a single-pollutant type (Wagenhoff et al., 2011). Focussing on both pollutants at the same time may also achieve the best outcomes for water quality improvement in aquatic ecosystems (Townsend et al., 2008; Wagenhoff et al., 2011). However, most offset programs only target one objective, such as nitrogen, phosphorus, or sediment.

A2.2.8 The importance of scale for nutrient offset projects

As outlined above, determining nutrient equivalency between nutrient sources is an important concept, but clarity around where the ecosystem health benefits from offsetting should be is also important. Mitigation activities to remediate areas of a catchment often do not occur at the site of point-source nutrient inputs. Some authors suggest that the same nutrient load reduction from mitigation sites that are closer to the problem zone may be more valuable to offset the point source nutrient discharge than when sites are further away from the zone (Hall, 2012). To offset the point source discharge, most nutrient offsetting projects require that non-point source nutrient abatements must be implemented from the upper stream of the point source discharge site. Additionally, the offset ratio between point and non-point sources is typically larger than 1:1 due to the non-point nutrient attenuation during transport from catchments to waterways. However, this location selection should consider whether this is the priority zone for remediation. In one example, the Dixie Drain Phosphorus Removal Facility in the City of Boise, Idaho, USA, was located downstream of municipal WWTPs is providing more environmental benefits to the entire catchment, including sediment reduction, improved fish and aquatic life habitat and river aesthetics, compared with an upstream site (EPA, 2022). Therefore, it is important to have clarity on what ecosystem health benefits should be achieved from the nutrient offsetting projects.

Studies vary in whether the benefits are upstream, near the site of point-source discharge or at the end of the catchment, i.e., river mouth, or in adjacent coastal waters. Depending on the distance between the location of source nutrient inputs and where ecosystem health improvements are needed, an understanding of the transformation and attenuation of nutrients as they are transported downstream is needed. Additionally, the ecosystem response of different waterways to nutrients also needs to be understood.

Nutrient offsetting programs can be set up for one or multiple catchments. Having a larger spatial scale for the offsetting market could increase the economic benefits due to more potential participants. The trade-off may be a decreased localized environmental benefit due to the uneven scales of trading across catchments (Hasan et al., 2022).

A2.3 Lessons learnt from international projects on nutrient offsetting

In addition to the bio-physical knowledge gaps outlined above (Sections 2.1 and 2.2), another challenge for nutrient offsetting programs can be a lack of participants (buyers and sellers). This is due to a range of reasons including: uncertainty about trading rules; trust and communication barriers; legal and regulatory obstacles to trading; the lack of adequate regulatory drivers for point source polluters; cheaper alternatives for point source polluters to meet regulatory requirements; and lack of motivation from sellers, e.g., insufficient revenue raised for taking agriculture BMPs (Morgan and Wolverton, 2005; Shortle, 2012). It is clear from the international literature that trading rules should also be clearly established before implementing nutrient offset projects. Trading rules should be designed to facilitate trading but also assure that water quality goals will be achieved. Complex and costly rules can create barriers to trading activities.

The motivation of buyers and sellers can be improved by tightening the policy/regulation and/or adjusting the offset ratios or the credit price. This includes strict nutrient discharge limits from regulation authorities as incentives for polluters to seek alternative options for pollution control. However, regulators should address the feasibility of an enforceable cap in a catchment to achieve the water quality goal, as well as consider the fairness in allocating nutrient reduction requirements among different sources, e.g., point and non-point sources (Filippelli et al., 2022). Active engagement and mediation from a third party that can facilitate offsetting projects can also significantly improve the participation of non-point polluters, such as farmers (Breetz et al., 2005).

The high compliance and transaction costs have been shown to outweigh the gains from trade to participants in some areas, leading to low market activity (fewer participants) and low efficiency of the market scheme (Kostel and Monchak, 2014; Stavins, 1995). Standardized tools, transparent processes, and online registries can be applied to minimize transaction costs. The use of third-party credit verification, aggregation, or audit service providers may add value to market-based programs without being overly burdensome or cost prohibitive to participants. Additionally, extra funding from different organizations, e.g., state governments, local governments, NGOs, and corporate sponsorship to support up-front investments in reducing transaction costs can also reduce the risk of project failure (Stephenson et al., 2010). For example, extra funding would permit examination of site-specific heterogeneity in delivery of nutrients and effectiveness of BMPs, rather than the buyers taking on this responsibility.

There can also be institutional impediments including problems establishing acceptable rules and units of exchange, methods of assigning trade risks, or monitoring or enforcement capabilities (King and Kuch, 2003). A well-designed nutrient offset market can help solve some of these issues (Shortle, 2012).

REFERENCES

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. Front. Mar. Sci. 6, 1–14. https://doi.org/10.3389/fmars.2019.00671
Ahiablame, L.M., Engel, B.A., Chaubey, I., 2012. Effectiveness of low impact development practices: Literature review and suggestions for future research. Water. Air. Soil Pollut. 223, 4253–4273. https://doi.org/10.1007/s11270-012-1189-2
Alexander, R.B., Smith, R.A., Schwarz, G.E., Boyer, E.W., Nolan, J. V., Brakebill, J.W., 2008. Differences in phosphorus and nitrogen delivery to the Gulf of Mexico from the Mississippi River Basin. Environ. Sci. Technol. 42, 822–830. https://doi.org/10.1021/es0716103
Angelopoulos, N. V., Cowx, I.G., Buijse, A.D., 2017. Integrated planning framework for successful river restoration projects: Upscaling lessons learnt from European case studies. Environ. Sci. Policy 76, 12–22. https://doi.org/10.1016/j.envsci.2017.06.005
Arabi, M., Govindaraju, R.S., Engel, B., Hantush, M., 2007. Multiobjective sensitivity analysis of sediment and nitrogen processes with a watershed model. Water Resour. Res. 43, 1–11. https://doi.org/10.1029/2006WR005463
Azimi Sardari, M.R., Bazrafshan, O., Panagopoulos, T., Sardooi, E.R., 2019. Modeling the impact of climate change and land use change scenarios on soil erosion at the Minab Dam Watershed. Sustainability 11, 3353. https://doi.org/10.3390/su11123353
Bartley, R., Hawdon, A.A., Henderson, A., Wilkinson, S., Goodwin, N., Abbott, B.N., Bake, B., Boadle, D., Ahwang, K., 2019. Quantifying the effectiveness of gully remediation on off-site water quality: preliminary results from demonstration sites in the Burdekin catchment (third wet season) - Report to the National Environmental Science Programme. Rep. to Natl. Environ. Sci. Progr. 115 pp.
Bartley, R., Poesen, J., Wilkinson, S., Vanmaercke, M., 2020. A review of the magnitude and response times for sediment yield reductions following the rehabilitation of gullied landscapes. Earth Surf. Process. Landforms 45, 3250–3279. https://doi.org/10.1002/esp.4963
Behera, M., Sena, D.R., Mandal, U., Kashyap, P.S., Dash, S.S., 2020. Integrated GIS-based RUSLE approach for quantification of potential soil erosion under future climate change scenarios. Environ. Monit. Assess. 192. https://doi.org/10.1007/s10661-020-08688-2
Bellwood, D.R., Hughes, T.P., Folke, C., Nyström, M., 2004. Confronting the coral reef crisis. Nature 429, 827–833. https://doi.org/10.1038/nature02691
Black, R.W., Moran, P.W., Frankforter, J.D., 2011. Response of algal metrics to nutrients and physical factors and identification of nutrient thresholds in agricultural streams. Environ. Monit. Assess. 175, 397–417. https://doi.org/10.1007/s10661-010-1539-8
Boleman, P., Jacobson, M., 2021. Nitrogen credit trading as an incentive for riparian buffer establishment on Pennsylvania farmland. Agrofor. Syst. 95, 1033–1045. https://doi.org/10.1007/s10457-021-00595-w
Breetz, H.L., Fisher-Vanden, K., Jacobs, H., Schary, C., 2005. Trust and communication: Mechanisms for increasing farmers' participation in water quality trading. Land Econ. 81, 170–190. https://doi.org/10.3368/le.81.2.170
Brooks, S.S., Lake, P.S., 2007. River restoration in Victoria, Australia: Change is in the wind, and none too soon. Restor. Ecol. 15, 584–591. https://doi.org/10.1111/j.1526-100X.2007.00253.x
Butler, E.C. V, Streten-Joyce, C., Tsang, J.J., Williams, D.K., Alongi, D.M., Furnas, M.J., Mckinnon, A.D., 2013. A procedure for evaluating the nutrient assimilative capacity of Darwin Harbour. Report for Aquatic Health Unit, NT Department of Land Resource Management. Darwin.
Cardoso-Mohedano, J.G., Canales-Delgadillo, J.C., Machain-Castillo, M.L., Sanchez-Muñoz, W.N., Sanchez-Cabeza, J.A., Esqueda-Lara, K., Gómez-Ponce, M.A., Ruiz-Fernández, A.C., Alonso-Rodríguez, R., Lestayo-González, J.A., Merino-Ibarra, M., 2022. Contrasting nutrient distributions during dry and rainy seasons in coastal waters of the southern Gulf of Mexico driven by the Grijalva-Usumacinta River discharges. Mar. Pollut. Bull. 178, 113584. https://doi.org/10.1016/j.marpolbul.2022.113584

Chen, H., Cai, Q., 2006. Impact of hillslope vegetation restoration on gully erosion induced sediment yield. Sci. China, Ser. D Earth Sci. 49, 176–192. https://doi.org/10.1007/s11430-005-0177-4
Chiejine, C., Igboanugo, A., Ezemonye, L., 2014. Modelling effluent assimilative capacity of Ikpoba River, Benin City, Nigeria. Niger. J. Technol. 34, 133. https://doi.org/10.4314/njt.v34i1.17
Cole, L.J., Stockan, J., Helliwell, R., 2020. Managing riparian buffer strips to optimise ecosystem services: A review. Agric. Ecosyst. Environ. 296, 106891. https://doi.org/10.1016/j.agee.2020.106891
Diaz, R.J., Rosenberg, R., 2008. Spreading dead zones and consequences for marine ecosystems. Science (80-.). 321, 926–929. https://doi.org/10.1126/science.1156401
Dietz, M.E., 2007. Low impact development practices: A review of current research and recommendations for future directions. Water. Air. Soil Pollut. 186, 351–363. https://doi.org/10.1007/s11270-007-9484-z
Doriean, N.J.C., Bennett, W.W., Spencer, J.R., Garzon-Garcia, A., Burton, J.M., Teasdale, P.R., Welsh, D.T., Brooks, A.P., 2021. Intensive landscape-scale remediation improves water quality of an alluvial gully located in a Great Barrier Reef catchment. Hydrol. Earth Syst. Sci. 25, 867–883. https://doi.org/10.5194/hess-25-867-2021
Dosskey, M.G., Helmers, M.J., Eisenhauer, D.E., 2006. An approach for using soil surveys to guide the placement of water quality buffers. J. Soil Water Conserv. 61, 344–354.
Doyle, M.W., Patterson, L.A., Chen, Y., Schnier, K.E., Yates, A.J., 2014. Optimizing the scale of markets for water quality trading. Water Resour. Res. 50, 7231–7244. https://doi.org/10.1002/2013WR014979.Reply
Duhon, M., Mcdonald, H., Kerr, S., Duhon, M., Mcdonald, H., Asher, G., Barns, S., Barton, M., Barton, S., Bennetts, T., Bowron, J., Fleming, G., Gardiner, R., Greenhalgh, S., Hayward, N., Mcleod, M., Palmer, J., Reeve, J., 2015. Nitrogen trading in Lake Taupo: An analysis and evaluation of an innovative water management policy. Wellington.
Eheart, J.W., Brill, E.D., Lence, B.J., Kilgore, J.D., Uber, J.G., 1987. Cost efficiency of time-varying discharge permit programs for water quality management. Water Resour. Res. 23, 245–251. https://doi.org/10.1029/WR023i002p00245
Elser, J.J., Bracken, M.E.S., Cleland, E.E., Gruner, D.S., Harpole, W.S., Hillebrand, H., Ngai, J.T., Seabloom, E.W., Shurin, J.B., Smith, J.E., 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecol. Lett. 10, 1135–1142. https://doi.org/10.1111/j.1461-0248.2007.01113.x
EPA, 2022. Nutrient reduction case study Dixie Drain Phosphorus Removal Facility. City of Boise.
EPA, 2015. National summary of state information, watershed assessment, tracking & environmental results. Washington, DC.
EPA, U., 2001. The national costs of the total maximum daily load program. EPA-841-D-01-003. Washington, DC.
Fabricius, K.E., Logan, M., Weeks, S.J., Lewis, S.E., Brodie, J., 2016. Changes in water clarity in response to river discharges on the Great Barrier Reef continental shelf: 2002-2013. Estuar. Coast. Shelf Sci. 173, A1–A15. https://doi.org/10.1016/j.ecss.2016.03.001
Fang, F., Easter, K.W., Brezonik, P.L., 2005. Point-nonpoint source water quality trading: A case study in the Minnesota River Basin. J. Am. Water Resour. Assoc. 41, 645–657. https://doi.org/10.1111/j.1752-1688.2005.tb03761.x
Farhadian, M., Haddad, O.B., Seifollahi-Aghmiuni, S., Loáiciga, H.A., 2015. Assimilative capacity and flow dilution for water quality protection in rivers. J. Hazardous, Toxic, Radioact. Waste 19, 04014027. https://doi.org/10.1061/(asce)hz.2153-5515.0000234
Ferreira, J.G., Bricker, S.B., 2016. Goods and services of extensive aquaculture: shellfish culture and nutrient trading. Aquac. Int. 24, 803–825. https://doi.org/10.1007/s10499-015-9949-9
Filippelli, R., Termansen, M., Hasan, S., Hasler, B., Hansen, L., Smart, J.C.R., 2022. Water quality trading markets – Integrating land and marine based measures under a smart market approach. Ecol. Econ. 200, 107549.

Fisher, J., Acreman, M.C., Fisher, J., Acreman, M.C., 2004. Wetland nutrient removal : a review of the evidence To cite this version : HAL Id : hal-00304953 Wetland nutrient removal : a review of the evidence Wetland nutrient removal : a review of the evidence 8, 673–685.
Folke, C., Carpenter, S., Walker, B., Scheffer, M., Elmqvist, T., Gunderson, L., Holling, C.S., 2004. Regime shifts, resilience, and biodiversity in ecosystem management. Annu. Rev. Ecol. Evol. Syst. 35, 557–581. https://doi.org/10.1146/annurev.ecolsys.35.021103.105711
Fonseca, B.M., Levi, E.E., Jensen, L.W., Graeber, D., Søndergaard, M., Lauridsen, T.L., Jeppesen, E., Davidson, T.A., 2022. Effects of DOC addition from different sources on phytoplankton community in a temperate eutrophic lake: An experimental study exploring lake compartments. Sci. Total Environ. 803, 150049. https://doi.org/10.1016/j.scitotenv.2021.150049
Franklin, H.M., Garzon-garcia, A., Burton, J., Moody, P.W., De Hayr, R.W., Burford, M.A., Hayr, R.W. De, Burford, M.A., 2018. A novel bioassay to assess phytoplankton responses to soil-derived particulate nutrients. Sci. Total Environ. 636, 1470–1479. https://doi.org/10.1016/j.scitotenv.2018.04.195
Fu, B., Merritt, W.S., Croke, B.F.W., Weber, T.R., Jakeman, A.J., 2019. A review of catchment-scale water quality and erosion models and a synthesis of future prospects. Environ. Model. Softw. 114, 75–97. https://doi.org/10.1016/j.envsoft.2018.12.008
Garzon-Garcia, A., Burton, J., Franklin, H.M., Moody, P.W., De Hayr, R.W., Burford, M.A., Hayr, R.W. De, Burford, M.A., De Hayr, R.W., Burford, M.A., 2018a. Indicators of phytoplankton response to particulate nutrient bioavailability in fresh and marine waters of the Great Barrier Reef. Sci. Total Environ. 636, 1416–1427. https://doi.org/10.1016/j.scitotenv.2018.04.334
Garzon-Garcia, A., Burton, J., HEllis, R., Askildsen, M., Finn, L., Moody, P., DeHayr, R., Moody, F.L., R., P.D., 2018b. Sediment particle size and contribution of eroded soils to dissolved inorganic nitrogen export in Great Barrier Reef catchments. Brisbane: Department of Environment and Science, Queensland Government, Brisbane.
Garzon-Garcia, A., Burton, J., Prance, M., Moody, P., DeHayr, R., 2019. Towards the standardisation of bioavailable particulate nitrogen in sediment methods Final report. Brisbane.
Gaylard, S., 2005. A tradeable rights instrument to reduce nutrient pollution in the Port waterways: feasibility study. Adelaide.
GESAMP, 1986. Environmental capacity: An approach to marine pollution prevention, Report Study GESAMP.
Gomez, B., Banbury, K., Marden, M., Trustrum, N.A., Peacock, D.H., Hoskin, P.J., 2003. Gully erosion and sediment production: Te Weraroa Stream, New Zealand. Water Resour. Res. 39, 1–8. https://doi.org/10.1029/2002WR001342
Graeber, D., Tenzin, Y., Stutter, M., Weigelhofer, G., Shatwell, T., von Tümpling, W., Tittel, J., Wachholz, A., Borchardt, D., 2021. Bioavailable DOC: reactive nutrient ratios control heterotrophic nutrient assimilation—An experimental proof of the macronutrient-access hypothesis. Biogeochemistry 155, 1–20. https://doi.org/10.1007/s10533-021-00809-4
Gren, I.M., Lindahl, O., Lindqvist, M., 2009. Values of mussel farming for combating eutrophication: An application to the Baltic Sea. Ecol. Eng. 35, 935–945. https://doi.org/10.1016/j.ecoleng.2008.12.033
Groffman, P.M., Baron, J.S., Blett, T., Gold, A.J., Goodman, I., Gunderson, L.H., Levinson, B.M., Palmer, M.A., Paerl, H.W., Peterson, G.D., Poff, N.L.R., Rejeski, D.W., Reynolds, J.F., Turner, M.G., Weathers, K.C., Wiens, J., 2006. Ecological thresholds: The key to successful environmental management or an important concept with no practical application? Ecosystems 9, 1–13. https://doi.org/10.1007/s10021-003-0142-z
Guildford, S.J., Hecky, R.E., Verburg, P., Albert, A., 2022. Phosphorus limitation in low nitrogen lakes in New Zealand. Inl. Waters. https://doi.org/10.1080/20442041.2021.2015994
Hall, M., 2012. The cost of pollution: Supporting cost-effective options evaluation and pollution reduction. Brisbane.

Hansen, A.T., Dolph, C.L., Foufoula-georgiou, E., Finlay, J.C., 2018. Contribution of wetlands to nitrate removal at the watershed scale. Nat. Geosci. 11, 127–132. https://doi.org/10.1038/s41561-017-0056-6
Hasan, S., Hansen, L.B., Smart, J.C.R., Hasler, B., Termansen, M., 2022. Tradeable nitrogen abatement practices for diffuse agricultural emissions: A 'Smart Market' approach, Environmental and Resource Economics. Springer Netherlands. https://doi.org/10.1007/s10640-022-00657-2
Hashemi Monfared, S.A., Dehghani Darmian, M., Snyder, S.A., Azizyan, G., Pirzadeh, B., Azhdary Moghaddam, M., 2017. Water quality planning in rivers: Assimilative capacity and dilution flow. Bull. Environ. Contam. Toxicol. 99, 531–541. https://doi.org/10.1007/s00128-017-2182-7
Hermoso, V., Pantus, F., Olley, J., Linke, S., Mugodo, J., Lea, P., 2015. Prioritising catchment rehabilitation for multi-objective management: An application from SE-Queensland, Australia. Ecol. Modell. 316, 168–175. https://doi.org/10.1016/j.ecolmodel.2015.08.017
Hoag, D.L.K., Arabi, M., Osmond, D., Ribaudo, M., Motallebi, M., Tasdighi, A., 2017. Policy Utopias for nutrient credit trading programs with nonpoint sources. J. Am. Water Resour. Assoc. 53, 514–520. https://doi.org/10.1111/1752-1688.12532
Hoffmann, C.C., Kjaergaard, C., Uusi-Kämppä, J., Hansen, H.C.B., Kronvang, B., 2009. Phosphorus retention in riparian buffers: Review of their efficiency. J. Environ. Qual. 38, 1942–1955. https://doi.org/10.2134/jeq2008.0087
Horan, R.D., Shortle, J.S., 2017. Endogenous risk and point-nonpoint uncertainty trading ratios. Am. J. Agric. Econ. 99, 427–446. https://doi.org/10.1093/ajae/aaw088
Hunt, R.J., Jardine, T.D., Hamilton, S.K., Bunn, S.E., 2012. Temporal and spatial variation in ecosystem metabolism and food web carbon transfer in a wet-dry tropical river. Freshw. Biol. 57, 435–450. https://doi.org/10.1111/j.1365-2427.2011.02708.x
Jabłońska, E., Wiśniewska, M., Marcinkowski, P., Grygoruk, M., Walton, C.R., Zak, D., Hoffmann, C.C., Larsen, S.E., Trepel, M., Kotowski, W., 2020. Catchment-scale analysis reveals high cost-effectiveness of wetland buffer zones as a remedy to non-point nutrient pollution in north-eastern Poland. Water (Switzerland) 12. https://doi.org/10.3390/w12030629
Jesus, J.M., Danko, A.S., Fiúza, A., Borges, M.T., 2018. Effect of plants in constructed wetlands for organic carbon and nutrient removal: a review of experimeantal factors contributing to higher impact and suggestions for future guidelines. Environ. Sci. Pollut. Res. 25, 4149–4164. https://doi.org/10.1007/s11356-017-0982-2
Karssies, L.E., Prosser, I.P., 1999. Guidelines for riparian filter strips for Queensland irrigators.
Kavehei, E., Roberts, M.E., Cadier, C., Griffiths, M., Argent, S., Hamilton, D.P., Lu, J., Bayley, M., Adame, M.F., 2021. Nitrogen processing by treatment wetlands in a tropical catchment dominated by agricultural landuse. Mar. Pollut. Bull. 172. https://doi.org/10.1016/j.marpolbul.2021.112800
Kay, D., Crowther, J., Stapleton, C.M., Wyer, M.D., 2018. Faecal indicator organism inputs to watercourses from streamside pastures grazed by cattle: Before and after implementation of streambank fencing. Water Res. 143, 229–239. https://doi.org/10.1016/j.watres.2018.06.046
King, D.M., Kuch, P.J., 2003. Will nutrient credit trading ever work? An assessment of supply and demand problems and institutional obstacles. News Anal. 5, 10352–10368.
Koci, J., Wilkinson, S.N., Hawdon, A.A., Kinsey-Henderson, A.E., Bartley, R., Goodwin, N.R., 2021. Rehabilitation effects on gully sediment yields and vegetation in a savanna rangeland. Earth Surf. Process. Landforms 46, 1007–1025. https://doi.org/10.1002/esp.5076
Kometa, C.G., 2019. Drivers of watershed degradation and its implications on potable water supply in the menchum river basin of cameroon. World J. Soc. Sci. Res. 6, 483. https://doi.org/10.22158/wjssr.v6n4p483
Kostel, J.A., Monchak, J., 2014. Feasibility assessment of a nutrient trading market in the Big Bureau Creek watershed. Final Report.

Laceby, J.P., Saxton, N.E., Smolders, K., Kemp, J., Faggotter, S.J., Ellison, T., Ward, D., Stewart, M., Burford, M.A., 2017. The effect of riparian restoration on channel complexity and soil nutrients. Mar. Freshw. Res. 68, 2041–2051. https://doi.org/10.1071/MF16338
Land, M., Granéli, W., Grimvall, A., Hoffmann, C.C., Mitsch, W.J., Tonderski, K.S., Verhoeven, J.T.A.A., 2016. How effective are created or restored freshwater wetlands for nitrogen and phosphorus removal? A systematic review. Environ. Evid. 5, 1–26. https://doi.org/10.1186/s13750-016-0060-0
Landis, W.G., 2008. Assimilative Capacity. Ecotoxicology 264–268. https://doi.org/10.1016/B978-008045405-4.00371-2
Li, Z., Zhang, Y., Zhu, Q., He, Y., Yao, W., 2015. Assessment of bank gully development and vegetation coverage on the Chinese Loess Plateau. Geomorphology 228, 462–469. https://doi.org/10.1016/j.geomorph.2014.10.005
Lichtenberg, E., 2004. Some hard truths about agriculture and the environment. Agric. Resour. Econ. Rev. 33, 24–33. https://doi.org/10.1017/S106828050000561X
Liu, X., Zhang, X., Zhang, M., 2008. Major factors influencing the efficacy of vegetated buffers on sediment trapping: A review and analysis. J. Environ. Qual. 37, 1667–1674. https://doi.org/10.2134/jeq2007.0437
Liu, Y., Engel, B.A., Flanagan, D.C., Gitau, M.W., McMillan, S.K., Chaubey, I., 2017. A review on effectiveness of best management practices in improving hydrology and water quality: Needs and opportunities. Sci. Total Environ. 601–602, 580–593. https://doi.org/10.1016/j.scitotenv.2017.05.212
Lötjönen, S., Ollikainen, M., Kotamäki, N., Huttunen, M., Huttunen, I., 2021. Nutrient load compensation as a means of maintaining the good ecological status of surface waters. Ecol. Econ. 188. https://doi.org/10.1016/j.ecolecon.2021.107108
Lu, H., Moran, C.J., Prosser, I.P., 2006. Modelling sediment delivery ratio over the Murray Darling Basin. Environ. Model. Softw. 21, 1297–1308. https://doi.org/10.1016/j.envsoft.2005.04.021
Mannino, I., Franco, D., Piccioni, E., Favero, L., Mattiuzzo, E., Zanetto, G., 2008. A cost-effectiveness analysis of seminatural wetlands and activated sludge wastewater-treatment systems. Environ. Manage. 41, 118–129. https://doi.org/10.1007/s00267-007-9001-6
Mayer, P.M., Reynolds, S.K., McCutchen, M.D., Canfield, T.J., 2007. Meta-analysis of nitrogen removal in riparian buffers. J. Environ. Qual. 36, 1172. https://doi.org/10.1111/j.1365-3091.1966.tb01583.x
McCulloch, M., Fallon, S., Wyndham, T., Hendy, E., Lough, J., Barnes, D., 2003. Coral record of increased sediment flux to the inner Great Barrier Reef since European settlement. Nature 421, 727–730. https://doi.org/10.1038/nature01361
McPherson, T.N., Burian, S.J., Stenstrom, M.K., Turin, H.J., Brown, M.J., Suffet, I.H., 2005. Dry and wet weather flow nutrient loads from a Los Angeles watershed. J. Am. Water Resour. Assoc. 41, 959–969. https://doi.org/10.1111/j.1752-1688.2005.tb03780.x
McPherson, T.N., Burian, S.J., Turin, H.J., Stenstrom, M.K., Suffet, I.H., 2002. Comparison of the pollutant loads in dry and wet weather runoff in a southern California urban watershed. Water Sci. Technol. 45, 255–261. https://doi.org/10.2166/wst.2002.0252
Morgan, C., Wolverton, A., 2005. Water quality trading in the United States. Washington, DC.
Mueller, J.M., Soder, A.B., Springer, A.E., 2019. Valuing attributes of forest restoration in a semi-arid watershed. Landsc. Urban Plan. 184, 78–87. https://doi.org/10.1016/j.landurbplan.2018.12.012
Muirhead, R.W., 2019. The effectiveness of streambank fencing to improve microbial water quality: A review. Agric. Water Manag. 223. https://doi.org/10.1016/j.agwat.2019.105684
O'Grady, D., 2008. Point to non-point phosphorus trading in the South Nation River watershed. WIT Trans. Ecol. Environ. 108, 189–195. https://doi.org/10.2495/EEIA080191
O'Mara, J.O., Hardie, R., White, K., Jackson, C., Belz, P., Clouston, B., Norrie, F., 2014. Water quality offset investment in riparian restoration to reduce sewerage treatment costs - Part 1: determining the sediment loss avoided through riparian restoration, in: Proceedings of the 7th Australian Stream Management Conference. pp. 123–132.

Obin, N., Tao, H., Ge, F., Liu, X., 2021. Research on water quality simulation and water environmental capacity in lushui river based on wasp model. Water 13. https://doi.org/10.3390/w13202819
Oldenborg, K.A., Steinman, A.D., 2019. Impact of sediment dredging on sediment phosphorus flux in a restored riparian wetland. Sci. Total Environ. 650, 1969–1979. https://doi.org/10.1016/j.scitotenv.2018.09.298
Olley, J.M., Murray, A.S., Wallbrink, P.J., 1996. Identifying sediment sources in a partially logged catchment using natural and anthropogenic radioactivity. Zeitschrift fur Geomorphol. Suppl. 105, 111–127.
Pantus, F., Hermoso, V., Lae, P., Olley, J., Linke, S., Mugodo, J., 2011. Catchment Rehabilitation Planner. A report to the Healthy Waterways Partnership Ltd., Australia.
Peternel-Staggs, K., Saito, L., Fritsen, C.H., 2008. Evaluation of a Modeling Approach to Assess Nitrogen Assimilative Capacity due to River Restoration. J. Water Resour. Plan. Manag. 134, 474–486. https://doi.org/10.1061/(asce)0733-9496(2008)134:5(474)
Pickett, S.T.A., 1989. Space-for-Time substitution as an alternative to long-term studies, in: Long-Term Studies in Ecology. pp. 110–135.
Prosser, I., Karssies, L., 2001. Designing filter strips to trap sediment and attached nutrient.
Pruski, F.F., Nearing, M.A., 2002. Climate-induced changes in erosion during the 21st century for eight U.S. locations. Water Resour. Res. 38, 34-1-34-11. https://doi.org/10.1029/2001wr000493
Puzyreva, M., Roy, D., Stanley, M., 2019. Case Study Research on Offsets for Water Quality Management, International Institute for Sustainable Development.
Qiu, Z., 2013. Comparative assessment of stormwater and nonpoint source pollution best management practices in suburban watershed management. Water (Switzerland) 5, 280–291. https://doi.org/10.3390/w5010280
Renaud, F.G., Sebesvari, Z., Gain, A.K., 2021. Assessment of land/catchment use and degradation, in: Handbook of Water Resources Management: Discourses, Concepts and Examples. Springer, pp. 471–487. https://doi.org/https://doi.org/10.1007/978-3-030-60147-8_15
Rinne, J.N., 1999. Fish and grazing relationships: The facts and some pleas. Fisheries 24, 12–21. https://doi.org/10.1577/1548-8446(1999)024<0012:fagr>2.0.co;2
Rittenburg, R.A., Squires, A.L., Boll, J., Brooks, E.S., Easton, Z.M., Steenhuis, T.S., 2015. Agricultural BMP effectiveness and dominant hydrological flow paths: Concepts and a review. J. Am. Water Resour. Assoc. 51, 305–329. https://doi.org/10.1111/1752-1688.12293
Roberts, M.E., Lu, J., Kavehei, E., Adame, M.F., 2021. A sensitivity analysis of a water quality model for a constructed wetland, in: Modelling Water Treatment and Water Quality Management Systems.
Roni, P., Beechie, T.J., Bilby, R.E., Leonetti, F.E., Pollock, M.M., Pess, G.R., 2002. A review of stream restoration techniques and a hierarchical strategy for prioritizing restoration in Pacific Northwest Watersheds. North Am. J. Fish. Manag. 22, 1–20. https://doi.org/10.1577/1548-8675(2002)022<0001:arosrt>2.0.co;2
Roni, P., Hanson, K., Beechie, T., 2008. Global review of the physical and biological effectiveness of stream habitat rehabilitation techniques. North Am. J. Fish. Manag. 28, 856–890. https://doi.org/10.1577/m06-169.1
Sado, Y., Boisvert, R.N., Poe, G.L., 2010. Potential cost savings from discharge allowance trading: A case study and implications for water quality trading. Water Resour. Res. 46, 1–12. https://doi.org/10.1029/2009WR007787
Scheffer, M., 1990. Multiplicity of stable states in freshwater systems. Hydrobiologia 200/201, 475–486. https://doi.org/10.1007/978-94-017-0924-8_42
Scheffer, M., Carpenter, S.R., 2003. Catastrophic regime shifts in ecosystems: Linking theory to observation. Trends Ecol. Evol. 18, 648–656. https://doi.org/10.1016/j.tree.2003.09.002
Selman, M., Greenhalgh, S., Branosky, E., Jones, C.Y., Guiling, J., 2009. Water quality trading programs: An international overview, Environmental Protection. Washington, DC 20002.
Sharma, A.K., Pezzaniti, D., Myers, B., Cook, S., Tjandraatmadja, G., Chacko, P., Chavoshi, S., Kemp, D., Leonard, R., Koth, B., Walton, A., 2016. Water sensitive urban design: An investigation of current systems, implementation drivers, community perceptions and potential to supplement urban water services. Water (Switzerland) 8. https://doi.org/10.3390/w8070272

Shortle, J., 2012. Water quality trading in agriculture, OECD Pblishing. Paris. https://doi.org/10.1007/978-3-319-18287-2_15
Sirivedhin, T., Gray, K.A., 2006. Factors affecting denitrification rates in experimental wetlands: Field and laboratory studies. Ecol. Eng. 26, 167–181. https://doi.org/10.1016/j.ecoleng.2005.09.001
Srinivas, R., Singh, A.P., Dhadse, K., Garg, C., 2020. An evidence based integrated watershed modelling system to assess the impact of non-point source pollution in the riverine ecosystem. J. Clean. Prod. 246, 118963. https://doi.org/10.1016/j.jclepro.2019.118963
Stavins, R.N., 1995. Transaction costs and tradeable permits. J. Environ. Econ. Manage. https://doi.org/10.1006/jeem.1995.1036
Stephenson, K., Aultman, S., Metcalfe, T., Miller, A., 2010. An evaluation of nutrient nonpoint offset trading in Virginia: A role for agricultural nonpoint sources? Water Resour. Res. 46, 1–11. https://doi.org/10.1029/2009WR008228
Stevenson, J.J., Bennett, B.J., Jordan, D.N., French, R.D., 2012. Phosphorus regulates stream injury by filamentous green algae, DO, and pH with thresholds in responses. Hydrobiologia 695, 25–42. https://doi.org/10.1007/s10750-012-1118-9
Sunohara, M.D., Topp, E., Wilkes, G., Gottschall, N., Neumann, N., Ruecker, N., Jones, T.H., Edge, T.A., Marti, R., Lapen, D.R., 2012. Impact of riparian zone protection from cattle on nutrient, bacteria, F-coliphage, Cryptosporidium , and Giardia loading of an intermittent stream. J. Environ. Qual. 41, 1301–1314. https://doi.org/10.2134/jeq2011.0407
Sweeney, B.W., Newbold, J.D., 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. J. Am. Water Resour. Assoc. 50, 560–584. https://doi.org/10.1111/jawr.12203
Torres-Bejarano, F.M., Verbel-Escobar, M., Atencia-Osorio, M.C., 2023. Water quality model-based methodology to assess assimilative capacity of waste - water discharges in rivers. Glob. J. Environ. Sci. Manag. 8, 449–472. https://doi.org/10.22034/gjesm.2022.04.01
Townsend, C.R., Uhlmann, S.S., Matthaei, C.D., 2008. Individual and combined responses of stream ecosystems to multiple stressors. J. Appl. Ecol. 45, 1321–1329. https://doi.org/10.1111/j.1365-2664.2007.0
Villota-López, C., Rodríguez-Cuevas, C., Torres-Bejarano, F., Cisneros-Pérez, R., Cisneros-Almazán, R., Couder-Castañeda, C., 2021. Applying EFDC Explorer model in the Gallinas River, Mexico to estimate its assimilation capacity for water quality protection. Sci. Rep. 11, 1–16. https://doi.org/10.1038/s41598-021-92453-z
Wagenhoff, A., Townsend, C.R., Phillips, N., Matthaei, C.D., 2011. Subsidy-stress and multiple-stressor effects along gradients of deposited fine sediment and dissolved nutrients in a regional set of streams and rivers. Freshw. Biol. 56, 1916–1936. https://doi.org/10.1111/j.1365-2427.2011.02619.x
Wainger, L.A., King, D., 2007. Establishing trading ratios for point- non-point source water quality trades: Can we capture environmental variability without breaking the bank? Maryland.
Wasson, R.J., Mazari, R.K., Starr, B., Clifton, G., 1998. The recent history of erosion and sedimentation on the Southern Tablelands of southeastern Australia: sediment flux dominated by channel incision. Geomorphology 24, 291–308. https://doi.org/10.1016/S0169-555X(98)00019-1
Wild-Allen, K., Herzfeld, M., Thompson, P.A., Rosebrock, U., Parslow, J., Volkman, J.K., 2010. Applied coastal biogeochemical modelling to quantify the environmental impact of fish farm nutrients and inform managers. J. Mar. Syst. 81, 134–147. https://doi.org/10.1016/j.jmarsys.2009.12.013
Wild-Allen, K., Thompson, P.A., Volkman, J.K., Parslow, J., 2011. Use of a coastal biogeochemical model to select environmental monitoring sites. J. Mar. Syst. 88, 120–127. https://doi.org/10.1016/j.jmarsys.2011.02.017
Wong, T.H.F., 2006. An overview of water sensitive urban design practices in Australia. Water Pract. Technol. 1. https://doi.org/10.2166/wpt.2006018
Xu, H., Paerl, H.W., Qin, B., Zhu, G., Hall, N.S., Wu, Y., 2015. Determining critical nutrient thresholds needed to control harmful cyanobacterial blooms in eutrophic Lake Taihu, China. Environ. Sci. Technol. 49, 1051–1059. https://doi.org/10.1021/es503744q
Xu, Z., Yin, X., Yang, Z., Cai, Y., Sun, T., 2016. New model to assessing nutrient assimilative capacity in plant-dominated lakes: Considering ecological effects of hydrological changes. Ecol. Modell. 332, 94–102. https://doi.org/10.1016/j.ecolmodel.2016.03.019