Assessing the cost-effectiveness of water quality interventions in South-East Queensland

*K. Alam, J. Rolfe and P. Donaghy**

The focus of this article is on the cost-effectiveness of mitigation strategies to reduce pollution loads and improve water quality in South-East Queensland. Scenarios were developed about the types of catchment interventions that could be considered, and the resulting changes in water quality indicators that may result. Once these catchment scenarios were modelled, the range of expected outcomes was assessed and the costs of mitigation interventions were estimated. Strategies considered include point and non-point source interventions. Predicted reductions in pollution levels were calculated for each action based on the expected population growth. The cost of the interventions included the full investment and annual running costs as well as planned public investment by the state agencies. Cost-effectiveness of strategies is likely to vary according to whether suspended sediments, nitrogen or phosphorus loads are being targeted.

Keywords: catchment modelling, cost-effectiveness analysis, environmental assets, water quality objectives

Increasing scale of economic activities together with rising populations has led to large increases in consumption and waste outputs in many Australian watersheds. The latter includes wastes discharged into waterways, which reduce levels of water quality and have subsequent economic, social (including public health) and environmental impacts. To address this issue, public investments in water quality improvement have increased substantially at all levels in Australia in recent years. For instance, under the *National Action Plan for Salinity and Water Quality,* $1.4 billion has been committed for 2002-09 with $162 million to be spent in Queensland to address salinity and water quality issues.

The *Reef Water Quality Protection Plan*, a joint initiative of the Australian and Queensland Governments, is now in operation to halt and reverse the decline in water quality by reducing land-sourced pollutants entering the Great Barrier Reef (GBR) lagoon and by rehabilitating and conserving wetlands, riparian zones and floodplain areas. In addition to these national and regional initiatives, there are substantial investments in public infrastructure such as sewerage treatment plants, and tighter controls over emissions from private industry.

Increasing commitments of public funding can generate questions about the economic efficiency of such investments. Queensland Treasury (1997) requires strategies and options in addressing significant environmental concerns to be identified and valued to assist in the ranking of alternative investment options. Considering the constraints and competing uses of resources, optimality in resource allocation is important. Economic analysis plays an important role in assessing the desirability of public investment.

An economic analysis of water quality improvement requires proper estimation of costs and benefits of different mitigation strategies to assess the desirability of particular interventions. Estimation of mitigation costs is often a comparatively easy task as information is available either within relevant public agencies or from market transactions. However, estimation of mitigation benefits tends to be more difficult, mainly due to many of the benefits not being included directly in market transactions. Improved or maintained water quality can generate direct use benefits (e.g. direct recreation), indirect use benefits (e.g. impact on health risks) and non-use benefits (e.g. protection of biodiversity and cultural heritage). These may not be priced in markets.

Many improvements in water quality (or avoidance of deterioration) are not included in market transactions because they have non-rival and non-excludable characteristics. One consequence is that government intervention is typically needed to address water quality issues. Another consequence is that information about the costs and benefits of such intervention is difficult to assess, and therefore can be difficult to include in an economic analysis.

The standard economic assessment tool used to evaluate the net benefits of an intervention measure in an

** Khorshed Alam is with the School of Accounting, Economics and Finance, Faculty of Business, University of Southern Queensland, Toowoomba; John Rolfe is with the Faculty of Business and Informatics, Central Queensland University, Rockhampton; and Peter Donaghy is with the Queensland Department of Primary Industries and Fisheries, Rockhampton. Email: alam@usq.edu.au*

economic welfare framework is cost benefit analysis (CBA). CBA operates by identifying, valuing, discounting and then comparing the costs and benefits that flow from a particular intervention strategy. Where a desired outcome has already been established, then cost-effectiveness analysis (CEA) may be employed. CEA determines the least-cost option of achieving a given target, and focuses on identifying the most cost-efficient ways of achieving set outcomes.

In assessing investments for water quality improvement, CBA is the most appropriate methodology to evaluate different policy options and the desirability of investment. However, in some situations CBA is difficult to apply because of issues involved in identifying and valuing different impacts (Gerasidi et al. 2003), and the difficulty of linking particular mitigation actions with community benefits (Alam et al. 2006). Where there is incomplete knowledge and high levels of uncertainty, decisions about resource allocation are often made through political processes. In these cases, the key policy question often becomes one of how to most efficiently meet the objectives that have been set by other processes. A CEA can be appropriate for this purpose, because it can avoid the difficulties of measuring benefits of environmental improvements by 'focusing on the costs of achieving a quantified non-economic objective' (Keplinger & Santhi 2002, p. 206).

CEA is being widely used in resource allocation decision-making. The technique is used extensively in the health industry to evaluate the most efficient ways of achieving certain health outcomes, where health-related benefits are usually expressed in a single measure, such as life years saved or quality-adjusted life-years (Abelson 2003), or disability adjusted life years (Fox-Rushby & Hanson 2001). The advantages of this approach are that the benefits of programs do not have to be measured (because the goals are already set), meaning that the analytical focus is on measuring and evaluating costs.

A review by Zanou et al. (2003) revealed that the majority (approximately 80 per cent) of the applications of CEA were in the area of health care. CEA is also used in other areas including water quality improvement and the identification of cost-effective pollution load reductions (see, for example, Gren et al. 1997a, 1997b; Schou et al. 2006). Gren et al. (1997a) calculated cost-effective nutrient reductions to the Baltic Sea. They included both point and non-point sources of pollution and considered two pollutants - nitrogen and phosphorus - in the Baltic Sea drainage basin. Cost-effective nutrient reduction measures were estimated for three different scenarios: reductions in either nitrogen or phosphorus loads to the coastal waters, and reductions in both nutrients by the same percentages. They found that the cost of reducing the load of nitrogen was higher than that of corresponding decreases in phosphorus loads, that is, at the 50 percent reduction levels, the cost of nitrogen reductions was about five times as high as the cost of phosphorus reduction. Elofsson (1997) also estimated cost-effective nitrogen reductions from the agricultural sector in the nine countries around the Baltic Sea basin.

Using a linear programming model, Schleich et al. (1996) calculated the total cost of achieving a 50 per cent phosphorus load reduction target established in various locations throughout the Fox-Wolf River basin in north-east Wisconsin. Lise and van der Veeren (2002) assessed cost-effective nutrient emission reductions in the Rhine River basin. They calculated the cost-effective joint nitrogen and phosphorus emission reduction to achieve a desired load to the North Sea. Yuan et al. (2002) applied CEA to evaluate the cost-effectiveness of alternative agricultural best management practices for sediment reduction in the Mississippi Delta. Using the Annualized Agricultural Non-point Source pollutant loading model (AnnAGNPS), the impacts of several combinations of best management practices on sediment yield were assessed, and the most cost-effective best management practices were identified. Atkins and Burdon (2006) estimated the benefits and costs of reducing eutrophication of the Randers Fjord in Denmark.

Previous studies on water quality improvement have focused mainly on nutrient reduction issues (Schleich et al. 1996; Gren et al. 1997a,1997b). However, there has been little work done in Australia to determine the economic efficiency of actions to achieve locally specific water quality objectives. The cost-effectiveness of non-point source best management practices received attention from policy makers and managers only recently during the implementation of the South-East Queensland Regional Water Quality Management Strategy in Australia. In this study we explore the broad economic case for improving water quality in South-East Queensland by using CEA to determine the most cost-effective mitigation strategy for achieving a new set of water quality objectives. This study differs from others with respect to the spatial scale, the inclusion of different types of point and non-point source polluters together, and consideration of different management scenarios.

This article is organised in the following way. The next section contains a brief description of water quality policy issues in Queensland and the study area.

Following this is an overview of the CEA technique. The results of the case study analysis on intervention strategies, estimates of costs and cost-effective pollutant load reductions are presented in the fourth section. The fifth section concludes the article.

Case study background

Water quality policy issues in Queensland

Water quality in Queensland is protected by the *Environmental Protection (Water) Policy 1997*. The management framework for achieving sustainable development of Queensland's water resources under this legislation, with respect to water quality, includes:

- identifying environmental values for Queensland waters to be protected in consultation with industry, government and the community
- deciding and stating water quality guidelines and objectives to protect environmental values
- integrating environmental values into the management of natural resources and making decisions about Queensland waters that promote efficient use of resources and best practice environmental management.

Environmental values are the categories and aspects of water use that communities think are important. Environmental values can be thought of as some measure of the differing impacts on society. These impacts are related to the qualities of waters that need to be protected from the effects of pollution and waste discharges to ensure healthy aquatic ecosystems and waters that are safe for recreation and productive use. Water quality objectives are measures of water quality indicators (including physical, chemical or biological measures) that protect the environmental values of the water.

The *Environmental Protection (Water) Policy* provides uniform water quality standards for all water bodies throughout the state. It covers a range of issues including the setting of environmental values for water quality and the establishment of water quality objectives for all water bodies in Queensland. However, water quality varies naturally due to location-specific variation in rainfall and runoff patterns, river discharge, land use, geology and soil type, topography (slope length and gradient) and land cover conditions. Therefore, irrespective of the level of pollutant load entering into a specific water body, the Policy provides the same environmental controls as throughout the State. Against this backdrop, the Queensland Environmental Protection Agency (EPA) has developed environmental values and water quality objectives for fresh, estuarine and coastal/marine waters of the Moreton Bay in South-East Queensland along with two other regions in Queensland (EPA 2004b). The aim of this initiative is to determine locally specific environmental values and water quality objectives and to integrate these values and objectives into existing legislation. For this, Schedule 1 (environmental values and water quality objectives for waters) of the *Environmental Protection (Water) Policy 1997* is proposed to change. When these environmental values and water quality objectives are integrated into the existing legislation, it will have a strict standard for waters in Queensland. This will provide better protection for environmental assets through achieving higher water quality standards.

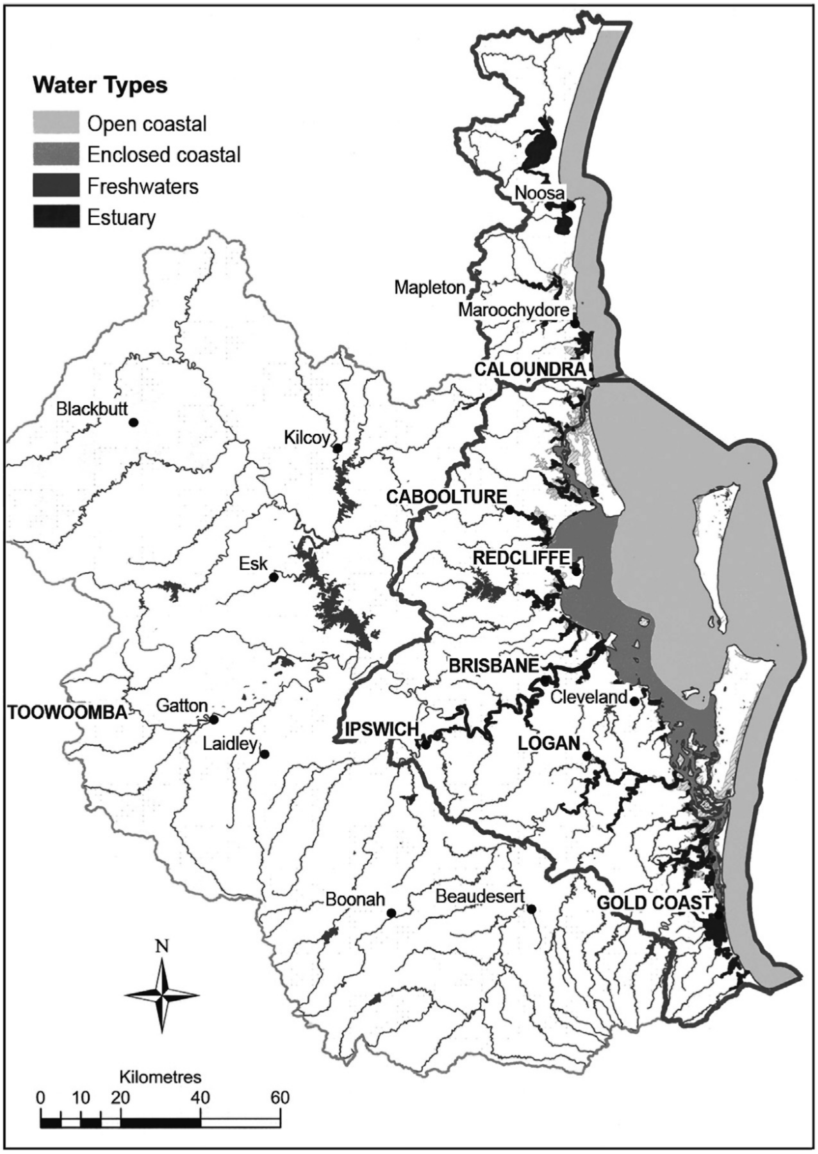
The study area

The South-East Queensland (SEQ) region occupies 22 415 square kilometres or 1.3 per cent of the area of Queensland (Map 1). With an estimated resident population of 2 654 000 in 2004 (OUM 2004), SEQ is Australia's fastest growing metropolitan region enjoying consistently high rates of intra and interstate net migration. Moreton Bay is a highly urbanised region with strong population and development growth centered around Brisbane, the capital city of Queensland. The bay is of national and international environmental significance, as recognized through the *Ramsar Agreement* and the declaration of the bay as a marine park by the state government.

Moreton Bay is the receiving water body for rivers and streams of a catchment area of 21 220 square kilometres, compared to the bay area itself of 1523 square kilometres. This represents about a 14:1 ratio of catchment to bay area (Dennison & Abal 1999). Land used for agriculture, grazing and private forestry accounts for 71 per cent of the catchment area, with urban and rural residential uses occupying 11 per cent and public lands 17 per cent (Capelin et al. 1998). In recent years, however, urban development has become the most dominant form of land use change due to economic growth and increasing population pressure.

The geographic scope of the SEQ study region includes:

- estuarine waters from Noosa to the Gold Coast (including Noosa, Maroochy, Mooloolah, Caboolture, Pumicestone Passage, Pine, Brisbane, Logan, Bremer, Albert, Coomera, Nerang and Gold Coast estuaries and creeks)
- Moreton Bay, the Broadwater and Queensland coastal waters



Source: Strategic Projects, EPA (2004) (reproduced from Rolfe et al. 2005)

Figure 1 **South-East Queensland study area**

- coastal catchments freshwaters (excluding the Logan, Albert, Bremer, Lockyer and Brisbane catchments).

The SEQ as a whole is characterised by high variability in water quality levels and issues. There are some areas in the region that are in close to pristine condition, while other parts have serious and declining levels of water quality. Moreton Bay has been receiving an increasing load of pollutants, principally nutrients, sediments and phosphorus chiefly due to human activities and catchment and land use changes (Neil 1998). Abal and Rogers (1999) reported that in the last 50-80 years in the Brisbane River, nitrate had increased by 22-fold, phosphate by 11-fold and suspended sediments by 4-fold.

The threats to water quality in the SEQ come from a variety of sources, broadly categorized as point and non-point sources. Protection of environmental assets requires the effective assessment and understanding of the sources of pollution loads entering the waterways so that mitigation strategies can be targeted to achieve water quality objectives.

To meet the water quality standards, reductions in the discharge of nutrient and sediments into the SEQ waters are required from all point and non-point sources. The South-East Queensland Regional Water Quality Management Strategy undertook a series of scientific studies and research programs to design and implement management strategies to deal with water quality and ecosystem health issues in waterways throughout the whole SEQ region (MBWCP 2001; WBM Oceanics Australia 2001; Low Choy et al. 2002). Sediment, nitrogen and phosphorus were identified as key pollutants in the SEQ. Runoff with high sediment, nitrogen and phosphorus concentrations was identified as a priority impact on water quality in SEQ. Non-point sources contributed more than 95 per cent of the suspended sediment load, while the majority of nitrogen and phosphorus flowing the waterways usually originated from point sources.

The Moreton Bay Catchment Water Quality Management Strategy cites sediment as a major cause of water quality degradation in western and southern Moreton Bay, particularly in Bramble Bay (Healthy Waterways 2001). The major source of sediment is storm water runoff from urban and rural areas. The majority of nitrogen entering SEQ waterways during dry periods originates from sewerage treatment plants. During periods of rainfall, urban and rural stormwater runoff also contributes nitrogen to waterways. Excessive nutrient and phosphorus inputs in some hotspots in the region are placing pressure on regulatory authorities to adopt tighter controls over nitrogen and phosphorus discharges into waters.

Water resource assets in SEQ provide a variety of important functions and uses. Some of these assets have very high water quality standards, and improved protection measures will help to maintain them. In other cases, assets are threatened by low or declining water quality levels, and improved protection measures are needed to protect or remediate assets. In many waterways, current loads are causing continued deterioration in water quality even before additional loads are considered. If water quality levels continue to decline, then a number of adverse economic and social impacts over the short, medium and longer term are expected (Rolfe et al. 2005).

Against this setting, the EPA has developed environmental values for discrete reaches of rivers, estuaries and coastal areas, with different categorisations for the study region. At an operational level, water quality objectives will need to be adjusted to suit each discrete reach of river, estuary and coastal area, so there will be many water quality objectives across the region (EPA 2004b, 2004c). This will provide location-specific water quality objectives in the study region.

Cost-effectiveness analysis

The purpose of a CEA in assessing water quality improvement is to ascertain which mitigation strategy or combination of strategies can achieve a set of environmental outcomes at the lowest cost. The underlying assumption is that different alternatives are associated with different costs and different environmental outcomes. By choosing those with the least cost for a given outcome, society can use its resources more effectively (Levin 1995).

A CEA of improved water quality can consist of the following steps:

(a) identify the water quality target to be met: This is typically set through a political process.

(b) determine potential mitigation strategies: The next step toward conducting a CEA is to identify the intervention strategies available to achieve the desired environmental outcome. The impact of different mitigation strategies can be assessed using hydrological or catchment modelling. The modelling outputs provide pollutant load reductions under different scenarios designed for a study. For example, the same level of water quality improvement may be achieved by different strategies that focus on urban, industrial or agricultural emissions.

(c) estimate investment costs: After alternative mitigation strategies have been identified, it is important to have estimates of intervention costs. In many cases the costs, such as production losses, are assessed from market data, but there may also be non-market costs to consider in some cases. The transaction costs associated with different mitigation strategies should also be assessed.

(d) calculate cost-effectiveness of the alternatives being considered: Once both the mitigation strategies and their associated costs of intervention are known, the efficiency of different actions can be assessed. This may also involve some assessment of the risks and uncertainties associated with the different mitigation strategies.

A CEA typically describes an intervention in terms of the ratio of incremental costs per unit of incremental outcome (Garber & Phelps 1997). In these cases the output is a ratio for each intervention, with the numerator showing costs and the denominator measuring intervention outcomes. CEA translates the environmental outcomes into a common denominator, for instance, the costs per reduced tonne of phosphorus and nitrogen. A simple form of CEA involves the comparison of cost-effectiveness ratios.

In the case of water quality improvement, reduction of pollutant loads into waterways is defined as the target, and cost-effectiveness means achieving the most amount of load reduction per monetary unit of cost. In that case, it is necessary to convert total costs to a per tonne load reduction cost figure for comparing cost-effectiveness of alternative interventions.

Case study results and discussion

Identifying water quality outcomes

The benefits of protecting environmental assets in the case study area were assessed by catchment modelling of pollutant loads. The Environment Management Support System software was used for the scenario analysis within SEQ catchments. Model outputs included the predicted diffuse loads to waterways in response to modelling scenarios. Point source estuary loads were also included to examine the overall predicted load impact of possible interventions. Estimates were made of total point and diffuse source loads for each of the major catchments in SEQ[1]. Suspended sediment, nitrogen and phosphorus loads were used as surrogate indicators of the characteristics needed to protect environmental assets in SEQ waterways. These objectives included a range of physical, chemical and biological parameters all of which provide a detailed description of catchment and overall water quality condition. Water quality indicators were expressed as annual loads to waterways.

1. Load modelling scenarios used in this paper are reported in Rolfe et al. (2005) and were estimated by WBM Oceanics Australia (2004).

Table 1 **Summary of modelled reductions in sediment, nutrient and phosphorus loads under different intervention scenarios**

	TSS (tonnes)	TN (tonnes)	TP (tonnes)
Base case (2004)	240 000	3800	1360
No intervention (2026)	280 000	5200	2000
Intervention (2026)	150 000	3100	540
Difference by 2026	130 000	2100	1460

Selected intervention scenarios (as surrogates for a wider range of possible management actions) included a range of planned and possible future actions by both government and the community (including industry), targeting the reduction of urban and rural point source and diffuse source loads emitted to waterways. Possible interventions focused on the setting of water quality objectives to protect the environmental assets for the waters in the project area. Such interventions were aimed at initially halting the decline of aquatic ecosystems and, over time, achieving sustainable management of the water environment. Possible interventions included both existing programs, such as the upgrades of sewerage treatment plants, and projected activities such as the restoration of riparian areas.

Based on the identified sources of pollution load, mitigation strategies were designed. For modelling pollutant load reduction in the catchments, three broad scenarios were considered (WBM Oceanics Australia 2004 and McMahon 2004 reported in Rolfe et al. 2005). The scenarios involved an assessment of expected annual pollutant loads for:

- *base case* scenario: the existing situation in 2004
- *no intervention* scenario: if no further management actions are implemented up until 2026
- *intervention* scenario: if a range of management actions and interventions are implemented up until 2026.

The scenarios defined above vary depending on whether or not management intervention strategies are implemented to address declining water quality. In the *no intervention* scenario water quality levels were projected to decline in line with current trends and increasing populations. This is a modelling scenario that does not include a number of current government and community initiatives. In the *intervention* scenario, management intervention strategies were introduced that enhance or protect water quality in spite of population increases, economic development and land use changes.

The key focus was on the cost-effectiveness of protecting the environmental values by achieving the water quality objectives through investing in water quality measures under the *intervention* scenario (including both current and future programs). These interventions would ensure two key components are achieved. The first is to avoid further reductions in water quality, and the second is to enhance the water quality above current levels at the end. In assessing the benefits of the intervention strategy, the appropriate comparison was between *no intervention* and *intervention* scenarios, as this represented the total improvement gained.

To make the modelling task more manageable, the scenarios were simplified in three important ways. Firstly, only a selected number of potential mitigation actions were chosen in each broad category of point, diffuse urban and diffuse rural sources. The actions selected were assumed to be broadly representative of the wider range of actions available within each category. Secondly, the impacts for only one level of each action were modelled. Thirdly, impacts have only been assessed in terms of three indicators of water quality: suspended sediments (SS), phosphorus (P) and nitrogen (N). This has the potential of understating impacts because it excludes impacts of pathogens, toxicants, acid sulphate soils and other issues from the analysis.

Levels of SS, P and N under a range of intervention strategies were predicted. Table 1 presents these modelled loads for the project area for the *base case, no intervention* and *intervention* scenarios.

To identify the most cost-effective strategy from a range of load reducing best management strategies, annual net changes need to be compared for the *no intervention* and *intervention* scenarios. The basis for this comparison was the annual difference between total suspended sediments (TSS), total phosphorus (TP) and total nitrogen (TN) loads for the two scenarios starting in 2004 and running through to 2026. In the *Intervention* scenario, TSS loads are predicted to fall below current levels by 90 000 tonnes per year (t/yr) to 150 000 t/yr. TN levels will have decreased by 700 t/yr below current levels to 3100 t/yr and TP by 820 t/yr to 540 t/yr (Table 1). These are the key water quality outcomes of the intervention strategies. The next step is to identify costs of alternative strategies to achieve these load reduction outcomes.

Mitigation strategies and estimates of costs

A number of intervention strategies have been identified to improve water quality levels in the case study area.

Table 2 **Rural diffuse intervention expenses**

Rural diffuse strategy	Kilometres of riparian strip	Total cost of establishing riparian strips ($M)	$M/yr for the next 20 years
Grassed riparian strip	5000	28.00	1.40
Riparian rehabilitation strip	2700	67.50	3.38
Total	7700	95.50	4.78

Prior and continuing scientific work, programs and policies being implemented by state and local government, industry and the community formed the rationale for the selection of mitigation strategies considered in this study. These include *Moreton Bay Catchment Water Quality Management Strategy 1998, South East Queensland Water Quality Management Strategy 2001* and *Environmental Protection (Water) Policy 1997*. These best practice scenarios were estimated by WBM Oceanics Australia (2004) (reported in Rolfe et al. 2005).

The broad areas where these may occur include:

- *diffuse sources in rural areas:* reducing sediment and nutrient movement off agricultural lands and down waterways
- *diffuse sources in urban areas:* including improvements to urban diffuse waste and greenfields development sites
- *point source facilities:* including improvements to sewerage treatment plants, industrial facilities and intensive agriculture sites.

In some cases the intervention strategies have already commenced and program costs committed by different public agencies and communities. In other cases a sample of representative programs has been approximately costed to provide a guide to potential intervention commitments. The broad types of programs that have been assessed are:

- waste water and sewerage treatment plant upgrades
- retrofitting urban facilities to reduce urban diffuse emissions
- establishing riparian grass buffers in partnership with landholders on rural lands
- rehabilitating riparian zones on selected major streams.

The cost estimates used in this analysis relate only to the additional costs of implementing the best practice management actions outlined under the *intervention* scenario. They do not include expenses outlined in the *no intervention* scenario or expenses associated with implementing best practice water quality management strategies for greenfield urban developments.

The cost estimates for these three types of programs are summarised below:

Rural diffuse mitigation expenses: The total length of first and second order streams in the SEQ region is approximately 5000 kilometres. The cost of establishing grassed riparian filter/buffer strips along the stream was estimated at $5600 per kilometre to cover capital expenses (e.g. fencing and off stream watering points) and annual maintenance costs[2].

Based on consideration of the SEQ Regional Water Quality Management Strategy's scientific results and characteristics of the region's various stream orders, it was decided that in addition to grassed riparian strips, riparian rehabilitation strips in SEQ should be established in half of the region's second order streams, all third order streams and half the fourth order streams (EPA 2004a). The total length of second, third and fourth order streams requiring riparian rehabilitation strips in the project area is 2700 kilometres (EPA 2004a). Riparian rehabilitation strips are considerably more expensive to construct than grassed riparian filter strips. An estimate of $25 000 per kilometre to cover establishment and 12 months maintenance costs was used in this study. These cost estimates do not include any allowance for opportunity costs (e.g. production losses), and therefore may understate full opportunity costs. Table 2 provides a summary of the kilometres of riparian and riparian rehabilitation strips included in the *intervention* case for the SEQ project area and the estimated cost (dollars per kilometre) of each.

Urban retrofit expenses: Urban retrofit expenses relate to investments in a number of structural and non-structural urban diffuse management actions in existing urban areas. These actions include increased compliance monitoring, education and awareness programs, construction of stormwater management devices (e.g. gross pollutant traps, sediment traps and mini-wetlands), and increased incidence of street sweeping and riparian vegetation protection in urban areas. The EPA (2004a)

2. This cost estimate is based on WBM Oceanics Australia (2004) and is adjusted following EPA (2004a).

Table 3 **Present value of intervention case costs**

Items	Costs ($M)
Waste water treatment plants	242.90
Riparian grassed	28.00
Riparian rehabilitation	67.50
Retrofit urban facilities	212.77
Total present value of annual expenses	551.17

estimated that in the SEQ region approximately $8 million per annum was spent on urban retrofit actions.

In the *intervention* scenario, a $580 million expenditure program over 20 years was necessary to effectively retrofit a combination of best practice water quality measures to existing urban and rural residential land in the SEQ region (WBM Oceanics Australia 2004). However, according to WBM Oceanics Australia (2004), approximately 40 per cent of this expense will occur via natural attrition as old plants are replaced with new and more efficient plants, and future redevelopment projects incorporate best practice water objectives as a requirement of their development approval. With this in mind, the anticipated additional urban retrofit costs associated with the introduction of best practice water quality management objectives in the SEQ region was estimated to be $350 million over 20 years or $17.5 million per year.

Point source expenses: Point source polluter expenses fall into two categories – intervention expenses associated with upgrades to existing sewerage treatment plants, and intervention strategies to reduce the quantity of point source pollution entering waterways via major industrial and aquaculture discharges in the SEQ project area. Information from the five-year forward estimates on submissions from local government, under the *Local Governing Bodies Capital Works Subsidy Scheme* for water and waste sewage infrastructure (40 per cent state government subsidy for eligible works) and sewage effluent re-use infrastructure (50 per cent state government subsidy for eligible works), have been used as the cost estimates for modelling cases. It should be noted that these costs do not differentiate between sewage plant upgrades to service population increases, or to achieve best practice environmental management.

The five-year forward estimate for planned works under the above scheme in 2005 was $544 million for SEQ (Rolfe et al. 2005). However a portion of these funds will be required to service population increases independent of the environmental assets and water quality objectives assessed here. Because it was not possible to differentiate between planned expenditure on sewage treatment upgrades to service population increases from expenditure on best management practices to reduce sewage nutrient emissions to waterways and water reuse strategies, a 50:50 split was assumed, based on the *Local Governing Bodies Capital Works Subsidy Scheme* for water and waste sewage infrastructure and sewage effluent re-use infrastructure. In the study, $272 million was assumed to be allocated to sewerage treatment plant upgrades to deal with anticipated population growth and $272 million was assumed to be allocated to best management practices to reduce sewage nutrient emissions to SEQ waterways.

In estimating both the pollutant load reductions and cost, other point source emissions regulated under the *Environmental Protection Act 1997* have been excluded as their proportional contribution to nutrient emissions was small on a regional scale and relatively few activities are involved. There are several thousand licensed industrial emitters in SEQ, and higher water quality standards may impact on some of these emitters, although existing licence conditions are expected to be maintained in the short term. The majority of costs of reducing industrial emissions are expected to be private costs, which will vary widely between sites and industry types. In this study estimates of those costs have not been assessed because:

- Modelled reductions in industrial emissions were a relatively low proportion of overall emissions.
- It was difficult to gain estimates of private costs.
- Costs were expected to vary according to which mechanism might be modelled for reducing industrial emissions.

The total costs for intervention strategies in 2004 dollars are summarised in Table 3.

The *intervention* scenario can be achieved through a number of actions targeting rural diffuse sources, urban diffuse sources and urban point sources. The total cost of these various actions in the period to 2026 was assessed at $551.17 million (in 2004 dollars). This translates to an annual cost of $23.96 million per year over the period. These amounts do not include potential private costs of industry and agriculture of reducing emissions further, or the private costs for greenfield urban developers. Furthermore, these estimates need to be treated with some caution, because:

(a) Data on waste water treatment plants have been estimated from planned expenditure by local government, with a 50 per cent apportionment for improving water

Table 4 **Cost-effectiveness of point and diffuse source load intervention strategies**

Pollution load	Average annual point source load reduction (t/yr)	Average annual cost of point source load reduction ($/t/yr)	Average annual diffuse load reduction (t/yr)	Average annual cost of diffuse source load reduction ($/t/yr)
TSS	NA	NA	86 948	54
TN	820	6 729	546	8 553
TP	1 022	5 400	118	39 735

quality (these cost estimates may be subject to change by local authorities).

(b) Estimates of riparian protection and rehabilitation costs may underestimate some opportunity costs to landholders, and hence may understate the costs if voluntary, wide-spread adoption is to be achieved.

(c) Estimates of private costs arising from impacts on point source emitters or urban greenfields development have not been included.

Cost-effectiveness analysis of mitigation strategies

A basic CEA has been performed to identify where strategies may be best targeted for water quality improvement in the study area. The results of the CEA presented in Table 4 are expressed in terms of cost per tonne of pollution load reduction. The broad focus of the analysis was to identify the effectiveness of load-reducing strategies. These measures should be comparable across various scenarios, and capable of capturing the impact of different interventions with different effects. All monetary values were expressed in 2004 dollars, and were annualized into present values using an inflation-adjusted discount rate of 6 percent.

Table 4 presents the results of the comparison of cost-effectiveness of pollutant control measures from point and diffuse sources respectively. Reducing sediment loads through diffuse management actions (i.e. riparian grassed filter strips) may be cost effective at $54 per tonne, in addition to reducing the associated nitrogen and phosphorus loads. However, previous work indicates that point source SS loads are negligible compared to diffuse source SS loads and were not included in the modeling (Rolfe et al. 2005). Therefore, a comparison between point and diffuse sources was not possible. A significant amount of TN can be removed through both point source and diffuse strategies. However, the point source strategies to reduce TN are cheaper ($1804 per tonne per year) to implement than diffuse improvements. Similarly a significant amount of TP can be removed as a result of both the diffuse and point source strategies. In this case however, per unit reductions in TP are significantly cheaper ($34 335 per year) to achieve through investment in point source reduction strategies than diffuse mitigation strategies.

The analysis indicates that the cost for reducing the load of N from point source is slightly lower than the cost for corresponding decreases from non-point sources. In contrast, the costs of reducing P are much higher from non-point sources than point sources, with the cost of P reductions from diffuse sources more than seven times the cost from point sources. As a whole, the CEA shows which mitigation strategies have the lowest average cost of reduction.

Issues of uncertainties

For a number of reasons, costs and effects of a mitigation strategy are seldom known with certainty. Some major caveats should be noted for the analysis undertaken in this study. They relate to:

- limited data availability, particularly limited scientific data linking mitigation strategies to expected benefits of water quality improvements (Alam et al. 2006)
- the variability of non-point emissions and their lack of observability (Braden & Segerson 1993; Shortle et al. 1998)
- uncertainty regarding the scope of benefits resulting from the modeled interventions
- uncertainty regarding the scope of costs to be included in the analysis
- the very broad scale of the modeling undertaken
- limitations about the type of economic analysis undertaken.

Some of these uncertainties such as those arising from modeled load reductions and cost estimation of intervention-related parameters can be addressed by sensitivity analysis, identifying those parameters to which the decision is sensitive, and determining how it would change if the parameters changed. As this study is based on the secondary sources of data, it was not possible to conduct a sensitivity analysis to validate the cost estimates or modeled outcomes.

Although it is likely that the transaction costs associated with the implementation of cost-effective measures can be relatively high, this study did not consider this issue in cost estimation. Similarly, this study did not take into account any non-market or flow-on effects in terms of cost estimation. Further studies are required to address these issues.

Conclusion

The aim of this study was to assess the cost-effectiveness of different water quality mitigation strategies in SEQ. The CEA conducted for this study demonstrates the value of the technique in informing policy makers about the choice of alternative mitigation actions for water quality improvement. To achieve water quality objectives, pollution loads can be reduced by implementing less costly (i.e. more cost-effective) strategies. Our analysis suggests there is substantial potential for cost savings by targeting intervention strategies in SEQ. The analyses provide some indication of the most cost-effective reduction strategies for TSS, TN and TP in the region. It appears to be more cost-effective to reduce TSS from diffuse sources, and to reduce TN and TP within point source loads.

These are general findings, and there will need to be some sensitivity to individual sites/catchments where variations in loads and appropriate intervention strategies can be expected. At the more localised case study level, it is likely that different mixes of intervention strategies for both diffuse and point sources will be optimal to meet desired targets.

It should be noted that the CEA results do not allow a clear conclusion to be drawn about the overall desirability of water quality improvement programs. To achieve that outcome, estimates of the public benefits of water quality improvements would need to be compared to the costs. Nevertheless, from the policy decision-making perspective, the CEA of competing alternatives can be used to determine which specific mitigation strategies should be funded over others. A more detailed cost benefit analysis would be needed to assess the net benefits of various *intervention* strategies at individual catchments. As well, impact assessment studies might be needed to identify the groups in society that bear any economic or social impacts of different mitigation strategies. This means that the overall desirability of cost-effective solutions should be evaluated on a case by case basis. Nevertheless, in many ways, the SEQ catchment is typical of urbanized and industrialized areas in the nation. Hence, the case study results should have broader implications in terms of determining the cost-effective mitigation strategies for on-going national efforts to improve water quality.

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