The Cost of Pollution: Supporting Cost-Effective Options Evaluation and Pollution Reduction

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The Urban Water Security Research Alliance (UWSRA) is a $50 million partnership over five years between the Queensland Government, CSIRO's Water for a Healthy Country Flagship, Griffith University and The University of Queensland. The Alliance has been formed to address South East Queensland's emerging urban water issues with a focus on water security and recycling. The program will bring new research capacity to South East Queensland tailored to tackling existing and anticipated future issues to inform the implementation of the Water Strategy.

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FOREWORD

Water is fundamental to our quality of life, to economic growth and to the environment. With its booming economy and growing population, Australia's South East Queensland (SEQ) region faces increasing pressure on its water resources. These pressures are compounded by the impact of climate variability and accelerating climate change.

The Urban Water Security Research Alliance, through targeted, multidisciplinary research initiatives, has been formed to address the region's emerging urban water issues.

As the largest regionally focused urban water research program in Australia, the Alliance is focused on water security and recycling, but will align research where appropriate with other water research programs such as those of other SEQ water agencies, CSIRO's Water for a Healthy Country National Research Flagship, Water Quality Research Australia, eWater CRC and the Water Services Association of Australia (WSAA).

The Alliance is a partnership between the Queensland Government, CSIRO's Water for a Healthy Country National Research Flagship, The University of Queensland and Griffith University. It brings new research capacity to SEQ, tailored to tackling existing and anticipated future risks, assumptions and uncertainties facing water supply strategy. It is a $50 million partnership over five years.

Alliance research is examining fundamental issues necessary to deliver the region's water needs, including:

- ensuring the reliability and safety of recycled water systems.
- advising on infrastructure and technology for the recycling of wastewater and stormwater.
- building scientific knowledge into the management of health and safety risks in the water supply system.
- increasing community confidence in the future of water supply.

This report is part of a series summarising the output from the Urban Water Security Research Alliance. All reports and additional information about the Alliance can be found at http://www.urbanwateralliance.org.au/about.html.

Chris Davis
Chair, Urban Water Security Research Alliance

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EXECUTIVE SUMMARY

Over the past decade, the water quality in the rivers and bays in South East Queensland (SEQ) has deteriorated. Legislation, such as the *Environmental Protection (Water) Policy (2009)*, specifies Water Quality Objectives to maintain environmental values of SEQ waterways. At a minimum, pollution reduction will be required to maintain current conditions as pressures increase from population growth. Further reduction will be required to improve the condition of the ecosystems.

Cost-effectiveness analysis was used to identify the cheapest abatement options for urban water pollution. This can lead to large cost savings to meet water quality targets and can be used with:

- marginal benefit abatement curves to define the 'optimum' level of pollution abatement and to define costs as part of a Cost Benefit Analysis.
- load reduction targets to support policy responses such as market based instruments as well as indicating a price on pollution to reach a reduction target. The US EPA Water Quality Trading Policy Cost noted cost savings of $900 million per year by adopting a flexible program that focused on cost differentials of pollution abatement.
- cost-effectiveness analysis in Total Water Cycle Management Plans to capture pollution related externalities.

Marginal Abatement Cost Curves were developed for Total Phosphorus, Total Nitrogen and Total Suspended Solids. Capital costs, operating costs and the pollutant removal effectiveness were documented separately to facilitate review and updating of pollution abatement estimates. This also identified drivers of cost-effectiveness as well as sources of uncertainty. The combination of poor data quality and an important cost driver highlighted areas for further research. The sensitivity of the results was also considered for various discount rates.

Figures 1, 2 and 3 provide a summary of the Marginal Abatement Cost curves and the cost-effectiveness data reviewed in this study (note the log scale). A detailed breakdown of each abatement option is provided in the report. The range in cost-effectiveness for each abatement measure indicates the variation from economies of scale and pollution abatement effectiveness. This variation may account for the conflicting comparison of cost-effectiveness for abatement options in the literature and is likely to be much greater than the uncertainty in the data. Consequently, an order of magnitude difference in abatement cost-effectiveness as shown on the following log chart is likely to be significant.

The application of this data to a particular location requires a review of cost allocations, the assumed value of water, environmental equivalences for pollutants and assumptions for the discount rate and period of analysis. Understanding the load profile of a catchment is also important to select relevant abatement measures.

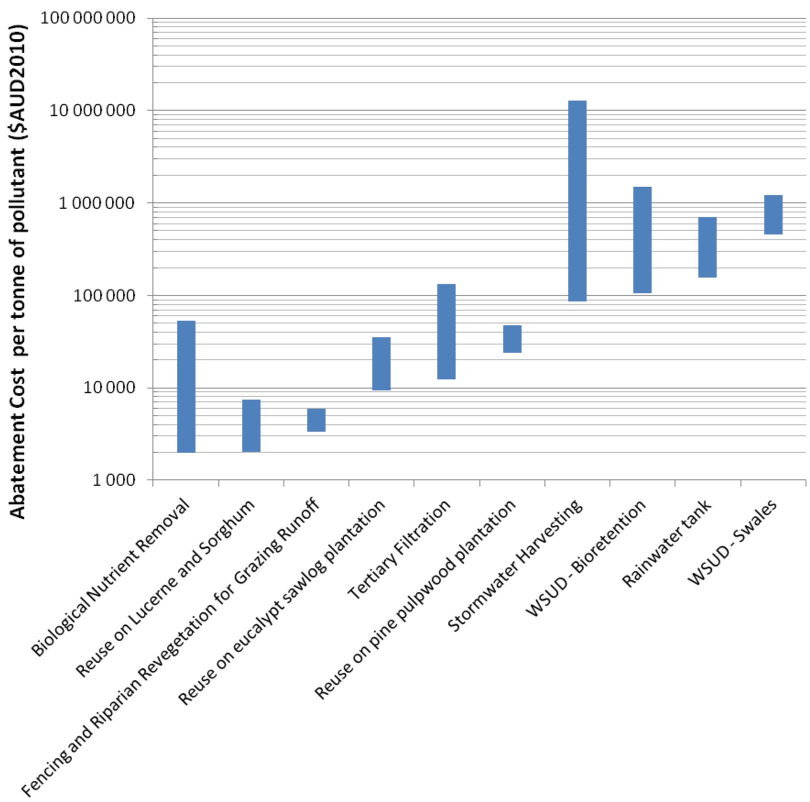

Figure 1: Review of high and low cost estimates for Total Nitrogen abatement (with a 3% discount rate and value of water based upon the bulk water price path).

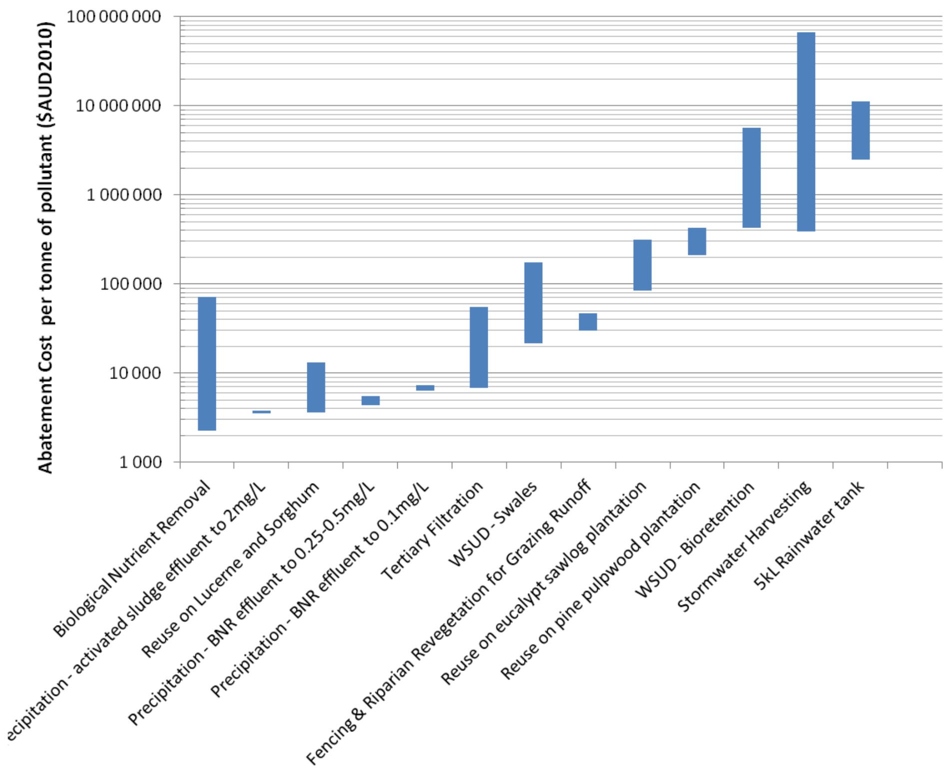

Figure 2: Review of high and low cost estimates for Total Phosphorus abatement (with a 3% discount rate and value of water based upon the bulk water price path).

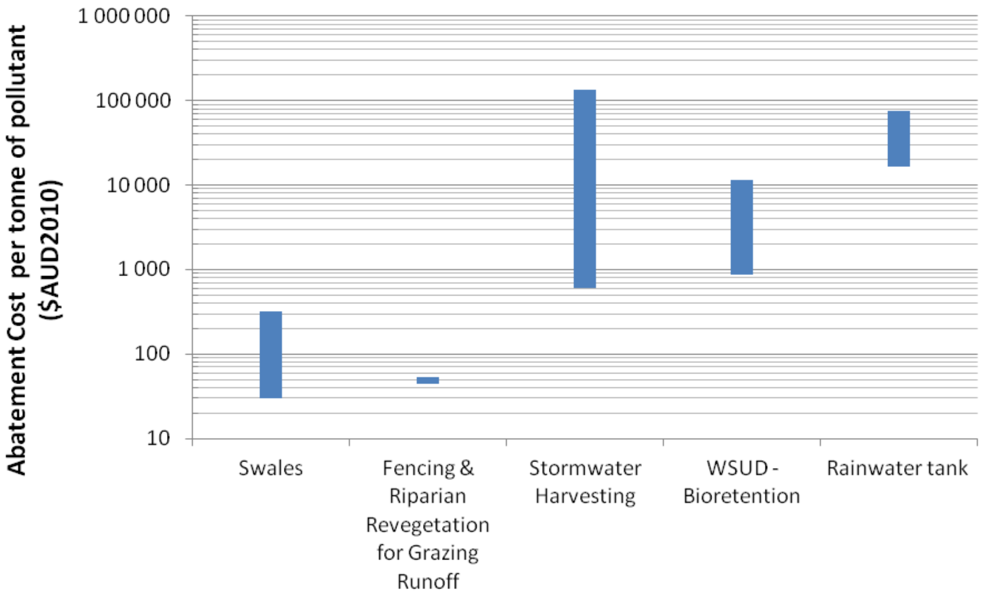

Figure 3: Review of high and low cost estimates for Total Suspended Solids abatement (with a 3% discount rate and value of water based upon the bulk water price path).

The difference between the least and most cost-effective abatement options was large and suggests the potential for large cost savings. In general, point source abatement and riparian revegetation provided the most cost-effective abatement measures for nutrients. Riparian revegetation and swales provided the most cost-effective sediment abatement. Rainwater tanks, bioretention and stormwater harvesting provided the least cost-effective nutrient and sediment abatement for the systems reviewed.

Large wastewater treatment plants were very cost-effective for nutrient removal compared to diffuse urban pollution abatement using Water Sensitive Urban Design. The cost for point source pollution abatement increased for smaller plants as well as for lower pollutant concentrations for subsequent pollution abatement. An economy of scale was less apparent for abatement measures with high operating costs such as phosphorus precipitation.

Reuse of wastewater effluent for forestry and agriculture also provided cost-effective point source abatement but may be limited by the large areas of land required and the distance from treatment plants.

Advanced Wastewater Treatment Plants (AWTP) can potentially provide nutrient abatement at minimal additional cost. However, further research is required to verify net reductions in nutrients and to confirm assumptions about the value of water produced.

Swales and riparian revegetation provided cost-effective sediment abatement. The abatement costs of both systems were sensitive to the amount of application and choice of location in the catchment. Other Water Sensitive Urban Design abatement measures such as stormwater harvesting, bioretention and rainwater tanks were less cost-effective and were sensitive to the value of water produced.

The ranking of cost-effectiveness of abatement options was not sensitive to the discount rate. Although a change in discount rate from 3% to 5.5% produced percentage changes of up to 50% for the cost-effectiveness of some abatement options, there was only minor change to the ranking of options by cost-effectiveness.

1. INTRODUCTION

Over the past decade, the condition of aquatic ecosystems in South East Queensland (SEQ) has deteriorated (EHMP, 2009). The *Environmental Protection (Water) Policy (2009)* and State *Planning Policy 4/10 Healthy Waters (2010)* legislate Water Quality Objectives to achieve the Environmental Values of waterways in SEQ. Pollution reduction will be required to stabilise the condition of aquatic ecosystems as the population grows over the coming decades. Further reduction in pollution will be required to restore the health of the waterways.

This raises important questions for policy makers and the water sector as a whole. What amount of pollution reduction is required, how can it be achieved, at what cost, and, can these costs be included in water planning for a growing population?

There have been a number of estimates of the cost and focus of pollution abatement to meet Water Quality Objectives and Environmental Values in SEQ. Some estimates focus on wastewater treatment plant upgrades (Driml, O'Sullivan *et al.*, 2005) p5 while others focus on diffuse sources (SEQHWP, 2007). Indeed, the estimates of cost-effectiveness varied significantly between studies and led to differing conclusions regarding the cost-effectiveness of point and diffuse pollution mitigation (BDAGroup, 2005; Rolfe, Donaghy *et al.*, 2005; BDAGroup, 2007; Alam, Rolfe *et al.*, 2008; EPA, 2008; WaterByDesign, 2010; WaterByDesign, 2010). The large range in costs and differing conclusions for cost-effectiveness was also noted in other regions in Australia (BDAGroup, 2005). This suggests the need for a consistent method and review of the factors which affect pollution mitigation cost-effectiveness. This is particularly important given the recent expenditure for water quality and the continued decline of a number of waterways. In SEQ, approximately $400 million has recently been spent on wastewater treatment plant upgrades, $60 million for stormwater quality improvements and $18 million for waterway management and restoration (Bunn, Abal *et al.*, 2008). A review of pollution abatement cost-effectiveness can inform targets and policy for the future direction for pollution control. This can also form a cost benefit approach by drawing upon existing estimates of the benefit of maintaining and improving water quality in SEQ (Udy, Bartkow *et al.*, 2001; Rolfe, Donaghy *et al.*, 2005; Binney, 2010).

The objective of cost-effectiveness is also important for water planning (QueenslandGovernment; QCA, 2010). The Queensland Government has current and committed projects of $9 billion for urban water in SEQ (QWC, 2008). Total Water Cycle Management Plans (TWCMP)1 are also being prepared to address future demands. This presents an opportunity to address the cost-effectiveness of *both* water supply and water quality objectives. This is also recommended by existing TWCMP Guidelines (Hurikino, Lutton *et al.*, 2010; WaterByDesign, 2010) as well as a number of national water policy reviews (DEWHA, 2010) (Fane, Blackburn *et al.*, 2010) – p12. This report aims to support the integration of water quantity and quality objectives by:

- Reviewing existing data for pollution abatement cost and effectiveness and considerations required for application to South East Queensland; and
- Describing a method for calculating the cost-effectiveness of abatement measures, the use of the Marginal Abatement Cost Curves to estimate the abatement cost for a given pollution target and the application to options evaluation in Total Water Cycle Planning.

[1] Also prescribed by the Environmental Protection (Water) Policy 2009.

2. METHOD

2.1. Application of Marginal Abatement Cost Curves

2.1.1. Water Pollution Management

The following three steps illustrate the context and application of Marginal Abatement Cost Curves (MACC). This approach is similar to the National Academy of Science process for designing stormwater control measures (SCM) on a catchment (watershed) scale (NAS, 2009 – pp422-423). The method provides a strategic approach for meeting pollution reduction targets in a catchment. The approach does not consider the effect of individual management actions upon water quality indicators at particular locations in the catchment. It also assumes that load reduction targets are independent and include the effects of changes in one pollutant upon another.

1. Define pollution status. Consideration of current catchment ecosystem health, current pollutant loads, future pollutant loads and sustainable pollutant load targets.
2. Develop a Marginal Abatement Cost Curve. Define cost-effectiveness of available abatement options and potential load reduction. This can be graphed as a MACC.
3. Application and policy response. The MACC can be used with:
   (a) marginal benefit abatement curves to define the 'optimum' level of pollution abatement and to define costs as part of a Cost Benefit Analysis;
   (b) load reduction targets to support policy such as market based instruments and provide an indication of a price on pollution to reach a reduction target; and
   (c) cost-effectiveness analysis in Total Water Cycle Management Plans to capture pollution related externalities.

2.1.2. Marginal Abatement Benefit Curves, 'Optimum' Pollution and Cost Benefit Analysis

The value of pollution can be calculated from the cost and benefit of pollution abatement. Figure 4 illustrates the theoretical relationship of marginal costs and benefits for estimating the 'optimum' level of abatement. This report provides data for the MACC. Studies such as *Managing what matters: The cost of environmental decline in South East Queensland* provide data for the marginal benefits curve (Binney, 2010). The benefits were calculated for changes in environmental condition for meeting 'SEQ Natural Resource Management Targets' and a 'Do nothing more' scenario compared to the current condition. This approach may be extended to consider further improvements in environmental condition (Windle and Rolfe, 2006) and other externalities associated with particular water options (Daniels, Porter *et al*., 2010). For example, stormwater harvesting may provide additional benefits to environmental flows, flood protection and public amenity.

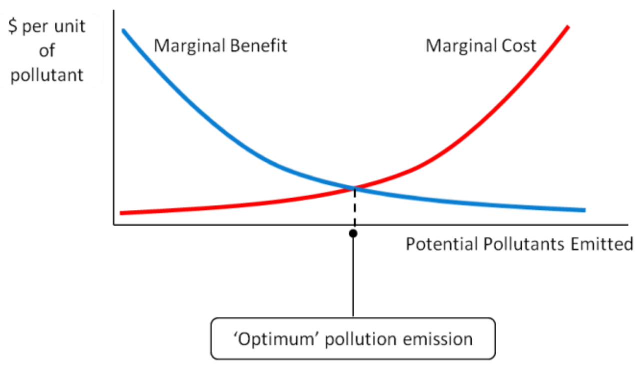

Figure 4: Marginal Cost and Marginal Benefit for Pollutant Abatement (Aldrich, 1996).

2.1.3. Pollution Reduction Targets and Pollution Value

Pollution reduction targets can be used with the MACC to calculate a weighted average value for pollution, illustrated by Figure 5. The Marginal Cost curve in Figure 4 (red curve) is approximated in Figure 5 by arranging abatement options from least to most costly. Each abatement option is represented by a bar with the height reflecting the cost per unit of pollution abatement and the width reflecting the amount of pollution abatement available for the option in a given catchment or region[2].

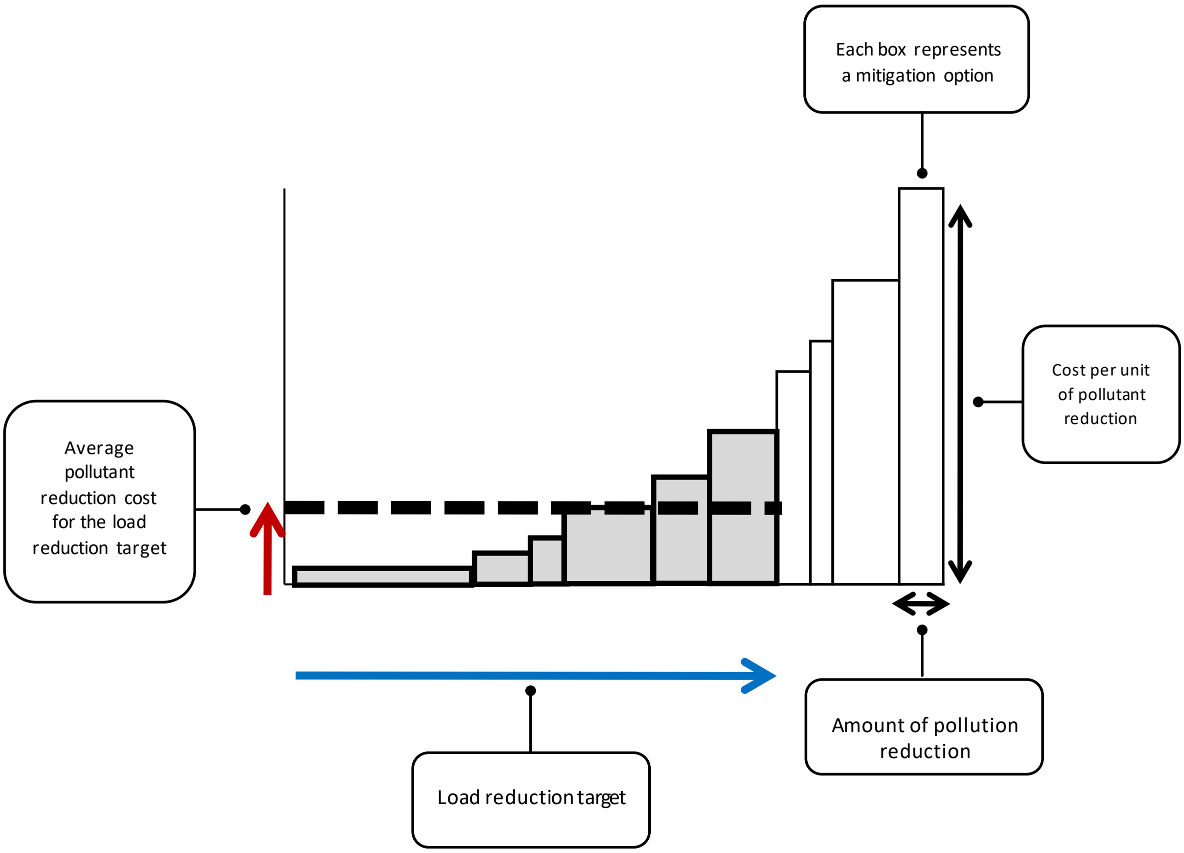

Figure 5: Pollutant Abatement Cost Curve and the Average Pollutant Cost to Meet a Load Reduction Target.

The reduction target can be defined by the 'optimum' level of pollution abatement from the benefits and costs of abatement. Alternatively, the reduction target can be based upon legislation or other targets for waterway health. In this case, the level of pollution abatement is defined by policy and is not necessarily the 'optimum' from a cost-benefit perspective. However, the definition of an externality depends on the level of impact that is 'tolerated' and can change over time (Leontief, 1986 – pp249-252). The concept of a 'duty of care' expressed in policy can be used to reflect the 'tolerated' impacts and can be a proxy 'optimum' level of pollution based upon a political 'willingness to pay' (Bowers and Young, 2000). This approach has the flexibility to cost various policy objectives and can inform Market Based Instruments. An example of this approach is the use of aquatic Ecosystem Values (EV) and Water Quality Objectives (WQO) to define the maximum concentration of total nitrogen and total phosphorus[3] (Driml, O'Sullivan *et al.*, 2005). Catchment modelling can then determine load reductions required to maintain or achieve the Water Quality Objectives as a population grows. The cost of abatement can be calculated for achieving the policy objective and can inform policy responses such as pollution trading or offsets.

[2] Some caution is required to avoid confusion with the description of cost-effectiveness as opposed to cost. For example an abatement option that is the *least* cost-effective can also be described as have the *greatest* costs per unit of abatement.
[3] For example, see example WQO to maintain EV at the following Department of Environment and Resource Management (DERM) website http://www.derm.qld.gov.au/environmental_management/water/environmental_values_environmental_protection_water_policy/what_are_evs__wqos.html

The use of MACC to calculate a value for pollution was popularised for greenhouse gas emissions by the McKinsey Cost Curve (McKinsey & Co, 2008). The McKinsey Cost Curve was used to calculate a weighted average price for pollution in response to various pollution reduction targets. MACC have also been developed to identify the least cost options for a potential nutrient trading in SEQ (BDA Group, 2005 – p87) as well as trading or offset schemes implemented in NSW (James, 1994). The US EPA Water Quality Trading Policy Statement notes that cost differentials for pollution control within a catchment offer greater flexibility and cost savings for meeting water quality objectives (USEPA, 2003). Savings were estimated as $900 million per year for meeting water quality goals in the US compared to the least flexible approach. The US EPA Water Quality Trading Policy supports trading in Total Nitrogen, Total Phosphorus and sediment loads (USEPA, 2003). In SEQ, the use of a Market Based Instruments has also recently been considered for managing sediments (DERM, 2010).

In general, MACC supports policy responses that are prudent (needed) and efficient (cost-effective) as recommended by the Queensland Competition Authority (QCA, 2010). It also meets the following preferences for policy intervention from a survey of households with respect to meeting natural resource management targets in SEQ (Binney, 2010 - pp55-56):

- spending local rates elsewhere in the region where it is more efficient;
- paying for ecosystem services (such as payments to farmers) where it is more efficient;
- preventative action to reduce environmental decline rather than rehabilitate later; and
- offsets for negative impacts of all future housing in SEQ.

For further discussion of using costs to value pollution, refer to Appendix 1.

2.1.4. Cost-Effectiveness Analysis

Cost-effectiveness Analysis (CEA) is an established economic method for evaluating the cost of an option to reach a defined target (Pearce, Atkinson *et al.*, 2006). Application of cost-effectiveness analysis has also recently been reviewed for water quality interventions in SEQ (Alam, Rolfe *et al.*, 2008). Cost-effectiveness analysis can be used for both pollution abatement options as well as for water supply options. In this case, pollution abatement costs were reviewed to extend the cost-effectiveness analysis of water supply options. This method was noted as being suitable for capturing sustainability issues of sub regional total water cycle planning, such as water supply conservation and water supply augmentation (Hurikino, Lutton *et al.*, 2010 – p12; Fane, Blackburn *et al.*, 2010). This method also supports National Water Initiative pricing principles for including full cost recovery, including recovery of environmental externalities (DEWHA, 2010). This study considers pollution mitigation costs from greenhouse gases, nutrients and sediments to extend the cost-effectiveness analysis. Equation 1 expresses the cost-effectiveness calculation including pollutant costs. The value of pollution (W_j) is considered further in the following section.

$$\mathbf{Y = C_p + O_p + \sum_{j}^{m} P_j . W_j} \qquad \text{Equation 1}$$

Where:

Y = extended cost-effectiveness

C_p = capital cost of a project

O_p = operating cost of a project over the period of analysis

P_j = pollution emitted by the project

W_j = value of pollution for a defined pollution reduction target

j = first pollutant considered

m = last pollutant considered.

2.1.5. Pollution Abatement Costs

Pollution abatement costs were calculated as the net present value of capital and operating costs divided by the pollution abated over the period of analysis, expressed in Equation 2. Note that the pollution abatement costs have the units of dollars in present value per unit mass of pollution abated. A number of other studies express the abatement cost as a *rate* using the annual reduction in pollutants and the present value over the period of analysis (and have the units of present value per unit of mass of pollution abated *per year*) (BDA Group, 2005; Rolfe, Donaghy *et al.*, 2005; BDA Group, 2007; Alam, Rolfe *et al.*, 2008). The difference in the two ways of expressing the abatement costs will be a factor equal to the period of analysis. For example, the rate pollution abatement cost will appear 20 times greater if the period of analysis is 20 years.

$$\mathbf{X} = \frac{C + \sum_{i=1}^{n} A_i / (\mathbf{1} + r)^i}{\sum_{i=1}^{n} P} \qquad \text{Equation 2}$$

Where:

X = abatement cost per unit of pollution

C= capital cost pollution abatement measure

A= annual operating costs of abatement measure

R = discount rate

P = annual amount of pollution abated

i = year

n= period of analysis.

2.1.6. The Discount Rate and the Period of Analysis

A discount[4] rate of 3% was assumed for a 20-year period of analysis, with a base date of 2010 for net present values. Historical inflation rates from the Reserve Bank of Australia were used to inflate cost estimates from literature to 2010 prices. The following discussion considers the effect of a discount rate for developing cost estimates of pollution abatement and provides background to the choice of discount rate and period of analysis. A real discount rate of 3% means that $100 spent in 20 years time has a present value of about $55. However, a real discount rate of 5.5% was also considered in the Sensitivity Analysis (IDC, 2003 - p28). The real interest rate of 5.5% captures the business risk on the time preference of money and is equivalent to an 8.5% nominal interest rate. This rate was used in a recent report for assessing the benefits of maintaining and improving the current environmental condition in SEQ (Binney, 2010).

The discount rate expresses how much *less* something is worth in the future compared to the present. This can raise a number of ethical and intergenerational issues (Ruth, 1993). In fact, an important principal of ecological sustainability is to ensure the future is not discounted for the present. Discounting has a large effect on expenditure that occurs over long time periods, such as operating costs for infrastructure. For example, if $100 is spent in 50 years time then a discount rate of 9% means that it has a present value of about $1.30. Figure 6 illustrates the effect of various discount rates and the period of analysis.

Table 1 presents some “approximate recommended” discount rates for cost benefit analysis which reflect the length of time of the analysis (Weitzman, 2001). This discount rate was based upon the rationale of gamma discounting which presents a sliding scale of discount rates depending on the time period considered. For example, a project with a time horizon of 26-75 years has a discount rate of

[4] The term discount rate refers to the inflation adjusted or real discount rate for application in Equation 1.

2%. The method was developed to address the uncertainty in discount rates in cost-benefit analysis (Weitzman, 2001). The discount rate can be the largest uncertainty in an economic analysis for environmental impacts over long time periods, as noted by Weitzman (2007 - p705) in his review of *Stern Review of the Economics of Climate Change*. In Australia, the *Garnaut Climate Change Review* used a discount rate in the range of '1.25 to 2.65 per cent in summing utility of income over the generations, and applying a higher rate, 4 per cent, in pricing emissions permits and allocating capital over the 21st century' (Garnaut, 2008 - p20).

Table 1: "Approximate Recommended" sliding scale real discount rates (Weitzman, 2001).

Time period	Name	Marginal discount rate (Percent)
Within years 1 to 5 hence	*Immediate* Future	4
Within years 6 to 25 hence	*Near* Future	3
Within years 26 to 75 hence	*Medium* Future	2
Within years 76 to 300 hence	*Distant* Future	1
Within years more than 300 hence	*Far-Distant* Future	0

In terms of the time preference for saving and investment, Garnaut noted that the long term inflation adjusted market return for government bonds in Australia and the US is 2.2% and 2.1% respectively (Garnaut, 2008 – p20). It was argued that Government sovereign debt rather than equity markets provides a more suitable rate because company specific risk is not relevant for time preference considerations of environmental analysis. This was also similar to the average interest rate of 2.3% for the US EPA Clean Water State Revolving Fund (CWSRF)[5] projects that aim to reduce pollutant impacts to waterways.

This is an important point given that the Australian Standard for Life Cycle Costing also notes that the discount rate for companies usually relates to the cost of loans and the returns required by the shareholders. Nonetheless, the Standard notes that Australian Governments usually set a real discount rate of 5 to 9% while many international studies use 3 or 4% (StandardsAustralia, 1999 – p24). Indeed, commercial interest rates may be applicable where returns are sought on borrowings (QCA, 2011 – p203) and appears common for many analyses (Standards Australia, 1999 – p24: DIP, 2008 – p37).

An analysis period of 20 years was adopted to provide a common period for considering a range of data sources and was chosen based on a number of studies for appraisal of capital works (NSW Treasury, 1999; BDA Group, 2005; DEC, 2006; BDA Group, 2007; BlighTanner, 2009), including the US EPA Clean Water State Revolving Fund (CWSRF)[6] which issued low interest loans for a period of up to 20 years. A time period of 20 years also corresponds to the planning period for the Regional Plan (QG, 2009).

Figure 6 shows the relationship between the discount rate and the period of analysis. For example, an amount spent in 20 years time has a present value of about one half, one third and a fifth of the original amount when discounted by 3%, 6% and 9% respectively. Consequently, if the discount rate is greater than 6%, then expanding the analysis period beyond 20 years will have a relatively small effect on the net present value. Figure 6 also illustrates that by reducing the discount rate following the 'approximate recommended' rates has the effect of increasing the analysis period by making future costs more significant to present values. For example, future cost are reduced by 50% in present value at time periods of about 24 years for a 3% discount rate, about 35 years for a 2% discount rate and about 70 years for a 1% discount rate. This is illustrated on Figure 6 by the points of intersection on the black dotted line.

[5] US EPA Accessed 20 July 2010 http://www.epa.gov/owm/cwfinance/cwsrf/.

However, a number of other studies use different time frames and discount rates. For example, Water by Design used a life cycle of 25 years for its business case for Water Sensitive Urban Design (Water by Design, 2009) and a real discount rate of 5.5% and cited guidance from Queensland Treasury. An evaluation of reuse of purified recycled water in SEQ used a discount rate of 7.5% and a time period of 25 years (CH2MHILL, 2008) while an evaluation of cost-effectiveness of rainwater tanks in Australia used a nominal discount rate of 8% (equivalent to about a 5% real discount rate) for levelised costs over a 50 year period (Marsden Jacob, 2007 - p23) based upon the standard home loan variable rate.

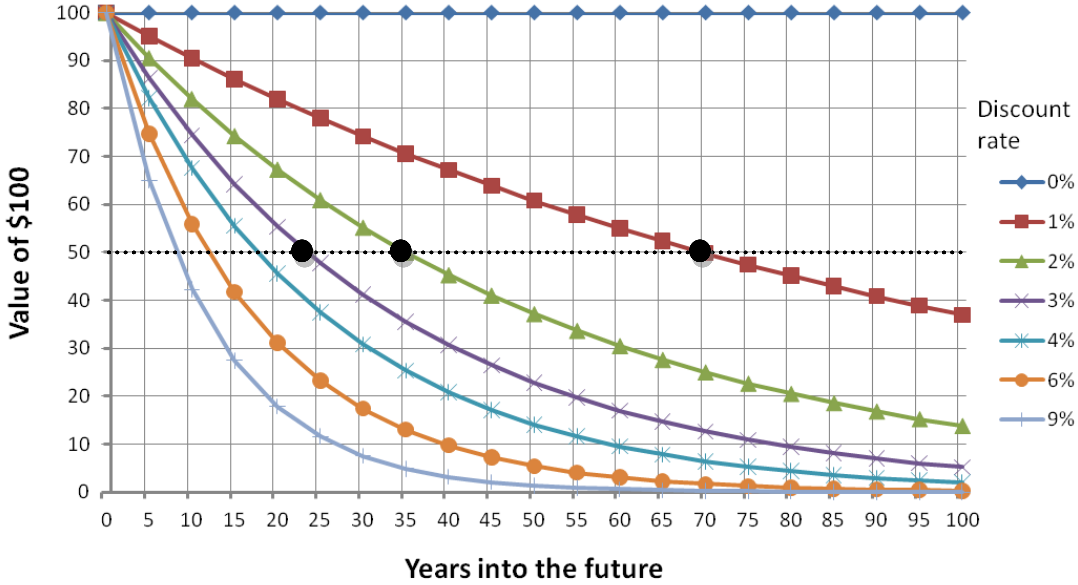

Figure 6: The effect of time and real discount rates on present value.

It was also noted that the length of analysis should consider the physical life of the longest lived component (Commonwealth of Australia, 2006). In cases where a shorter time period was used then the residual value of the asset needs to be considered. However, it was also noted that this can be impractical to estimate in many cases (DIP, 2008). For example, the residual value of a water treatment plant upgrade in 20 years time is difficult to estimate and may depend upon a range of other factors such as future water quality objectives and other treatment plant upgrades. The residual value may also be negative in some cases. For example, it was estimated that decommissioning could be up to 40% of capital costs for bioretention systems and up to 20% for other WSUD infrastructure (Water by Design, 2010 – p17). Decommissioning costs were included where the life span of the infrastructure was similar to the analysis period. However, the actual value of the infrastructure at the end of the analysis period was not included in the analysis.

2.1.7. Allocation and Environmental Equivalences

Some pollution abatement options provide a number of functions and require a procedure to allocate the costs of the system. For example Biological Nutrient Removal removes both phosphorus and nitrogen and a proportion of treatment costs need to be allocated to each pollutant. Stormwater harvesting not only reduces a number of pollutants but also provides a water supply. The allocation of costs for pollutants is outlined below and requires review before application to a particular location. In particular, the allocations are dependent on the characteristics of the receiving water in a particular context. For example, the relative importance of phosphorus and nitrogen removal can change depending on whether the effluent is disposed to inland waters, estuaries or the ocean and the limiting nutrient in the receiving water. Allocation procedures are also used in systems analysis such as Life Cycle Assessment (ISO, 2006b – p14) which notes the use of mass of outputs, chemical relationships, economic value or stakeholder values to allocate system resources and flows.

In addition, the effect of a pollutant upon the receiving environment (known as the equivalence ratio) may also change depending on the location. For example, a pollutant may have a bigger impact when released close to a sensitive estuary compared to release in the upper catchment. This is particularly important if the aim is to improve the receiving water quality for downstream environments such as Moreton Bay (ARUP, 2007). Environmental equivalences were not considered in this report due to the general nature of the review. However, some insight into environmental equivalences can be taken from a study for a nutrient trading scheme for the Logan-Albert catchment (ARUP, 2007). For example, considering the effect of the pollutants at the mouth of the Logan River, the reduction of one kilogram (kg) of Total Nitrogen (TN) from grazing was considered to be environmentally equivalent to the reduction of approximately 0.6 kg TN from urban areas (ARUP, 2007 - p25). It is important to note that the environmental equivalence is not necessarily the same as the trading ratios used in nutrient trading schemes. The trading ratio can be used to reflect uncertainty in achieving a reduction target (BDA Group, 2005; Nguyen, Woodward *et al.*, 2006) rather than the environmental equivalence.

Biological Nutrient Removal

The allocation for phosphorus and nitrogen from biological nutrient removal was based upon cost drivers reported by Australian utilities (Pickering and Marsden, 2007 – p41). The cost drivers reported for a biological nutrient removal plant was 75% of the treatment costs allocated to nitrogen removal and 25% allocated to phosphorus removal. This was similar to a cost allocation used by the USEPA which allocated approximately a quarter of the costs to phosphorus compared to nitrogen (USEPA, 2008 – pp3-37). The USEPA examples also allocated a proportion of costs to Biochemical Oxygen Demand for plants where this was an additional role provided in the upgrade. The cost allocation may require review if the receiving environment is more sensitive to phosphorus than nitrogen.

Diffuse Pollutant Reduction: Water Sensitive Urban Design (WSUD) and Rural Riparian Revegetation and Fencing

Construction and operating costs for a WSUD device were allocated to the pollutants based upon the design objectives from *Draft Urban Stormwater Queensland Best Practice Environmental Management Guidelines 2009* (Guidelines) (DERM, 2009 – Table 2.1b). These objectives were used for the Water by Design case studies (Water by Design, 2010). The objectives are expressed in the Guidelines as ‘Minimum reductions in mean annual loads from unmitigated development (%)’. Table 2 shows relative importance of one abatement option compared to another based upon the percentage reductions sought for pollutants. For example, the percentage Total Suspended Solids (TSS) load reduction objective is approximately twice as big as the percentage load reduction for TN. As a result, approximately twice as much cost for constructing and operating the WSUD device was allocated to TSS compared to TN. However, when expressed in terms of the cost per load reduction, the allocation reflects the catchment loads because the load reduction objectives cancel. This allocation was also applied to rural diffuse pollution to allow comparison of cost-effectiveness results.

The ‘WSUD drivers’ require review and provide an initial cost allocation for WSUD devices and diffuse rural pollution abatement. For example, the drivers may need to be modified to consider objectives for sustainable load targets in particular catchments.

Table 2: Cost Allocation based upon WSUD Pollutant Reduction Targets.

SEQ Region	TSS	TP	TN	Total
Minimum reductions in mean annual loads from unmitigated development (%) (DERM 2009)	80	60	45	185
Minimum reduction expressed as WSUD driver *	0.43	0.32	0.24	1.00

*note these numbers have been rounded which makes the sum appear less than the total

Rainwater Tanks and Stormwater Harvesting

The allocation procedure for rainwater tanks and stormwater harvesting was the same as for WSUD with the additional consideration of the value of the water captured. In these cases, the value of the water was removed from the cost of the system to allocate the remaining costs to pollution abatement. It was assumed that a new supply such as stormwater harvesting would offset the need for bulk water from the State Government supply. The value of the water supply was estimated using the revised price path for bulk water tariffs from the Queensland Water Commission (Queensland Water Commission http://www.qwc.qld.gov.au/reform/bulkwaterprices.html). Table 3 presents bulk water price adjusted for inflation. However, note that these prices were not discounted and only extend to 2017-2018. It was assumed that bulk water prices for 2019 to 2030 would not increase in real terms (ie when inflation is taken into account) beyond the 2017-2018 price. This price path is conservative (a low estimate) compared to an extrapolation of the price path from 2010-11 to 2016-17. Figure 7 presents the assumed price path with no discount rate, a 3% discount rate and a 5.5% discount rate. The effect of the discount rate on pollution abatement options was considered in the sensitivity analysis.

Table 3: Queensland Water Commission Indicative Price Path for Bulk Water.

	Current Price ($AUD 2010/ML)	Revised Prices ($AUD 2010/ML)	Indicative Prices ($AUD2010/ML)					
Council	2010-11	2011-12	2012-13	2013-14	2014-15	2015-16	2016-17	2017-18
Brisbane	$1,517	$1,743	$1,958	$2,161	$2,353	$2,534	$2,705	$2,812

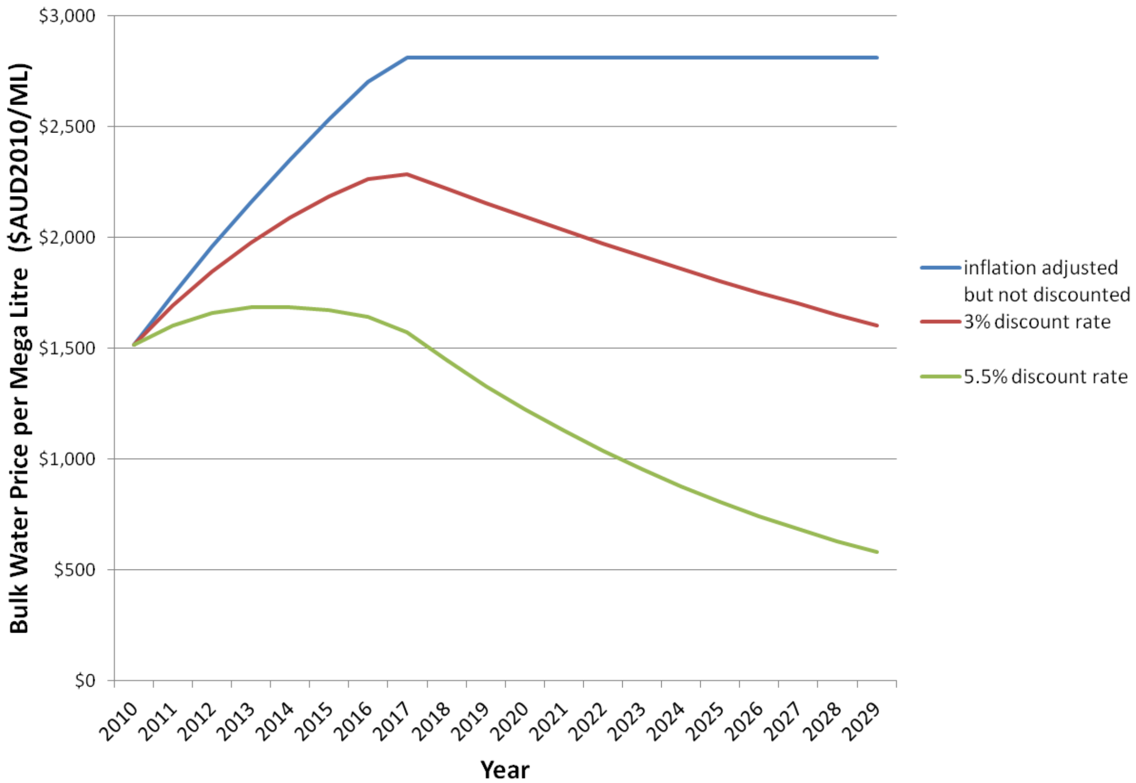

Figure 7: Assumed bulk water price path in present values.

Advanced Wastewater Treatment Plants

The allocation procedure for wastewater recycling used the drivers reported for Biological Nutrient removal as well as the value of water noted above.

2.2 Uncertainty

This report used literature which was often sparse and incomplete. In addition, the following summary of uncertainty was also limited and requires review. The review of uncertainty aimed to provide indicative ranges and highlight areas for further research. Uncertainty estimates may also inform 'trading ratios' for pollutants. For example, *A Guide to Market-Based Approaches to Water Quality* noted that a number of programs used a trading ratio of 2:1 for pollution reduction from non-point sources: point sources (BDA Group, 2005; Nguyen, Woodward *et al.*, 2006). The trading ratio reflected the higher uncertainty with non-point source reduction and aimed to ensure pollution reduction targets were achieved despite uncertainty with reduction estimates.

The uncertainty analysis focussed on uncertainty due to lack of knowledge (incertitude) rather than uncertainty due to variability. Uncertainty due to incertitude can be reduced by collecting more and better data. On the other hand, uncertainty due to variability is not reduced by collecting more data but is better understood and the estimate is more reliable (Burgman, 2005 – p26, 27). A mean and confidence interval were calculated where there was sufficient data. The International Stormwater Best Management Practices Database provides an example of reducing the variability through the collection of a large data set (WERF, 2010).

Uncertainty estimates were made for each variable in the cost-effectiveness calculation, namely capital costs, operating costs and the pollutant removal effectiveness. Relatively high operating costs can have a large affect on present values over the analysis period. The operating costs will also be sensitive to the discount rate and this was explored in the sensitivity analysis. The data review attempted to reduce uncertainty by identifying factors that affected the range of estimates for each variable. For example, the economy of scale of an abatement option can have a large affect on the cost-effectiveness of abatement. Similarly, the context of the abatement option may affect its performance. For example, particular locations and designs for riparian revegetation may be more effective than others for sediment abatement.

An initial assessment of uncertainty was made using data quality descriptors. Table 4 presents the data accuracy descriptors from the *GHG Protocol guidance on uncertainty assessment in GHG inventories and calculating statistical parameter uncertainty* (GHG Protocol, 2001). Table 5 presents a summary of the data accuracy descriptors applied to the capital, operating and effectiveness estimates in this report. More detail of the rationale for the data accuracy assessment is provided with the review of each abatement measure. The data accuracy assessment was based upon the information presented and was not a formal expert elicitation process.

Table 4: Data accuracy rating and corresponding intervals used in the GHG Protocol uncertainty tool.

Data Accuracy	Interval as Percent of Mean Value
High	± 5%
Good	± 15%
Fair	± 30%
Poor	± More than 30 %

Source: GHG Protocol (2001)

Table 5: Initial Assessment of Data Accuracy.

	Pollution Abatement	Capital	Operating	Pollutant Removal Effectiveness
Point Source	Biological nutrient removal	Fair	Poor	Good
	Tertiary filtration	Fair	Fair	Good
	Chemical precipitation	Fair	Fair	Good
	Advanced Wastewater Treatment	Fair	Fair	Poor
	Reuse for controlled irrigation	Fair	Fair	Fair
Diffuse	Stormwater harvesting	Poor	Poor	Fair
	Rainwater tanks	Good	Fair	Fair
	Water Sensitive Urban Design – Bioretention	Good	Poor	Fair
	Rural riparian revegetation and fencing	Fair	Poor	Poor

In general, capital and operating costs for point source data accuracy were considered to be ‘fair’. In most cases, SEQ specific data was available for the particular treatment processes. The data accuracy for pollutant removal effectiveness was also considered to be ‘good’ based upon the general engineering limitations of the process performance, process control in a treatment plants and the required monitoring of effluent concentrations to meet licence conditions. The exception was Advanced Wastewater Treatment Plants where the pollution abatement effectiveness was questionable due to the addition of ammonia to control biofouling at some plants. This may change over time as experience with the technology increases. Urban diffuse pollution control generally had ‘good’ data accuracy for capital costs. This reflects recent projects and reviews of unit rates for rainwater tanks, bioretention and vegetation planting for catchment control. However, data accuracy for operating costs was considered to be ‘poor’ due to lack of data for actual operating costs. The data accuracy for pollutant removal effectiveness was considered to be ‘fair’ but should be reviewed when monitored data is available for the actual performance of systems. Rural diffuse data accuracy was generally ‘fair’ for capital data but ‘poor’ for operating and pollutant removal effectiveness. However, strategic assessment of sediment abatement locations was considered which improves the applicability of the data for SEQ.

Finally, the combined uncertainty of the cost-effectiveness estimates as a result of the uncertainty in the capital, operating and effectiveness was not calculated. Not only is the data accuracy descriptor of ‘poor’ open-ended, but combined uncertainty calculations were beyond the purpose of the data descriptors in this report. Consequently, cost-effectiveness data was presented on a logarithmic chart to facilitate comparison in terms of ‘orders of magnitude’ rather than differences which may be due to uncertainty. Cost-effectiveness estimates that were an order of magnitude different are likely to be significant and provide a starting point for considering refinement and application of this data to a particular location.

2.2.1 Example Marginal Abatement Cost Curve

The following Marginal Abatement Cost Curve for greenhouse gas emissions was developed by McKinsey and Company (2008) and was updated in 2010 (Climate Works, 2010)[7]. The cost curve identifies cost savings, mostly from energy efficiency measures to buildings and transport as well as costs for abatement from agriculture, forestry and power sectors. From an economy wide perspective, the savings from energy efficiency can be used to balance the costs for a number of other abatement options. By analogy, similar cost saving measures may be available for other pollutants considered in this report, such as more efficient use of fertilizers in agriculture. The cost curve was used with

[7] The McKinsey cost curve was adopted because it presents costs over a time period that matches the period of analysis considered in this report, ie a period of twenty years.

pollution targets to calculate a price for carbon abatement. For example, using Figure 8, it was calculated that a 30% reduction in greenhouse gas emission below 1990 levels by 2020 would have a volume weighted average cost of approximately \$49/tonneCO_2e[8] (McKinsey & Co, 2008). In July 2011, the Commonwealth Government released its Clean Energy Future policy and set a price on carbon of \$23/tonneCO_2e, which passed through parliament in November 2011. The 2011 Clean Energy Future policy aims to reduce pollution by at least 5 per cent compared with 2000 levels by 2020, which will require cutting net expected pollution by at least 23 per cent in 2020. The long term goal of the proposed Clean Energy Future policy is to cut pollution to 80 per cent below 2000 levels, by 2050. The July 2012 set price for carbon will rise by 2.5% a year in real terms until July 2015 when the price will be set by the market and the number of permits capped by the government. If targets are increased, then there is likely to be an increase in the price of carbon to capture additional abatement measures.

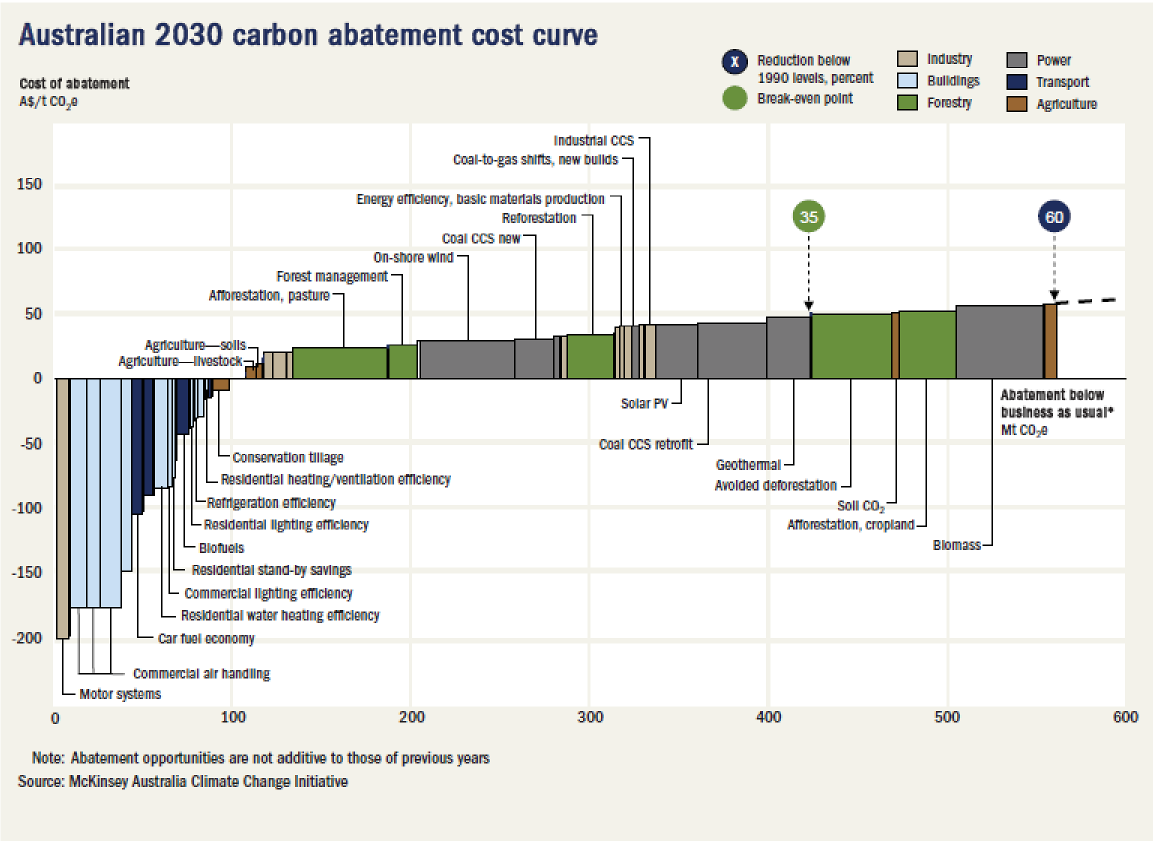

Figure 8: McKinsey and Co Greenhouse Gas Abatement Cost Curve (McKinsey & Co, 2008).

[8] Reported as \$45/tonne in 2007 in the original report and then converted to 2010 prices with a 3% rate of inflation.

3. REVIEW OF POLLUTANT ABATEMENT DATA

The following literature review provides order of magnitude estimates, assumptions and variables for pollution abatement costs. Additional data from government departments, utilities and Treasury is required to reduce uncertainty for capital and operating costs and to define the effectiveness in particular locations. Nonetheless, the literature draws upon a number of studies in SEQ and uses other Australian or international studies to fill data gaps. In most cases there was sufficient data to approximate capital and operating costs to capture economies of scale and to outline the uncertainty.

There have been a number of reviews of costs and effectiveness for pollution abatement for waterways. In particular, the South Creek Nutrient Trading Scheme in the Hawkesbury-Nepean provides an Australian example of reviews of cost-effectiveness followed by the implementation of a market based instrument (Frecker and Cuddy, 1994; James, 1994; Attwater, Booth *et al.*, 2002). In SEQ, there have also been a number of reviews of cost-effectiveness for pollution abatement (Rolfe, Donaghy *et al.*, 2005; Alam, Rolfe *et al.*, 2008) and reports considering possible nutrient trading schemes (BDA Group, 2005; BDA Group, 2007). There have also been a number of reviews of stormwater pollution control in SEQ as well as Australian (Taylor, 2005; Taylor, 2010; Water by Design, 2010) and international reviews (USEPA, 1993; WERF, 2010).

Part of the challenge of using data from the literature review was developing a common basis for comparison and expressing the data in performance bands to reduce the range of estimates. Unfortunately, there was limited agreement between a number of the main references as to the most cost-effective means of reducing particular pollutants in SEQ. One main study in SEQ identified point source abatement as the most cost-effective measure for nitrogen and phosphorus (Rolfe, Donaghy *et al.,* 2005 – Table 3.17). This study assumed that 50% of forward estimates of capital expenditure for treatment plant upgrades was for nutrient abatement. Another SEQ study reported a large range in cost effectiveness estimates for point and diffuse source abatement (BDAGroup, 2007 – Table 12.1). This provided a more complex picture where some treatment plant processes were relatively cost effective while other were not and similarly for diffuse abatement (BDAGroup, 2007 – Appendix 3). A third study in SEQ provided estimates that were many orders of magnitude higher than both of the other studies (EPA, 2008).

The large uncertainty in the estimates suggests the difficulty of a general conclusion about the most cost-effective abatement without specifying abatement details, the calculation procedure and the context of the abatement. This was reflected in a number of different contexts of pollution abatement in Australia. For example, it was noted that in the NSW South Creek Nutrient Trading Scheme that nutrient abatement from agriculture was one thousand times more cost-effective than the marginal cost of upgrading wastewater treatment plants. In Port Phillip Bay, Victoria, best practice as well as buffer strips on horticultural land was the most cost-effective nutrient removal, followed by Wastewater Treatment Plan (WWTP) upgrades, while reducing nutrients from urban runoff was the most expensive option. In South Australia, WWTP upgrades were the most expensive option for nitrogen removal while wetlands were the most expensive option for phosphorus removal (BDA Group, 2005 – Table 16, p54). The cost-effectiveness of point source abatement is particularly difficult to assess from literature without specifying the current level of treatment and change in concentration sought and the scope of costs considered.

The range of abatement options was not exhaustive and focussed on a number of common approaches currently used in SEQ. The review is organised by point source and diffuse pollution abatement measures.

4. POINT SOURCE POLLUTION ABATEMENT

Typical untreated domestic wastewater has a range of Total Nitrogen (TN) of 20-70 milligrams per litre (mg/L) and a 'typical' value of 40 mg/L while Total Phosphorus (TP) has a range of 4-12 mg/L and a 'typical' value of 7 mg/L (Asano, Burton *et al.*, 2007 - Table 3-12). In SEQ, untreated sewage has a range of total nitrogen from 49-65 mg/L and a range of total phosphorus of approximately 7-13 mg/L (De Haas, Foley *et al.*, 2008 - Table 1). Many plants in SEQ have biological nutrient removal and the license conditions in 2006/7 ranged from 3- 5 mg/L for total nitrogen and 1-10 mg/L for total phosphorus (De Haas, Foley *et al.*, 2008 - Table 1). Table 6 presents the typical range of effluent quality after secondary treatment for a range of treatment processes presented by Metcalf and Eddy (Asano, Burton *et al.*, 2007 - Table 3-14). Advanced wastewater treatment plants and chemical precipitation were also considered.

Table 6: Typical range of nitrogen and phosphorus concentration after treatment stage.

		Range of Effluent Quality after Secondary Treatment							
Constituent	**Untreated wastewater**	**Conventional activated sludge**	**Conventional activated sludge with filtration**	**Activated sludge with BNR**	**Activated sludge with BNR and filtration***	**Membrane Bioreactor**	**Activated sludge with microfiltration and reverse osmosis**	**Advanced wastewater treatment**	**Chemical precipitation**
Refer note	*#1*				*#2*			*#3*	*#4*
Ammonia nitrogen (mg N/L)	12-45	1-10	1-6	1-3	1-2	<1-5	≤0.1	Uncertain	
Nitrate nitrogen (mg N/L)	0-trace	10-30	10-30	2-8	1-5	<10	≤1		
Nitrite nitrogen (mg N/L)	0-trace	0-trace	0-trace	0-trace	0-trace	0-trace	0-trace		
Total nitrogen (mg N/L)	20-70	15-35	15-35	3-8	2-5	<10	≤1	Uncertain	
Total phosphorus (mg P/L)	4-12	4-10	4-8	1-2	≤2	<0.3-5	≤0.5	0.13	0.1

Notes:
#1. In SEQ, untreated sewage has a range of total nitrogen from 49-65 mg/L and a range of total phosphorus of approximately 7-13 mg/L (De Haas, Foley et al., 2008 – Table 1).
#2. The performance of micro and ultra filtration can vary significantly depending upon membrane manufacturer and operating conditions. The rejection percentage ranges for both micro filtration and ultra filtration were reported by Metcalf and Eddy as ammonia 5-15%, nitrate 0-2% and phosphate 0-1% (Asano, Burton et al., 2007 – Table 8-21). However, one US study notes that it is possible to achieve phosphorus concentrations of 0.2 mg/L using filtration after chemical precipitation or biological phosphorus removal (Faeth, 2000 - p24).
#3. The feedwater was assumed to be 1mg/L phosphorus to avoid fouling of the membranes and a final concentration of 0.13 mg/L phosphorus was assumed. Nitrogen removal depends on the quality of the feedwater (CH2MHILL, 2008). If ammonia is added to control fouling of the membrane then there may be a net increase is nitrogen emissions (Ramsay, Hermanussen et al., 2010).
#4. Chemical precipitation can be used to lower total phosphorus to concentrations as low as 0.1mg/L (Asano, Burton et al., 2007 – p328). Biological nutrient removal was reported as being more cost-effective than metal dosing as the first stage of phosphorus removal (Faeth, 2000 – p25, 26), which is reflected in the order of treatment processes in many SEQ wastewater treatment plants.

The order of treatment processes and the assumed change in effluent concentration affects the cost-effectiveness calculations. It may not be technically possible to achieve particular effluent concentrations by simply applying more of a treatment process or combining a number of treatment processes. As the names suggest, the order of treatment follows primary, secondary and then tertiary treatment. Nutrient removal is a focus of many tertiary treatment processes. Treatment measures such as 'activated sludge' and Biological Nutrient Removal are often used as the first stage of nutrient removal. This can be followed by additional measures to reduce effluent concentrations such as filtration and precipitation. Chemical treatment processes such as precipitation can be used to achieve very low concentrations and can also be used to reduce concentrations to suitable levels for further processing in membrane systems of Advanced Wastewater Treatment.

Tables 7 and 8 provide a summary of the treatment trains considered in the following review of pollution abatement costs for wastewater treatment plants. The concentrations noted in brackets are the indicative ranges effluent quality after each treatment stage (Asano, Burton *et al.*, 2007 - Table 3-14).

Table 7: Assumed treatment train for wastewater treatment plant nitrogen abatement.

Treatment Train	Step 1	Step 2	Step 3
A	Biological nutrient removal after conventional activated sludge (from 15-35mg/L to 3-8mg/L)	Tertiary filtration (from 3-8mg/L to 2-5 mg/L)	Advanced wastewater treatment following tertiary filtration (from 2-5 mg/L to the design performance of 0.81 mg/L. This assumes that tertiary filtration would avoid the use of ammonia in the AWTP plant and give the design reduction in nitrogen)
B	Biological nutrient removal after conventional activated sludge (from 15-35mg/L to 3-8mg/L)	Advanced wastewater treatment following biological nutrient removal (from 3-8 mg/L with a 5% reduction based on current performance in SEQ)	
C	Controlled irrigation after activated sludge		

Table 8: Assumed treatment train for wastewater treatment plant phosphorus abatement.

Treatment Train	Step 1	Step 2	Step 3
A	Biological nutrient removal (from 4-10 mg/L to 1-2 mg/L)	Chemical precipitation (from 1-2mg/L to 0.1 mg/L)	
B	Biological nutrient removal (from 4-10 mg/L to 1-2 mg/L)	Filtration (1-2mg/L to 0.13 mg/L)	
C	Biological nutrient removal (from 4-10 mg/L to 1-2 mg/L)	Chemical precipitation (from 1-2mg/L to 1 mg/L)	Advanced wastewater treatment plant (from 1mg/L to 0.1 mg/L)
D	Chemical precipitation (from 4-10mg/L to 2 mg/L)	Filtration (from 2 mg/L to 0.13 mg/L)	
E	Controlled irrigation		

4.1. Biological Nutrient Removal

4.1.1. Capital and Operating Cost

The US EPA reviewed processes and costs for 43 projects in Marylands and 23 projects in Connecticut for Biological Nutrient Removal (BNR) (EPA, 2007) as part of the US EPA Clean Water State Revolving Fund (CWSRF). Only upgrades were considered as part of the data below to capture the incremental cost for nutrient removal. Operating and maintenance costs only included incremental costs associated with the new equipment (EPA, 2007 - p11). The unit costs for operating and maintenance were reported as approximately 22% of capital costs for very small plants (4000 gallon per day) and reduced to about 17% of capital costs for small plants (100 000 gallon per day plant, noting that 0.1 mg/d (million gallons per day) = 0.37 megalitres/day) (EPA, 2007 - p12, Exhibit 11). Operating and maintenance costs were not reported for medium and larger sized plants and were also assumed to be 17% in the absence of additional information.

A review of Australian new BNR plants and upgrades performed between 1985 and 1989 also provided capital and operating costs (Hartley, 1995). Capital costs were in a similar range to those in the US EPA review but operating costs were generally much higher. This may reflect transport costs for sludge disposal which was not included in US estimates (USEPA, 2008). Sludge disposal costs are relatively high in SEQ where off-site disposal is required (see Chemical Precipitation - Metal Dosing) and are unlikely to change with increases in economies of scale. A separate study estimated operating, maintenance and sludge removal costs for a 1 mg/d biological phosphorus removal plant was approximately $US 35 000 per year and the capital cost for the plant was approximately $US 1million (Faeth, 2000 -p25, 26). This is equivalent to 4% of the capital costs for sludge disposal alone.

A range of BNR treatment processes were used in the US retrofits reported by the EPA and included Modified Ludzack-Ettinger (MLE), A/O Process, Step Feed Process, Bardenpho Process (Four-Stage), Modified Bardenpho Process, Sequencing Batch Reactor Process, Modified University of Cape Town (UCT) Process, Rotating Biological Contactor (RBC) Process and Oxidation Ditch. A range of processes are used in SEQ including activated sludge, phoredox, oxidation ditch and Bardenpho process (five-stage). Consequently, the US EPA data only provides an indication of cost for BNR treatment and may not capture the focus of treatment in SEQ. In a review of 35 wastewater treatment plants in SEQ, it was reported that in 2006/7 three plants accounted for approximately 60% of average dry weather flow, which is reflected in the total loads of phosphorus and nitrogen from the wastewater treatment plants.(De Haas, Foley *et al.*, 2009) In addition, many plants have been upgraded since 2006/7 including advanced water treatment plants and water recycling.

Biological nutrient removal can be a source of nitrous oxide which is a potent greenhouse gas (Foley and Lant, 2007; De Haas, Foley *et al.*, 2008; Foley, Lant *et al.*, 2008; De Haas, Foley *et al.*, 2009). The cost of greenhouse gas pollution has not been considered in the following cost review.

Table 9: Estimate of Capital and Operating Costs for Biological Nutrient Removal.

Plant size(MLd)	Plant capital cost ($US 2006)	Plant operating costs per year ($US million 2006)	Plant capital costs ($AUD million 2010)	Plant operating costs per year ($AUD million 2010)	Plant operating costs over a 20 year period & 3% discount ($AUD million 2010)	Total capital and operating costs over 20 years ($AUD million 2010)	Operating costs per mega litre day plant capacity ($AUD millions/MLD)	Capital costs per mega litre day plant capacity ($AUD millions/MLD)	Cost allocation to nitrogen removal ($AUD million 2010)	Cost allocation to phosphorus removal ($AUD million 2010)
#1			*#1, #2*	*#1, #2, #3*	*#3*				*#4*	*#4*
0.379	0.70	0.03	0.87	0.035	0.52	0.55	1.37	2.30	0.42	0.14
3.79	1.74	0.07	2.18	0.087	1.30	1.39	0.34	0.58	1.04	0.35
37.9	5.89	0.24	7.36	0.294	4.38	4.68	0.12	0.19	3.51	1.17

Notes:
#1 Based on EPA (2007 – Exhibit 8) for average data and Asano, Burton et al. (2007 – pp1463) for unit conversions. Cost was provided for plant capacity ranges of 0-0.379, 3.79-37.9 and >37.9 MLD. The upper value in each range was used to calculate cost-effectiveness for each plant capacity range.
#2 Assumed a 2010 exchange rate of $1AUD = $0.9USD based upon the Reserve Bank of Australia's exchange rates for 2010 http://www.rba.gov.au/statistics/hist-exchange-rates/ and inflated to 2010 prices using an inflation rate of 3% based upon the Reserve Bank of Australia's inflation calculator http://www.rba.gov.au/calculator/annualDecimal.html
#3 Assumed operating cost to be 4% of capital cost based upon (Faeth, 2000 - p25, 26) and a discount rate of 3%.
#4 It was assumed that 75% of BNR treatment costs were allocated to total nitrogen removal and 25% to total phosphorus removal (Pickering and Marsden, 2007 – p41).

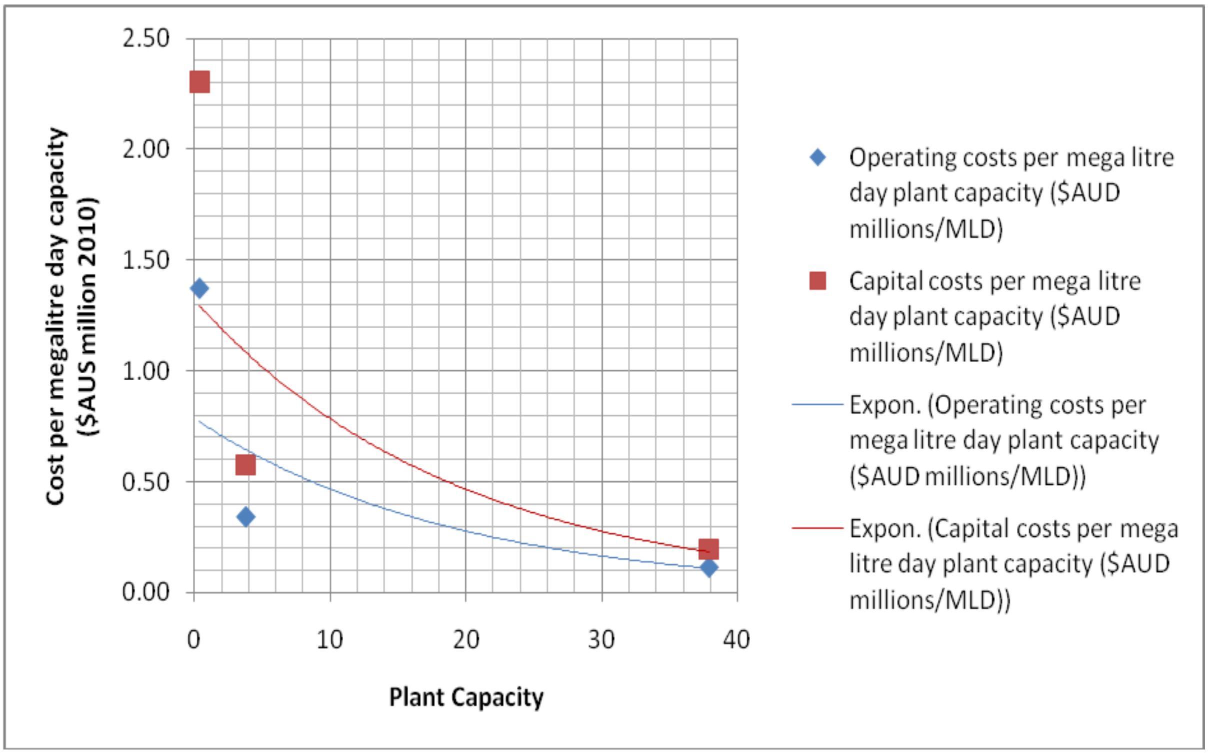

Figure 9: Relationship of capital and operating costs to plant capacity for Biological Nutrient Removal upgrades.

4.1.2. Cost-Effectiveness

The amount of nutrient removed by the BNR plant depends on the assumed feedwater to the BNR process. The Maryland Department of Environment undertook BNR upgrades of wastewater treatment plants to reduce total nitrogen from 40-60 mg/L to 8 mg/L and total phosphorus from 6-8 mg/L to 2 mg/L (Yu, 2007). About half of the plants in Connecticut were upgraded to 6-8 mgTN/L while the remaining plants were upgraded to 3-5 mg/L (EPA, 2007 – p9). The initial concentrations were similar to 'typical' total nitrogen and total phosphorus concentrations in untreated domestic wastewater. This suggests that plants were upgraded from primary treatment to biological nutrient removal. However, SEQ treatment plants are generally secondary, tertiary or advanced wastewater treatment plants. Consequently, any upgrade from a treatment process such as conventional activate sludge to biological nutrient removal would have a smaller reduction in concentration. Conventional activated sludge achieves TN and TP concentrations of 15-35 mg N/L and 4-10 mg P/L respectively (Asano, Burton *et al.*, 2007 - Table 3-14). It was assumed that upgrades of plant in SEQ would be from conventional activated sludge to biological nutrient removal. Biological nutrient removal achieves TN and TP concentrations of 3-8 mg N/L and 1-2 mg P/L respectively (Asano, Burton *et al.*, 2007 - Table 3-14). This is equivalent to a change in concentration of total nitrogen from 12-27 mg N/L and change in total phosphorus concentration of 3-8 mg P/L. Table 10 presents a review of the cost-effectiveness for pollution abatement using Biological Nutrient Removal.

Table 10: Cost-effectiveness for nitrogen and phosphorus removal for Biological Nutrient Removal.

Plant size (MLd)	Cost allocation to nitrogen removal ($AUD million 2010)	Cost allocation to phosphorus removal ($AUD million 2010)	LOW Reduction in total nitrogen concentration (mg/L)	HIGH Reduction in total nitrogen concentration (mg/L)	LOW Reduction in total phosphorus concentration (mg/L)	HIGH Reduction in total phosphorus concentration (mg/L)	LOW Total nitrogen load reduction over 20 years (tonnes)	HIGH Total nitrogen load reduction over 20 years (tonnes)	LOW Total phosphorus load reduction over 20 years (tonnes)	HIGH Total phosphorus load reduction over 20 years (tonnes)	LOW Cost-effectiveness for TN removal ($AUD/tonne)	HIGH Cost-effectiveness for TN removal ($AUD/tonne)	LOW Cost-effectiveness for TP removal ($AUD/tonne)	HIGH Cost-effectiveness for TP removal ($AUD/tonne)
#1	*#1*	*#1*	*#2*	*#2*	*#2*	*#2*								
0-0.379	0.42	0.14	12	27	3	8	33	75	8	22	12524	5566	16699	6262
3.79-37.9	1.04	0.35	12	27	3	8	332	747	83	221	3129	1391	4172	1565
>37.9	3.51	1.17	12	27	3	8	3320	7470	830	2213	1056	469	1408	528

Notes:
#1 Calculated in Table 9.
#2 Conventional activated sludge achieves TN and TP concentrations of 15-35 mg N/L and 4-10 mg P/L respectively (Asano, Burton et al., 2007 - Table 3-14). It was assumed that upgrades of plant in SEQ would be from conventional activated sludge to biological nutrient removal. Biological nutrient removal achieves TN and TP concentrations of 3-8 mg N/L and 1-2 mg P/L respectively (Asano, Burton et al., 2007 - Table 3-14). This is equivalent to a change in concentration of total nitrogen from 12-27 mg N/L and change in total phosphorus concentration of 3-8 mg P/L.

4.1.3. Uncertainty

Capital costs were based upon 65 treatment plant upgrades as part of a US EPA program and were considered as fair (+/- 30%). Operating costs were also based upon the EPA data, however less detail was available for these costs and they were much lower than reported costs in a study of Australian plants. This may reflect high costs for sludge disposal. Operating costs for medium to large plants were limited and not consistent. The data was considered as poor (>30%). Effectiveness was based upon changes in concentration achieved in the US EPA program and was consistent with performance reported in engineering textbooks. The variation was captured with a high and low range estimate and was considered good (+/- 15%). However, the applicability of the data will depend upon plant configuration.

4.2. Tertiary Filtration

4.2.1. Capital and Operating Costs

Filtration is used to remove a range of pollutants including suspended solids as well as large organic molecules, large colloidal particles and many microorganisms in the case of micro filtration or ultra filtration (Asano, Burton *et al.*, 2007 – p430). There are a number of different types of filtration, although the cost estimates considered do not provide specific details of the processes considered. Tertiary filtration is also used for removal of other pollutants including nutrients. It was assumed that nutrient removal was the reason for the addition of filtration to the treatment process and the filter would remove both nitrogen and phosphorus. Table 11 presents an overview of available data for capital and operating costs for tertiary filtration and was arranged by plant capacity.

Table 11: Review of Tertiary Filtration capital and operating costs.

Reference	Plant size (ML/d)	Capital cost ($US million in year 2000)	Operating cost per year ($US million in year	Capital cost ($AUD million in year 2005)	Operating cost per year (($AUD million in year 2005)/year)	Capital cost ($AUD million in year 2010)	Operating cost per year (($AUD million in year 2010)/year)	Operating cost over 20 years & 3% discount rate ($AUD million in year 2010)	Total cap op ($AUD million in year 2010)	Capital cost per mega litre per day treatment capacity ($AUD million 2010/ MLD)	Operating cost per year per mega litre day treatment capacity ($AUD million
						#1	#1				
(Hausler 2006) - p 45	1			0.65	0.033	0.754	0.038	0.57	1.32	0.75	0.57
(Faeth 2000) – p25, 26	3.79	2.5	0.08			3.73	0.12	1.78	5.51	0.98	0.47
(QWC 2010) – Tab. A-4	7.9					6.9	0.38	5.65	12.55	0.87	0.72
(BDAGroup 2005) – Fig. A3.3	25			11	0.25	12.8	0.29	$4.3	$4.6	0.51	0.17
(QWC 2010) – Tab. A-4	50					24	1.30	19.34	43.34	0.48	0.39
(BDAGroup 2005) – Fig. A3.3	50			15	0.5	17.4	0.58	8.6	$9.2	0.35	0.17
(BDAGroup 2005) – Fig. A3.3	75			17	0.75	19.7	0.87	12.9	13.8	0.26	0.17
(BDAGroup 2005) – Fig. A3.3	100			22	1	25.5	1.16	17.2	18.4	0.26	0.17

Note:
#1. Assumed a 2010 exchange rate of $1AUD = $0.9USD based upon the Reserve Bank of Australia's exchange rates for 2010 http://www.rba.gov.au/statistics/hist-exchange-rates/ and inflated to 2010 prices using an inflation rate of 3% based upon the Reserve Bank of Australia's inflation calculator http://www.rba.gov.au/calculator/annualDecimal.html

Figure 10 plots the data and suggests an economy of scale for tertiary filtration. However, one of the references noted that there was only small decreases in unit costs as plants became bigger (Faeth, 2000 – p26). The data includes references from SEQ where the capital and operating costs and the change in concentration were noted.

Central Queensland University (CQU) also estimated the cost of plant upgrades to remove nitrogen (Rolfe, Donaghy *et al.*, 2005 - p56). The estimates for point source load reductions were made from EPA capital projections for plant upgrades. The average annual cost-effectiveness for reducing nitrogen from point sources was reported as $AUD 6,729 (2005)/tonne. However, capital and operating costs were not reported separately, the size of the plants was not noted or the change in concentration achieved. The estimate was similar to the cost-effectiveness of nitrogen abatement of $AUD 9,375 (2005)/tonne calculated for Luggage Point STP (BDA Group, 2005 - Table 24) if expressed as the total cost divided by the total load over a 20 year period with a 3% discount rate.

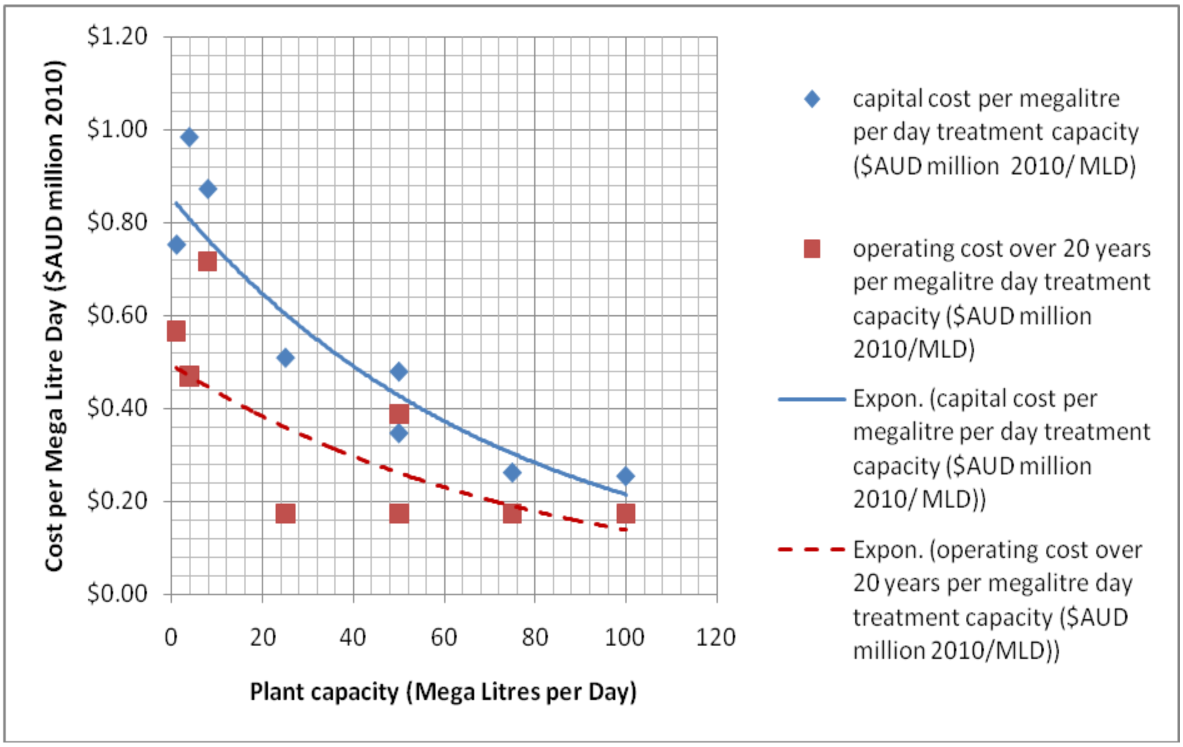

Figure 10: Tertiary filtration capital and operating costs per unit of treatment capacity.

Table 12 presents the capital and operating costs for tertiary filtration for a range of plant sizes based upon Figure 10. The same cost drivers as assumed for biological nutrient removal were assumed for tertiary filtration, that is, 75% of the costs were allocated to nitrogen removal and 25% to phosphorus removal (Pickering and Marsden, 2007 – p41). Table 13 uses the cost allocations with the range in performance to calculate a range of cost-effectiveness for tertiary filtration. In summary, the cost-effectiveness was sensitive to the size of the plant, changes in concentration and the allocation of costs between nitrogen and phosphorus.

Table 12: Tertiary Filtration capital and operating costs based upon fitted curve for economies of scale.

Plant capacity (MLD)	Capital cost per mega litre day capacity ($AUD million 2010/MLD)	Operating cost per mega litre day capacity ($AUD million 2010/MLD)	Capital cost per plant ($AUD million 2010/plant)	Operating cost per plant ($AUD million 2010/plant)	Total capital and operating cost per plant ($AUD million 2010/plant)	75% allocation to N of total capital and operating costs ($AUD million 2010)	25% allocation to P of total capital and operating costs ($AUD million 2010)
	#1	#1				#2	#2
5	0.8	0.48	4	2.4	6.4	4.8	1.6
10	0.72	0.44	7.2	4.4	11.6	8.7	2.9
20	0.64	0.38	12.8	7.6	20.4	15.3	5.1
50	0.42	0.26	21	13	34	25.5	8.5
100	0.22	0.14	22	14	36	27	9

Notes:
#1 Based upon Figure 10.
#2 It was assumed that 75% of BNR treatment costs were allocated to total nitrogen removal and 25% to total phosphorus removal (Pickering and Marsden, 2007 – p41).

4.2.2. Cost-Effectiveness

Table 13 presents a review of cost-effectiveness for tertiary filtration.

Table 13: Tertiary Filtration Cost-effectiveness.

Plant capacity (MLD)	75% allocation to N of total capital and operating costs ($AUD million 2010)	25% allocation to P of total capital and operating costs ($AUD million 2010)	N change in concentration (mg/L)		P change in concentration (mg/L)		N Load reduced (tonnes over 20 years)		P Load reduced (tonnes over 20 years)		N cost-effectiveness ($AUD/tonne)		P cost-effectiveness ($AUD/tonne)	
			low	high	low	high	low	high	low	high	low	high	low	high
	#1	*#1*	*#2*	*#2*	*#2*	*#2*								
5	4.8	1.6	1	3	0.8	1.8	37	110	29	66	131507	43836	54795	24353
10	8.7	2.9	1	3	0.8	1.8	73	219	58	131	119178	39726	49658	22070
20	15.3	5.1	1	3	0.8	1.8	146	438	117	263	104795	34932	43664	19406
50	25.5	8.5	1	3	0.8	1.8	365	1095	292	657	69863	23288	29110	12938
100	27	9	1	3	0.3	1.2	730	2190	219	876	36986	12329	23117	10274

Notes:
#1 From Table 12.
#2 Based on Asano, Burton *et al.* (2007 - Table 3-14). It was assumed that tertiary filtration would follow biological nutrient removal and reduce the concentration of nitrogen from 3-8 mg/L to 2-5 mg/L and the concentration of phosphorus from 1-2 mg/L to 0.2 mg/L.

4.2.3. Uncertainty

Capital costs were based upon a review of eight plants and appeared to closely follow an economy of scale relationship. The data was considered as fair (+/-30%). Operating costs were based upon a review of eight plants and appeared to follow and economy of scale relationship but with some variation. The data was considered as fair (+/-30%).

Effectiveness was based upon an engineering textbook for pollutant removal performance for nitrogen. The range was documented and captured with a low and high estimate to reduce the uncertainty and was considered as good (+/-15%). However, the pollutant removal effectiveness for phosphorus was based upon a single plant and the data accuracy was considered poor (>30%) and requires further review.

4.3. Chemical Precipitation - Metal Dosing

4.3.1. Capital and Operating Costs

There was a small economy of scale for capital costs for precipitation of phosphorus. Operating costs were relatively large due to the cost of metal salts required to precipitate phosphorus and sludge disposal. This suggests that the cost-effectiveness will be sensitive to operating costs. Capital costs begin to be more significant for small plants with small reductions in the concentration of phosphorus. This may account for one study which considered small plants and noted that cost per unit treatment declines significantly as the plant size increases (Faeth, 2000 – p25).

BDA Group estimated the cost-effectiveness for precipitation using phosphorus at a number of wastewater treatment plants in Brisbane(BDAGroup, 2005- Table 26) as well as a reporting of a general relationship for capital and operating costs for plant size (BDAGroup, 2005 – Figure A3.1).

Note that operating costs in Figure 11 are present values over a 20-year period. The discount rate for the present values was not reported. In addition, the operating costs in Figure 11 do not include sludge disposal costs (BDAGroup, 2005 – p129). However, it was estimated that phosphorus precipitation can lead to increases in sludge production by 40% or more (Driver, Lijmbach *et al.*, 1999). It was also reported that the annual cost for operating, maintenance and sludge disposal for chemical precipitation of phosphorus was $US 60 000 per mega gallon per day plant size in the year 2000, with no economy of scale for operating costs (Faeth, 2000 – p25). This is equivalent to $AUD 0.47 million in 2010 for a 20 ML/day (megalitre per day) plant (assuming an exchange rate of 0.9 and an inflation rate of 3%). This *annual* cost is greater than the estimate of both the net present value of capital and operating over 20 years for a plant of this scale as estimated in Figure 11.

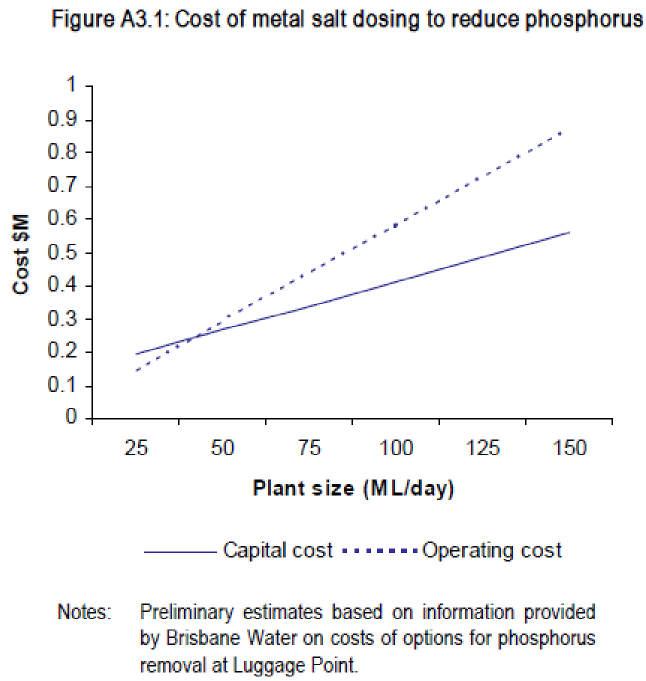

Figure 11: Precipitation of phosphorus capital and operating costs and plant capacity NOT including sludge management (Source: BDAGroup, 2005 – Fig A3.1).

The USEPA *Process Design Manual for Phosphorus Removal* recommends that chemical sludge volumes for phosphorus precipitation be calculated at 35% greater than suggested by the stoichiometry (USEPA, 1976 – p11-10). On the other hand, another reference notes that concentrations of 2-3 times greater than the stoichiometry are required to achieve substantial phosphorus reduction (Driver, Lijmbach *et al.*, 1999). The difference in the estimates is related to the concentration of phosphorus sought in the effluent. This point is crucial for the development of the marginal cost for phosphorus abatement using chemical precipitation.

The scenario presented by BDA was for a reduction in total phosphorus from 6-7 mg/L to 2 mg/L (BDAGroup, 2005 - Table 26; and Figure A3.1). These concentrations suggest chemical precipitation applied to conventional activated sludge processes (Asano, Burton *et al.*, 2007 - Table 3-14), or at least minimal phosphorus removal if operating biological nutrient removal (BNR) processes, and is approximately a 70% reduction in phosphorus concentrations. A 75% reduction in phosphorus requires a typical mole ratio of 1.4 Aluminium:1 Phosphorus (Asano, Burton *et al.*, 2007 – Table 6-6). Further reductions in phosphorus concentrations beyond about 2 mg/L for secondary effluent would incur significant cost. For example, a 95% reduction in phosphorus requires dosing of precipitant at a mole ratio of 2.3 Aluminium:1 Phosphorus (Asano, Burton *et al.*, 2007 – Table 6-6). The stoichiometry defines that each milligram of phosphorus removed using aluminium precipitate results in 7.48 mg of chemical sludge (USEPA, 1976 – p11-10). This means that a mole ratio of 1.4 Aluminium:1 Phosphorus to achieve up to 75% reduction in phosphorus will produce approximately

10.5 mg of chemical sludge for each milligram of phosphorus removed. Similarly, to achieve up to 95% reduction in phosphorus requires a mole ratio of 2.3 Aluminium:1 Phosphorus and will produce approximately 17.2 mg of chemical sludge for each milligram of phosphorus removed.

However, it was noted that biological nutrient removal was more cost-effective than metal dosing as the first stage of phosphorus removal (Faeth, 2000 – p25, 26). This is important for considering the target concentration and the need to use increasing amounts of precipitant. For example, assuming that BNR can reduce total phosphorus concentrations to 1-2 mg/L (Asano, Burton *et al.*, 2007 - Table 3-14) then a 75% reduction in concentration using chemical precipitation will result in concentrations of 0.25-0.5 mg/L. The literature notes that chemical precipitation can lower total phosphorus concentrations to 0.1 mg/L (Asano, Burton *et al.*, 2007 – p328). This correlates to about a 95% reduction in phosphorus from BNR effluent and would require double the dose of precipitant (and create almost double the sludge volumes).

Sludge management costs were estimated from data for sludge management practices for biosolids. Sludge management and handling costs are significant and can be up to 50% of total wastewater treatment costs (NRMMC, 2004 – p10). In addition, transportation costs were identified as the most significant constraint to beneficial reuse programs (DERM, 2002 – p90). Transport costs can be reduced through concentrating the waste (reducing the moisture content) (DERM, 2002 – p90). Consequently, any cost estimates will be sensitive to assumptions about costs included in sludge management as well as moisture content and the distance travelled for disposal or reuse.

Sludge management costs include processes such as dewatering operations at the wastewater treatment plant, as well as costs for transport, land application and any other fees associated with disposal. The Australian and New Zealand Biosolids Partnership recently reported that the 'average cost for biosolids management is in the order of $300/dry tonne' (ANZBP, 2011 – p2). In 2002, DERM noted that the cost for dewatering sludge was $300/dry solid tonne (dst) based on contract dewatering prices of $12.5 per m^3 at 3.75% solids infeed (DERM, 2002 - p56). The cost for transport and application was estimated to be about $110/dst based upon costs from Sydney Water (application costs of about $50-60/dst and transport of $50/dst) (DERM, 2002 - p51, 52). If these values are appreciated to 2011 values (using a 3% inflation rate based on RBA data for the period) then the costs are approximately $390/dst for dewatering and $140/dst for transport and processing.

The following estimate of sludge management costs was based upon a large wastewater treatment plant in SEQ. It was noted that $50/wet tonne was an appropriate value for cartage and beneficial application to agriculture (Pers. Com. Grahame Simpson, Urban Utilities, 15 June 2011). This is equivalent to approximately $250/dry tonne assuming a moisture content of approximately 80%. The capital and operating costs for processes downstream of the digesters were estimated at approximately $150/dst (Pers. Com. Grahame Simpson, Urban Utilities, 17 June 2011).

This included:

> Capital - $70/dry tonne (based on a $7.5M investment incorporating centrifuges, polymer (silo, batching + dosing), sludge building, sludge outloading, storage, design and commissioning costs with 20 year asset life and 7% cost of money)
>
> Operating and Maintenance - $35/dry tonne
>
> Polymer - $40/dry tonne
>
> Power - $5/dry tonne
>
> Total - $150/dry tonne

The above cost estimates were used for the following two sludge disposal scenarios.

Scenario 1 applies to treatment plants that transport sludge for about 50-100 km for application or disposal. A total cost of $400/dst was assumed based upon a capital and operating cost of $150/dst for processing at the treatment plant and $250/dst for transport and application/disposal.

Scenario 2 applies to treatment plants that do not transport sludge and is a minimum estimate that includes only the wastewater treatment plant operations. A total cost of $150/dst was assumed based upon the capital and operating costs for processing sludge at the treatment plant.

Table 14, 15 and 16 present the capital and operating costs including two sludge management cost options. The tables also capture different effluent types and target concentrations for precipitation. A sludge disposal cost of $400/dst means that most of the cost for phosphorus precipitation for conventional activated sludge effluent would be related to sludge management. If the sludge management cost was $150/dst, then about a third of the cost was related to sludge management. There was only a small economy of scale when operating costs were high.

Table 14: Capital and operating costs for precipitation of 75% phosphorus from conventional activated sludge effluent to 2 mg/L.

Plant size (Mega litres per day)	Capital cost ($AUDmillion2010)	Present Value of operating cost NOT including sludge ($AUD2010million)	Present Value of capital and operating cost NOT including sludge disposal ($AUD2010million)	Present Value of Sludge Management Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management Costs ($AUD2010million)	Present Value of Sludge Management and Disposal Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management and Disposal Costs ($AUD 2010million)
Note	*#1, #2*			*#3*		*#4*	
20	0.26	0.21	0.4	0.77	1.23	2.05	2.51
50	0.35	0.44	0.7	1.92	2.71	5.12	5.91
75	0.44	0.60	1.0	2.88	3.92	7.68	8.72
100	0.52	0.75	1.2	3.84	5.11	10.24	11.51

Notes:
#1. Based upon (BDAGroup, 2005 – Figure A3.1). Note that operating costs do not account for the increased volume of sludge produced by phosphorus precipitation (BDAGroup, 2005 –p129) and operating values are present values over 20 years and the discount rate is not reported.
#2. Inflated to 2010 prices using an inflation rate of 3% based upon the Reserve Bank of Australia's inflation calculator http://www.rba.gov.au/calculator/annualDecimal.html
#3. 13.4 mg chemical sludge per mg of phosphorus removed was assumed. This was based upon a 35% dose rate greater than the 9.96 mg chemical sludge per mg of phosphorus removed based on the stochiometry of precipitation with iron precipitant (USEPA, 1996 –p 11-10). The amount of phosphorus removed was 1.25 mg/L based upon an assumed effluent concentration from Biological Nutrient Removal effluent of 1-2 mg/L (Asano, Burton *et al.*, 2007 - Table 3-14) and 75% reduction in concentration. A unit cost of sludge disposal of $150/dry solid tonne was assumed based upon data provided for Luggage Point WWTP for biosolids disposal. This does not include transport and disposal costs.
#4. As for #3 but assuming a unit cost of $400/dry solid tonne and includes transport and disposal costs.

Table 15: Capital and operating costs for precipitation of 75% phosphorus from Biological Nutrient Removal effluent from 1-2 mg/L to 0.25-0.5 mg/L.

Plant size (Mega litres per day)	Capital cost ($AUDmillion2010)	Present Value of operating cost NOT including sludge ($AUD2010million)	Present Value of capital and operating cost NOT including sludge disposal ($AUD2010million)	Present Value of Sludge Management Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management Costs ($AUD2010million)	Present Value of Sludge Management and Disposal Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management and Disposal Costs ($AUD 2010million)
Note	*#1, #2*	*#1, #2*		*#3*		*#4*	
20	0.26	0.21	0.4	0.21	0.67	0.55	1.01
50	0.35	0.44	0.7	0.51	1.30	1.37	2.16
75	0.44	0.60	1.0	0.77	1.81	2.06	3.10
100	0.52	0.75	1.2	1.03	2.30	2.74	4.02

Notes:
#1. Based upon (BDAGroup, 2005 – Figure A3.1). Note that operating costs do not account for the increased volume of sludge produced by phosphorus precipitation (BDAGroup, 2005 – p129) and operating values are present values over 20 years and the discount rate is not reported.
#2. Inflated to 2010 prices using an inflation rate of 3% based upon the Reserve Bank of Australia's inflation calculator http://www.rba.gov.au/calculator/annualDecimal.html
#3. 13.4 mg chemical sludge per mg of phosphorus removed was assumed. This was based upon a 35% dose rate greater than the 9.96 mg chemical sludge per mg of phosphorus removed based on the stochiometry of precipitation with iron precipitant (USEPA, 1996 –p 11-10). The amount of phosphorus removed was 1.25 mg/L based upon an assumed effluent concentration from Biological Nutrient Removal effluent of 1-2 mg/L (Asano, Burton *et al.*, 2007 - Table 3-14) and 75% reduction in concentration. A unit cost of sludge disposal of $150/dry solid tonne was assumed based upon data provided for Luggage Point WWTP for biosolids disposal. This does not include transport and disposal costs.
#4. As for #3 but assuming a unit cost of $400/dry solid tonne and includes transport and disposal costs.

Table 16: Capital and operating costs for precipitation of 95% phosphorus from Biological Nutrient Removal effluent from 1-2 mg/L to 0.1 mg/L.

Plant size (Mega litres per day)	Capital cost ($AUDmillion2010)	Present Value of operating cost NOT including sludge ($AUD2010million)	Present Value of capital and operating cost NOT including sludge disposal ($AUD2010million)	Present Value of Sludge Management Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management Costs ($AUD2010million)	Present Value of Sludge Management and Disposal Costs ($AUD 2010million)	Present Value of Capital and Operating Costs with Sludge Management and Disposal Costs ($AUD 2010million)
Note	*#1, #2*			*#3*		*#4*	
20	0.26	0.21	0.4	0.40	0.86	1.07	1.53
50	0.35	0.44	0.7	1.00	1.79	2.66	3.45
75	0.44	0.60	1.0	1.50	2.54	3.99	5.04
100	0.52	0.75	1.2	2.00	3.27	5.33	6.60

Notes:
#1. Based upon (BDAGroup, 2005 – Figure A3.1). Note that operating costs do not account for the increased volume of sludge produced by phosphorus precipitation (BDAGroup, 2005 – p129) and operating values are present values over 20 years and the discount rate is not reported.
#2. Inflated to 2010 prices using an inflation rate of 3% based upon the Reserve Bank of Australia's inflation calculator http://www.rba.gov.au/calculator/annualDecimal.html
#3. 22.9 mg chemical sludge per mg of phosphorus removed was assumed. The dosage of precipitant was assumed to be 2.3 times greater than stoichiometry based upon (Asano, Burton *et al.*, 2007 – Table 6-6). The US EPA stoichiometry presented for other chemical precipitation was modified to reflect this change (USEPA, 1996 – p 11-10). The amount of phosphorus removed was 1.43 mg/L based upon an assumed effluent concentration from Biological Nutrient Removal Effluent of 1-2 mg/L (Asano, Burton *et al.*, 2007-Table 3-14) and 95% reduction in concentration. A unit cost of sludge disposal of $150/dry solid tonne was assumed based upon data provided for Luggage Point WWTP for sludge management.
#4. As for #3 but assuming a unit cost of $400/dry solid tonne and includes transport and disposal costs.

4.3.2. Cost-Effectiveness

The change in load was coupled together with the costs for phosphorus precipitation.

Table 17: Cost-effectiveness for precipitation of 75% phosphorus from conventional activated sludge effluent to 2mg/L.

Plant size (Mega litres per day)	P change in concentration (mg/L)	P load reduction over 20 years (tonne)	Cost-effectiveness for including sludge management ($AUD2010million/ tonne)	Cost-effectiveness with Sludge management and disposal ($AUD2010million/ tonne)
Note	#1		#2	#2
20	4.5	657	0.00187	0.00382
50	4.5	1643	0.00165	0.00360
75	4.5	2464	0.00159	0.00354
100	4.5	3285	0.00156	0.00350

Notes:
#1. See previous section for details on processes and concentrations; #2. See Table 14.

Table 18: Cost-effectiveness for precipitation of 75% phosphorus from Biological Nutrient Removal effluent from 1-2 mg/L to 0.25-0.5 mg/L.

Plant size (Mega litres per day)	P change in concentration (mg/L)	P load reduction over 20 years (tonne)	Cost-effectiveness for including sludge management ($AUD2010million/ tonne)	Cost-effectiveness with Sludge management and disposal ($AUD2010million/ tonne)
Note	#1		#2	#2
20	1.25	183	0.00367	0.0055
50	1.25	456	0.00285	0.0047
75	1.25	684	0.00265	0.0045
100	1.25	913	0.00252	0.0044

Notes:
#1. See previous section for details on processes and concentrations; #2. See Table 15.

Table 19: Cost-effectiveness for precipitation of 95% phosphorus from Biological Nutrient Removal effluent from 1-2 mg/L to 0.1 mg/L.

Plant size (Mega litres per day)	P change in concentration (mg/L)	P load reduction over 20 years (tonne)	Cost-effectiveness for including sludge management ($AUD2010million/ tonne)	Cost-effectiveness with Sludge management and disposal ($AUD2010million/ tonne)
Note	#1		#2	#2
20	1.425	208	0.00415	0.00735
50	1.425	520	0.00344	0.00663
75	1.425	780	0.00326	0.00646
100	1.425	1040	0.00315	0.00634

Notes:
#1. See previous section for details on processes and concentrations; #2. See Table 16.

4.3.3. Uncertainty

Capital costs were from a study that generalised a relationship of capital costs to plant scale based upon a number of plants in Brisbane. The data was considered fair (+/-30%).

Operating costs for sludge disposal were based upon literature and primary data collected from a single plant in Brisbane. This enabled a high and low range estimate to be developed based upon disposal requirements for chemical sludge. Sludge production volumes were based upon a US EPA method derived from the stoichiometry for precipitation and the target change in concentration. The data was considered as fair (+/- 30%).

Effectiveness was well defined in engineering textbooks and was considered as good (+/-15%).

4.4. Advanced Wastewater Treatment

The cost-effectiveness of pollution abatement from Advanced Wastewater Treatment Plants (AWTP) was very uncertain. In particular, it was not clear that a net reduction in nutrients was achieved by recently commissioned AWTPs in Brisbane. In addition, the cost allocated to pollutant removal was also sensitive to the assumed value of water. However, if an AWTP achieves its design performance and the water produced is valued as bulk water, then the cost allocated to nutrient removal is very small. Costs for dual reticulation associated with some demands for recycled water were not incorporated into the following estimates.

4.4.1. Capital and Operating Costs

CH2MHill's report on the reuse of purified recycled water in SEQ was used as the basis for cost estimates for advanced wastewater treatment (CH2MHill, 2008). The processes considered for advanced wastewater treatment were microfiltration, reverse osmosis and advanced oxidation. It was assumed that alum dosing, phosphorus precipitation, sludge thickening and dewatering were undertaken in existing sewage treatment plants, although it was noted that this was not the case for the Western Corridor Recycled Water Scheme. The report identified that it was generally more efficient and economical to reduce the nutrients as much as possible at the wastewater treatment plant before the effluent was fed to the advanced wastewater treatment plant (CH2MHill, 2008 - p20). In fact, effluent quality from the wastewater treatment plant was important to protect microfilters and membranes. Chloramination using aqueous ammonia and sodium hypochlorite was required to reduce biological activity which can effect microfiltration and reverse osmosis membranes. This, in turn, has an effect on the net reduction in total nitrogen, as discussed below. Chemical phosphorus removal was also required to reduce scaling on membranes if total phosphorus levels were above 1-2 mg/L in the wastewater effluent (CH2MHill, 2008 - p20).

The consideration of pipes and pumps required to convey wastewater to and from a treatment plant was also important for cost estimates. A preliminary cost estimation for three recycling options in SEQ noted that pipeline and pump stations accounted for three quarters of the total capital cost for one system (Hausler, 2006 – p8). The preliminary cost estimates were similar to the capital and operating costs reported by CH2MHill in the following figures. For example, for Loganholme to Beaudesert Recycled Supply it was estimated that 34 ML/day were available as a secondary effluent feed. It was reported that total capital costs were $97 million and total operating costs were $3.4 million per year (Hausler, 2006 – Table 3.3). In comparison, a 34 ML/day secondary effluent flow plant had a capital cost of approximately $125 million and operating cost of about $7 million per year using the CH2MHill figures presented below. The pipe and pumps were included by using cost estimates which specified the pipe and pumping costs.

The following figures were based upon QWC cost data and reproduced from the CH2MHill report (CH2MHill, 2008 - Figure 6-1 and Figure 6-2). The figures were reproduced to explore economy of scale relationships but were not used for cost estimates because they do not include pipe and pump costs. The report notes an economy of scale for treatment plant capital costs. However, there were limited data points in the figure for smaller plants and the curve fitted to the data appears to overstate the 'economies of scale' effect for larger scale plants. An alternative 'line of best fit' illustrates an alternative interpretation for all but the smallest plants (<5 ML/d inflows such as Noosa and Bribie Island schemes).

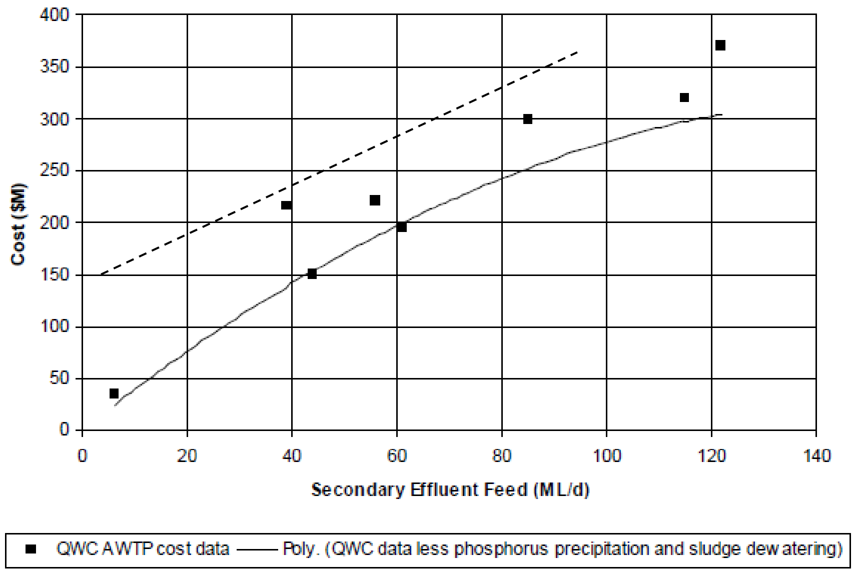

Figure 12: AWTP capital cost by plant size (CH2MHill, 2008 - Fig 6-1).

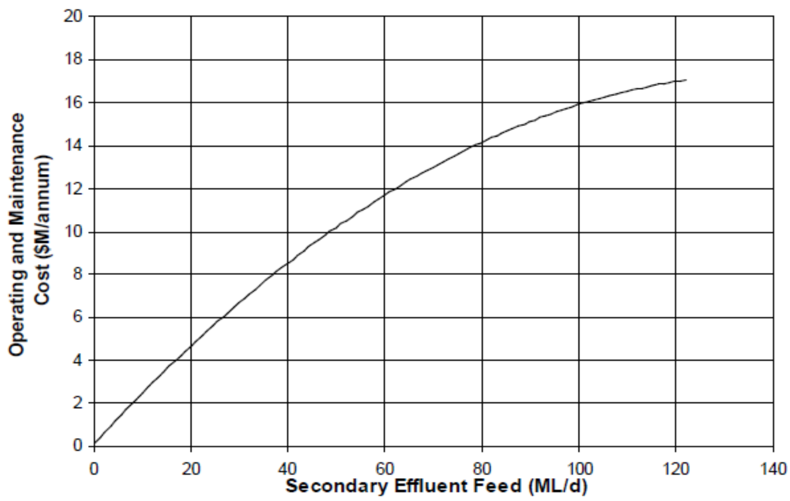

Figure 13: AWTP operating costs by plant size(CH2MHill, 2008 - Fig 6-2).

Table 20 illustrates the present value of capital and operating costs for AWTP schemes in SEQ based upon costs presented by CH2MHill (CH2MHill, 2008 – Table 6-2). The value of the water produced by the AWTP was important to determine the costs allocated to nutrient removal. CH2MHill used the capital and operating costs to calculate a levelised cost for potable water supply from AWTP. However, the following cost estimates subtracted the value of the water from capital and operating costs and the remaining cost was allocated to nutrient removal. In this case, if the present value of bulk water was based on the QWC indicative price path with a 3% discount rate, then there were no remaining costs for allocation to nutrient removal. However, the cost allocation to nutrients was sensitive to the discount rate. For example, if the discount rate was increased to 5.5% then the scheme costs become greater than the value of the water. The volume of water was assumed to be the plant design capacity. However, if plants were under-utilised then costs allocated for pollution abatement would increase. It should also be noted that the CH2MHill study assumed a 25-year evaluation period and a 7.5% discount rate (CH2MHill, 2008 – p52). In addition, the upper estimates of demand for some of the schemes were associated with developments with dual reticulation (CH2MHill, 2008 – p33). Additional costs associated with providing dual reticulation were not included in the following estimates.

Table 20: Capital and operating costs for AWTP and pipe systems compared to the value of recycled water.

	2026 Purified Recycled Water Flow (ML/d)	Scheme Capital (Million $AUD 2010)	Scheme Operating and Maintenance (Million $AUD2010/year)	Present Value Scheme Capital and Operating Excluding value of water (Million $AUD 2010)	Present Value of Water using 2026 ML/d (Million $AUD 2010)
North Pine Scheme – All Pine Rivers and Redcliffe, Sandgate, East Burpengary and South Caboolture WWTPs	80	494	22	829	1128
Gold Coast Scheme – All Gold Coast WWTPs	154	602	24	965	2171
All Gold Coast WWTPs excluding Beenleigh	130	570	25	942	1833
Sunshine Coast Scheme - All Maroochydore - (excluding Kenilworth), all Caloundra (excluding Maleny) and Noosaville WWTPs - Traveston Crossing Dam Stage 1	25	221	10	375	352
Redlands Scheme - Capalaba, Thorneside, Cleveland and Victoria Point WWTPs	32	250	11	419	451
Wyaralong Dam Scheme - Beaudesert North WWTPs	14	133	5	213	197
Toowoomba Scheme - Wetalla WWTP	14	85	5	154	197

Further discussion is required for the appropriate value of water to apply this approach. An alternative approach is to allocate a percentage of capital costs based on drivers for investment. An occasional paper from WSAA identified costs for wastewater services provides a possible guide (Pickering and Marsden, 2007). The removal of suspended solids was allocated a 5% share of both capital and operating costs for water recycling (Pickering and Marsden, 2007 – Table 12 and 13). The same assumption could be applied to other pollutants such as nutrients. Nitrogen and phosphorus removal could then be allocated 75% and 25% of the pollutant removal cost respectively based upon cost drivers reported for biological nutrient removal.

4.4.2. Cost-Effectiveness

An initial review by DERM showed a large range in performance for the reduction of total nitrogen from advanced waste water treatment plants (Ramsay, Hermanussen *et al.*, 2010). It appears that the use of ammonia for control of biological growth in microfilters and reverse osmosis membranes may lead to an *increase* in the load of total nitrogen in some cases. The overall effect of the three advanced water treatment plants (Bundamba, Gibson Island and Luggage Point) that became operational in 2008/2009 was reported as 'minimal' with a reduction of approximately 5%. However, only limited data was available (less than a year) and the influence of commissioning was not considered (Ramsay, Hermanussen *et al.*, 2010). The design brief for the three advanced water treatment plants required an 82% reduction in total nitrogen (CH2MHill, 2008 – p21). The reported water quality target for the plants was based upon the Western Corridor Recycled Water Scheme which assumed background concentration levels in Wivenhoe Dam of total nitrogen 0.81 mg/L and total phosphorus 0.13 mg/L (CH2MHill, 2008 – p16). To achieve an 82% reduction in nitrogen to 0.81 mg/L implies a feed water to the advanced water treatment plant with a total nitrogen concentration about 4.5 mg/L, which is within the range expected for biological nutrient removal (Asano, Burton *et al.*, 2007 – Table 7-7). Cost-effectiveness was not explored further until more data becomes available for abatement performance.

4.4.3. Uncertainty

Capital and operating costs were based upon a recent review of cost estimates for treatment plants and pipes and pumps. Costs for dual reticulation were not included. The data was considered to be fair (+/- 30%).

The value of water was based upon offsetting potable bulk water supplies and assumes a demand for recycled water and a value of potable water.

Effectiveness was very uncertain due to limited data and problems with process control. The data was considered to be poor (+/- more than 30%).

4.5. Reuse of Effluent for Controlled Irrigation

4.5.1. Capital and Operating Costs

The reuse of wastewater treatment plant effluent for irrigation provides additional wastewater treatment as well as a fertiliser resource. *Sustainable Effluent Irrigated Plantations: An Australian Guideline* (Myers, Bond *et al.*, 1999) (Guideline) notes that no effluent reuse option considered was profitable when the capital and operating costs were considered with the income from the plantations (Myers, Bond *et al.*, 1999 - p15 and Table 7.2). Nonetheless, the Guideline reports that the cost for effluent reuse for plantations was lower than upgrading wastewater treatment plants for nutrient removal (Myers, Bond *et al.*, 1999 – Table 7.8-7.10 and p160). However, some caution is required for this comparison due to the large number of assumptions. For example, the wastewater treatment plant upgrades focussed on phosphorus reductions and relied upon chemical dosing in one stage to achieve phosphorus concentrations of 1 mg/L, two-stage dosing to achieve 0.5 mg/L and two-stage dosing plus filtration to achieve 0.3 mg/L. The ‘desired’ nitrogen concentration was in the range of 10-15 mg/L. In addition, the cost-effectiveness of irrigation with effluent has a number of assumptions and the sensitivity to particular variables such as the availability and cost of land, the distance from the treatment plant to the plantation, yields and price for plantation products.

The WATCOST model developed by the Australian Bureau of Resource Economics (ABARE) was used to develop capital, operating and income from plantations using effluent from wastewater treatment plants. WATCOST was part of the Guideline (Myers, Bond *et al.*, 1999).The model was firstly run to generate the results as presented in the Guideline. The discount rate was then changed to 3% to be consistent with other cost-effectiveness evaluations in this report and the effluent use rate set to a humid coastal climate (as opposed to arid inland climate). A range of irrigation volumes were considered to reflect a range of wastewater treatment plants capacities. This has implications for the area required and gives insight into the practicability of the option in urban areas. There is an economy of scale and plants under 1.4 ML/day (500 ML/year) were reported as having much higher costs per unit of effluent. The economy of scale beyond 1.4 ML/day is a gradual improvement in cost-effectiveness and this relationship was extended beyond the 8.2 ML/day reported (3000 ML/year) (Myers, Bond *et al.*, 1999 – Figure 7.2) to the upper range of treatment plant size to give insight into the land area required as well as the cost-effectiveness. The most cost-effective option in humid climates reported in the Guideline was the eucalypt sawlog plantation with flood irrigation while the least cost-effective was pine pulp wood plantation using a sprinkler system (Myers, Bond *et al.*, 1999 – Table 7.2). These two plantations were used to give an upper and lower range of cost-effectiveness. The costs were presented in 1996 values to retain transparency with the WATCOST tool and then inflated to 2010 values. Details of the assumptions for the costing are provided in the Appendices. Table 21, Table 22 and Table 23 provide a summary of the results. Note that the value per mega litre considers the value of the product harvested.

Table 21: Capital and operating costs and income for a eucalypt sawlog plantation using effluent from wastewater treatment plants in a humid coastal climate.

Effluent Volume (ML per year)	Effluent Volume (ML per day)	Area required (Hectares)	Capital cost ($AUD 1996/hectare)	Operating cost ($AUD 1996/hectare over 15 years with discount of 3%)	Timber plantation income ($AUD 1996/hectare over 15 year cropping period with a discount of 3%)	Present value cost per ML ($AUD1996/ML)	Present value cost per ML ($AUD2010/ML)
183	0.5	19	18085	23051	15021	258	390
365	1	37	15170	19620	15021	197	298
730	2	74	13771	14620	15021	134	203
1825	5	185	13242	13135	15021	113	171
3650	10	370	12642	11968	15021	96	145
7300	20	739	12329	10720	15021	80	121
18250	50	1848	12128	9969	15021	71	107
36500	100	3695	12054	9716	15021	67	101

Table 22: Capital and operating costs and income for a pine pulpwood plantation using effluent from a wastewater treatment plant in a humid coastal climate.

Effluent Volume (ML per year)	Effluent Volume (ML per day)	Area required (Hectares)	Capital cost ($AUD 1996/hectare)	Operating cost ($AUD 1996/hectare over 15 years with discount of 3%)	Timber plantation income ($AUD 1996/hectare over 15 year cropping period with a discount of 3%)	Present value cost per ML ($AUD1996/ML)	Present value cost per ML ($AUD2010/ML)
183	0.5	20	18644	27231	10416	367	555
365	1	40	16449	24017	10416	311	470
730	2	80	15342	19398	10416	249	377
1825	5	201	14928	19294	10416	243	368
3650	10	401	14990	16966	10416	212	321
7300	20	802	14259	15799	10416	197	298
18250	50	2006	14107	15096	10416	188	284
36500	100	4012	14050	14860	10416	185	280

Capital and operating costs were also calculated for effluent irrigation of a 4-year lucerne hay and 1-year sorghum rotation. As noted, the hay crop takes up nutrients which can be regularly removed through harvesting. The sorghum crop uses any build-up of nitrogen in the soil as well as providing a possible disease control break (Myers, Bond *et al.*, 1999). The WATCOST model was run for the lucerne and sorghum in a humid coastal environment and further assumptions are provided in the Appendix. The capital costs appear to be similar to the FILTER (Filtration and Irrigated Cropping for Land Treatment and Effluent Reuse) system for a similar scale system (Coughlan, Gardner *et al.*, 2003 - p28) which was trialled in Gatton, Queensland. Operating costs as well as income from crops were not provided for FILTER and the cost-effectiveness was not further explored in this report. However, if it was assumed that operating costs were approximately equal to income (as appears to be the case for the following plantations with effluent volumes of approximately 5 ML/day) then the capital cost could be coupled together with the estimated load reduction.

Table 23: Capital and operating costs and income for a lucerne hay and sorghum rotation using effluent from a wastewater treatment plant in a humid coastal climate.

Effluent Volume (ML per year)	Effluent Volume (ML per day)	Area required (Hectares)	Capital cost ($AUD 1996/hectare)	Operating cost ($AUD 1996/hectare over 15 years with discount of 3%)	Lucerne and sorghum income ($AUD 1996/hectare over 15 year cropping period with a	Cost per mega litre ($AUD1996/ML)	Cost per hectare per year ($AUD 1996/ha/yr)	Cost per ML ($AUD2010/ML)	Cost per hectare per year ($AUD2010/ha/yr)
183	0.5	14	21627	31876	19081	258	3363	390	5087
365	1	28	17812	27732	19081	198	2581	299	3904
730	2	56	16168	21428	19081	139	1812	210	2741
1825	5	140	15311	19472	19081	117	1525	177	2307
3650	10	279	14528	17960	19081	100	1308	151	1979
7300	20	559	14122	16323	19081	84	1097	127	1659
18250	50	1396	13862	15334	19081	75	980	113	1483
36500	100	2793	13767	15000	19081	71	928	107	1403

4.5.2. Cost-Effectiveness

The report *Agroforestry systems for recycling secondary-treated municipal effluent in the dry tropics of north Queensland* (Dickinson and Cox, 2008) undertook field trials of a plantations of timber and grass species. The annual uptake of nutrients from a particular plantation tree species was reported from literature and compared to the wastewater treatment plant effluent (Table 24). Nitrogen effluent inputs match peak nitrogen requirements but exceed long term nitrogen uptake for the tree species. It was noted that this may cause a risk for nitrogen leaching over the long term. Phosphorus effluent inputs greatly exceed phosphorus uptake. This was not considered a problem in the short term because phosphorus is readily bound to soil particles but could be a risk for phosphorus leaching if the soil becomes saturated over the long term (Dickinson and Cox, 2008 - p28). The long term nutrient uptake for the plant was selected in this study as the sustainable long term (many decades) uptake of nutrients (assuming that nutrient uptakes can be balanced given the wastewater plant effluent characteristics).

Table 24: Nutrient uptake by a eucalypt species (Dickinson and Cox, 2008 - Table D3).

Element	Annual *E. grandis* nutrient uptake*		Quantity in 10 ML Mareeba effluent
	Peak nutrient uptake (age 3-8 years)	Long-term nutrient uptake (> 8 years)	
Nitrogen	121 kg N/Ha/year	80 kg N/Ha/year	120 kg N/Ha/year
Phosphorus	8 kg P/Ha/year	3kg P/Ha/year	60kg P/Ha/year
Potassium	79 kg K/Ha/year	41kg K/Ha/year	110 kg K/Ha/year
Calcium	105 kg Ca/Ha/year	90 kg Ca/Ha/year	220 kg Ca/Ha/year
Magnesium	40 kg Mg/Ha/year	30 kg Mg/Ha/year	110kg Mg/Ha/year

(* Source; Turner & Lambert 2007).

Table 25: Cost-effectiveness for a eucalypt sawlog plantation using treatment plant wastewater in a humid coastal climate.

Effluent Volume (ML per day)	Area required (Hectares)	Present value cost per ML ($AUD2010/ML)	Cost per hectare per year ($AUD2010/ha/yr)	75% Cost allocation to N ($AUD2010/ha/yr)	25% Cost allocation to P ($AUD2010/ha/yr)	Load reduction of N (tonnes/ha/yr)	Load reduction of P (tonnes per hectare/year)	N Cost-effectiveness ($AUD 2010/tonne)	P Cost-effectiveness ($AUD/tonne)
Notes				*#1*	*#1*	*#2*	*#2*		
0.5	19	390	3748	2811	937	0.08	0.003	35142	312370
1	37	298	2940	2205	735	0.08	0.003	27558	244961
2	74	203	1999	1500	500	0.08	0.003	18745	166623
5	185	171	1686	1265	422	0.08	0.003	15807	140511
10	370	145	1432	1074	358	0.08	0.003	13429	119372
20	739	121	1195	897	299	0.08	0.003	11206	99611
50	1848	107	1061	795	265	0.08	0.003	9943	88381
100	3695	101	1001	751	250	0.08	0.003	9385	83424

Notes:
#1. It was assumed that 75% treatment costs were allocated to total nitrogen removal and 25% to total phosphorus removal based upon the same allocation for biological nutrient removal (Pickering and Marsden, 2007 – p41).
#2. Based upon the long term nutrient uptake of *E. grandis* (Dickinson and Cox, 2008 - Table D3).

Table 26: Cost-effectiveness for nutrient removal from a pine pulpwood plantation using treatment plant wastewater in a humid coastal climate.

Effluent Volume (ML per day)	Area required (Hectares)	Present value cost per ML ($AUD2010/ML)	Cost per hectare per year ($AUD2010/ha/yr)	75% Cost allocation to N ($AUD2010/ha/yr)	25% Cost allocation to P ($AUD2010/ha/yr)	Load reduction of N (tonnes/ha/yr)	Load reduction of P (tonnes per hectare/year)	N Cost-effectiveness ($AUD 2010/tonne)	P Cost-effectiveness ($AUD/tonne)
Notes				*#1*	*#1*	*#2*	*#2*		
0.5	20	555	5065	3799	1266	0.08	0.003	47489	422123
1	40	470	4293	3219	1073	0.08	0.003	40243	357712
2	80	377	3437	2578	859	0.08	0.003	32220	286399
5	201	368	3337	2503	834	0.08	0.003	31287	278108
10	401	321	2919	2189	730	0.08	0.003	27364	243234
20	802	298	2712	2034	678	0.08	0.003	25428	226024
50	2006	284	2587	1940	647	0.08	0.003	24254	215591
100	4012	280	2546	1909	636	0.08	0.003	23867	212150

Notes:
#1. It was assumed that 75% treatment costs were allocated to total nitrogen removal and 25% to total phosphorus removal based upon the same allocation for biological nutrient removal (Pickering and Marsden, 2007 – p41).
#2. Based upon the long term nutrient uptake of *E. grandis* (Dickinson and Cox, 2008 - Table D3). Nutrient uptake from *E. grandis* may not be applicable to pine plantations.

Hay such as 'fine cut' Rhodes grass was reported as having much higher nutrient stripping than tree species such as *E. grandis*. With flood irrigation, 'fine cut' Rhodes grass hay was reported as having an uptake of nitrogen of 517 kg/ha and 97 kg/ha phosphorus (Dickinson and Cox, 2008 - p52) for flood irrigation (Table 27). This is an order of magnitude greater than nutrient uptake by timber species such as *E. grandis* (Table 24) and has a significant effect of the cost-effectiveness (Tables 25, 26 and 28). Other grass varieties trialled also had relatively high nutrient uptake as noted in Table 27. The load reduction was coupled together with the costs calculated for lucerne and sorghum irrigation in Table 23 to produce an order or magnitude estimate of cost-effectiveness (Table 28).

Table 27: Nutrient load reduction from grasses irrigated with wastewater effluent (Dickinson and Cox, 2008).

Table E11. Mean total estimates of elements (kg dry matter/ha) removed as hay during seven harvests over 20 months by four varieties of tropical grasses grown on a site flood-irrigated with secondary-stage treated effluent and fertilised for healthy plant growth. Herbage was sampled at three positions in each irrigation bay, 'Upper' closest to the outlet, 'Lower' furthest away.

Experimental treatment factor	Mean total element extracted as hay[1] N	P	K	S	Mg	Ca	Na	Cl
Grass Variety								
Bambatsi	482.1	90.5	542.1 b	75.7 b	75.4	96.8 b	163.0 b	268.7 b
Finecut	516.7	96.9	599.0 b	152.0 a	57.2	216.2 a	415.5 a	801.5 a
Jarra	367.5	75.5	760.2 a	41.1 c	69.8	87.3 b	33.7 c	289.6 b
Tully	394.5	108.3	527.7 b	57.3 bc	76.4	110.2 b	145.4 b	238.4 b
LSD (P<0.05)	119.8#	24.2#	124.7*	23.1***	15.9#	26.0***	52.6***	70.5***
Sample Position								
Upper	491.2 a	105.6 a	657.1 a	90.5 a	77.0	138.8	198.0	440.5
Middle	439.6 ab	101.9 a	643.4 a	87.7 a	68.6	130.0	212.5	399.1
Lower	389.3 b	70.8 b	522.1 b	66.3 b	63.5	114.0	156.3	359.3
LSD (P<0.05)	56.2**	16.1***	89.6*	19.9*	10.9#	NS	NS	NS
Interaction[2]	NS	NS	NS	NS	NS	NS	NS	NS

* significance class 1: F probability 0.01 – 0.05
** significance class 2: F probability 0.001 – 0.009
*** significance class 3: F probability <0.001
\# F probability level 0.05 – 0.1
1 mean of three bay replicates each calculated using dry matter sampled by quadrat method (3 per bay) on the nominated date multiplied by nutrient concentration averaged for 28 October 2005 and 15 October 2006.
2 Interaction of Variety x Sample position

Table 28: Cost-effectiveness for nutrient removal from a hay and sorghum rotation using treatment plant wastewater in a humid coastal climate.

Effluent Volume (ML per day)	Area required (Hectares)	Cost per ML ($AUD2010/ML)	Cost per hectare per year ($AUD2010/ha/yr)	75% Cost allocation to N ($AUD2010/ha/yr)	25% Cost allocation to P ($AUD2010/ha/yr)	Load reduction of N (tonnes/ha/yr)	Load reduction of P (tonnes per hectare/year)	N Cost-effectiveness ($AUD 2010/tonne)	P Cost-effectiveness ($AUD/tonne)
Notes				*#1*	*#1*	*#2*	*#2*		
0.5	14	390	5087	3815	1272	0.517	0.097	7380	13111
1	28	299	3904	2928	976	0.517	0.097	5664	10062
2	56	210	2741	2056	685	0.517	0.097	3976	7064
5	140	177	2307	1730	577	0.517	0.097	3347	5946
10	279	151	1979	1484	495	0.517	0.097	2871	5100
20	559	127	1659	1244	415	0.517	0.097	2407	4276
50	1396	113	1483	1112	371	0.517	0.097	2151	3822
100	2793	107	1403	1053	351	0.517	0.097	2036	3617

Notes:
#1. It was assumed that 75% treatment costs were allocated to total nitrogen removal and 25% to total phosphorus removal based upon the same allocation for biological nutrient removal (Pickering and Marsden, 2007 – p41).
#2. 517 kg/ha and 97 kg/ha phosphorus for 'fine cut' Rhodes grass with flood irrigation in a humid coastal environment (Dickinson and Cox, 2008-p52).

4.5.3. Uncertainty

Capital and operating costs were based upon a model of irrigation using effluent. The data was considered to be fair (+/-30%).

Effectiveness was based on measured trials in the dry tropics of nutrient uptake by particular grass and eucalypt species. The data was considered to be fair (+/30%) but caution is required for application for other areas and plant species.

5. DIFFUSE SOURCE POLLUTANT ABATEMENT

5.1. Bioretention

Cost and performance data from *A Business Case for Best Practice Urban Stormwater Management* (WaterByDesign, 2010) were used to develop a unit cost for pollution reduction using Water Sensitive Urban Design (WSUD) devices. The case studies cover a range of development types including:

Case study 1: Residential greenfield (large scale) on sloping topography

Case study 2: Residential greenfield on flat topography

Case study 3: Residential townhouse development

Case study 4: Urban renewal development

Case study 5: Commercial development

Case study 6: Industrial development

Table 29 and Table 30 provide an example of the calculation process for bioretention pollution abatement using Case study 1: Residential greenfield site (sloping topography) in Brisbane. Only the costs for bioretention (incremental costs) and pollution reduction as a result of the bioretention were considered. The pollution abatement cost calculations were also repeated for the other case studies and are presented as a summary table.

Table 29: Brisbane case study bioretention unit costs and net present value.

	Capital	Operating ($)	Renewal ($)	Decommission at end of life ($)	Total ($)	*Note*
Unit costs ($)	1605900	19695 per year	642360	642360		*1*
NPV ($)	1605900	293012	477976	355659	2732547	*2*

Notes:
#1. WaterByDesign, 2010 - Table 20 presents the capital, maintenance, renewal and decommission costs. The renewal and decommission costs were assumed to be 40% of capital costs (WaterByDesign, 2010 – p17). These costs were referred to as incremental costs and are the additional costs for using bioretention to meet stormwater objectives (WaterByDesign, 2010 – p10).
#2. A 20-year time period assumed with 3% real discount rate. The renewal was assumed to occur at half way through the period with decommissioning at the end of the time period.

In general, the cost for pollution abatement for WSUD was affected by the relatively high capital costs compared to the amount of pollution reduced. For example, for Case study 1, the capital costs were about $1.6 million while the amount of phosphorus and nitrogen reduced per year was about 0.1 and 0.3 tonnes respectively (Table 30).

Table 30: Bioretention pollution abatement costs for a residential greenfield develop in Brisbane.

	Bioretention System	TSS	TP	TN	*Note*
Total Net Present Value ($)	2732547				*1*
WSUD cost driver		0.43	0.32	0.24	*2*
Allocated Net Present Value ($)		1181642	886232	664674	
Reduction (kg/yr)		67860	103.09	313.14	*3*
Reduction over 20 years (kg/20 years)		1357200	2061.8	6262.8	
Pollution abatement cost ($/tonne)		871	429834	106130	

Notes:
#1. See Table 29; #2. See Table 2; #3. (WaterByDesign, 2010 - Table 19) – the reduction attributed to WSUD was unmitigated load minus the basecase reduction.

Over a 20-year period with the discounting of operating costs and the cumulative reduction in pollutants, the cost per tonne of pollution abatement was approximately $0.4 and $0.1 million per tonne respectively for phosphorus and nitrogen (Table 30). Pollution abatement in a residential greenfield developments was more cost-effective than in townhouses. It was also more cost-effective to abate pollution on a sloping rather than flat greenfield development (Table 31).

Table 31: Summary of pollution abatement costs for bioretention by development type in Brisbane.

	Abatement Cost ($/tonne)		
	TSS	**TP**	**TN**
Case study 1 Greenfield residential (sloping topography)	871	429834	106130
Case study 2 Greenfield residential (flat topography)	2250	1058258	255442
Case study 3 Townhouses	11480	5706057	1497703
Case study 4A Urban Renewal	1810	794356	157477
Case study 4B Urban Renewal	2141	867617	149215
Case study 5 Commercial Development	10249	4724846	810707
Case study 6 Industrial Development	4875	2386216	568887

5.1.1. Uncertainty

Capital costs were based upon recent projects in Queensland and the data was considered as good (+/- 15%).

Operating costs were estimated in the reference as a percentage of capital costs due to a noted lack of data. The data was considered to be poor (+/- more than 30%).

Effectiveness was based upon modelled performance of bioretention in particular types of developments. The data was considered good (+/-15%) but some caution is required when applying this in different contexts.

5.2. Stormwater Quality Offsets

The Melbourne Water Land Development Manual outlines the requirements for Land Developers for drainage services(MelbourneWater, 2011). This data provides a comparison to capital costs calculated for bioretention in SEQ. The comparison is only for order of magnitude checks and requires the method and climate factors to be further considered.

In Victoria, the best practice targets aim to retain 80% of the suspended solid annual load, 45% of total phosphorus and 45% of total nitrogen annual loads (VictorianStormwaterCommittee (VSC), 1999). However, the fees were divided into a ‘hydraulic component’ for the conveyancing of stormwater and a ‘water quality component’ for Water Quality Treatment Works to meet the Best Practice Environment Management Objectives (VSC, 1999). The fees were based on a cost recovery basis and vary for locations (which were municipality based) for development in existing residential areas (‘non greenfield’ which was labelled in the table below as brownfield/infill) and were also provided for greenfield developments. Table 32 provides a summary of the cost per hectare for existing and new standard residential developments based upon data accessed in April 2011. The Appendices contain a copy of the data used to calculate the summary. Table 32 suggests that the average water quality stormwater fees were approximately a third of the average hydraulic stormwater fees for greenfield developments. The average water quality fee for greenfield developments was also similar to the average water quality fees for brownfield/infill developments. However, there was a large difference in the standard deviation with much greater variation about the average for greenfield developments. Nonetheless, the large sample size of 183 greenfield developments produced a relatively tight confidence interval. Average water quality fees for greenfield developments were approximately one third of the average hydraulic stormwater fees (and one quarter of the total hydraulic and water quality charge). However, the fees levied upon the developer are only for the construction of the stormwater system and operating and maintenance costs need to be considered separately.

Table 32: Hydraulic and water quality fees for development for Melbourne Water.

Development Type	Current Base Rate (standard residential) ($/ha)	
	Hydraulic	Water Quality
Greenfield		
median	24450	8730
average	27077	9671
standard deviation	18605	7532
Upper 95% confidence	29722	10763
Lower 95% confidence	24431	8580
Brownfield/Infill		
median		8317
average		8520
standard deviation		1876
Upper 95% confidence		9056
Lower 95% confidence		7983

Table 33 presents the capital costs per hectare for Water by Design case studies for comparison. The Water by Design case studies have higher capital costs per hectare. The capital costs are likely to be different due to factors such as different climatic conditions. For example, for the Water by Design Case Study 1, the capital cost for Brisbane was approximately $1.6 million. However, if the climatic conditions were changed to Mackay, then the capital costs increase to $2.4 million (WaterByDesign, 2010 – Table 20). However, there is a large difference between costs for urban renewal and brownfield sites. This difference is unlikely to be due to climatic factors alone and requires further research.

Table 33: Capital cost per hectare for Water by Design case studies.

Case Study	Capital cost per hectare
Case study 1 Greenfield residential (sloping topography)	21130
Case study 2 Greenfield residential (flat topography)	33210
Case study 3 Townhouses	29680
Case study 4A Urban Renewal	36300
Case study 4B Urban Renewal	52800
Case study 5 Commercial Development	51100
Case study 6 Industrial Development	46200

5.3. Swales

The method for calculating the cost-effectiveness of swales was similar to the ‘Simple Cost Based on Drainage Area’ method outlined in the Water Environment Research Foundation Best Management Practice whole of life cost model for stormwater (WERF, 2009). Unit rates for capital and operating were based on literature. Sizing and performance curves were based on SEQ technical guidelines for stormwater and coupled together with catchment pollutant loads for SEQ case studies.

5.3.1. Capital and Operating Costs

Table 34 provides a summary of capital and operating costs based upon unit rates from literature. Note that the costs were expressed for the land area of swale and not the catchment drainage area. Construction costs can vary depending on the amount of earthworks such as clearing, grubbing, levelling, filling, and sodding (USEPA, 1999). Operating costs can vary depending on the amount of establishment and operation required. For example, the first two years requires the most maintenance for weed removal and replanting during establishment (MBWCP, 2006 - p2-27). The estimates were expressed in 2010 values and an average and 95% confidence interval calculated. The large range of the confidence interval suggests that different site preparation and system designs were considered in the sample. Nonetheless, over the 20-year life of the device, the present value of the operation costs was much greater than the capital costs. Design of the system to minimise operating costs is an important consideration for the life cycle cost of swales.

Table 34: Review of capital and operating costs for swales.

	Capital ($AUD2010 /m^2/yr)	Operating ($AUD2010 /m^2/yr)	Capital ($AUD2010 /ha/20yr)	Operating ($AUD2010 /ha/20yr)	Capital and operating ($AUD2010 /ha/20yr)	*Note*
	36	4				*1.*
	12					*2.*
		4				*3.*
	8	5.8				*4.*
	15					*5.*
		3				*6.*
	20	3.3				*7.*
	6					*8.*
	11					*9.*
	38					*10.*
	40					*11.*
average	21	4	205208	593995	799204	
stdev	10.8	2.1				
95% confidence	8.0	2.1				
Upper 95% confidence limit	28	6.0	284938	899689	1184628	
Lower 95% confidence limit	13	1.9	125478	288301	413779	

Note All estimates from the following references were inflated to 2010 values for the Australian dollar.
#1. (DPLG 2010) citing UPRCT (2004)
#2. (Taylor 2005) citing URS (2003)
#3. (Taylor 2005) citing Beecham (2002)
#4. (USEPA 1999) converted to Australian dollars with an exchange rate of 0.9
#5. (USEPA 1999) converted to Australian dollars with an exchange rate of 0.9
#6. Lloyd (2002) - grassed swales
#7. Lloyd (2002) - vegetated swales. This includes the first two years as the WSUD tech guidelines note that this IS the high maintenance period. All costs provided for maintenance were included up to year 6 and then the value for year six used for following years up to year 20. The value presented is the average annual maintenance over 20 years.
#8. (Taylor 2005) citing Fletcher *et al* (2003) for using seed turf
#9. (Taylor 2005) citing Fletcher *et al* (2003) for using rolled turf
#10. (Taylor 2005) citing Leinster (2004)
#11. (Taylor 2005) citing Lane (2004) assumes a 3m width of swale.
Confidence intervals calculated based on capital and operating unit rates and then applied to life cycle costs.

5.3.2. Cost-Effectiveness

The pollutant removal effectiveness was based upon *Water Sensitive Urban Design Technical Design Guidelines for South East Queensland, Version 1*; 2006 (MBWCP, 2006) and a recent study of *An Assessment of Stormwater Treatment Trains for Moreton Bay Regional Council* (van Woerden, 2010). A pareto approach was used to define the application of the swale as a proportion of the catchment and the resulting pollution reduction and is described in detail in the Appendices. Table 35 provides a summary of the proportion of a catchment used for swales to provide the maximum load reduction possible for a swale, to meet compliance and an assumed maximum based upon the pareto approach outlined in the Appendices. Note that the Western Region has a lower rainfall and the area required for the swale is lower.

Table 35: Proportion of a catchment used for swales to meet various load reduction targets.

		Swale top area (as % of Total Catchment Area)			
TOTAL SUSPENDED SOLIDS		Greater Brisbane	North Coast	Western Region	South Coast
Maximum load reduction	95%	2	2	2	2
Compliance reduction for Best Practice	80%	0.2	0.2	0.15	0.2
Assumed maximum cost-effective application	76%	0.15	0.15	0.15	0.15
TOTAL PHOSPHORUS		**Swale top area (as % of Total Catchment Area)**			
Maximum load reduction	67%	2	2	2	2
Compliance reduction for Best Practice	60%	0.5	0.5	0.2	0.5
Assumed maximum cost-effective application	54%	0.2	0.2	0.1	0.2
TOTAL NITROGEN		**Swale top area (as % of Total Catchment Area)**			
Maximum load reduction	18%	5	5	2	5
Compliance reduction for Best Practice	45%				
Assumed maximum cost-effective application	14%	1.5	1.5	0.8	2

Table 36 draws upon Table 34 and Table 35 to express the cost of swales in terms of the catchment drainage area. There was an order of magnitude difference in the cost for the 'assumed maximum cost-effective application' and maximum load reduction possible for TSS and TP.

Table 36: Cost of swales for a hectare of catchment drainage area.

TOTAL SUSPENDED SOLIDS	**Greater Brisbane ($/hectare catchment drainage area)**	**North Coast ($/hectare catchment drainage area)**	**Western Region ($/hectare catchment drainage area)**	**South Coast ($/hectare catchment drainage area)**
Maximum load reduction	6912	6912	6912	6912
Compliance reduction for Best Practice	691	691	518	691
Assumed maximum cost-effective application	518	518	518	518
TOTAL PHOSPHORUS				
Maximum load reduction	5184	5184	5184	5184
Compliance reduction for Best Practice	1296	1296	518	1296
Assumed maximum cost-effective application	518	518	259	518
TOTAL NITROGEN				
Maximum load reduction	9720	9720	3888	9720
Compliance reduction for Best Practice	0	0	0	0
Assumed maximum cost-effective application	2916	2916	1555	3888

Table 37 presents the load reduction by swales. Case study 1 was a sloping greenfield site located in Brisbane with a catchment area of 75.8 hectares (WaterByDesign, 2010 – p21-26). The catchment area and unmitigated loads from the Water by Design case studies were used for the load profile.

Table 37: Load reduction by swales for a Brisbane greenfield development.

UNMITIGATED LOAD			
Brisbane	**Unmitigated per catchment (tonne/yr/ catchment)**	**Unmitigated load over 20 years in catchment (tonne.20 years/catchment)**	**Unmitigated load over 20 years per hectare (tonne.20years/ha)**
TSS	87	1740	22.95515
TPP	0.169	3.38	0.044591
TN	0.921	18.42	0.243008
LOAD REDUCTION			
TSS	**Load reduction (%)**	**TSS Load reduction over 20 years per catchment (tonne.20 years)**	**Load reduction over 20 years per hectare**
Maximum load reduction	0.95	1653	21.8
Compliance reduction for Best Practice	0.8	1392	18.4
Assumed maximum cost-effective application	0.76	1322	17.4
TP	**Load reduction (%)**	**TP Load reduction (tonne.20 years)**	
Maximum load reduction	0.67	2.26	0.0299
Compliance reduction for Best Practice	0.6	2.03	0.0268
Assumed maximum cost-effective application	0.536	1.81	0.0239
TN	**Load reduction (%)**	**TN load reduction (tonne.20 years)**	
Maximum load reduction	0.18	0.608	0.00803
Compliance reduction for Best Practice	0.45	1.52	0.0201
Assumed maximum cost-effective application	0.144	0.487	0.00642

Table 38 provides the cost-effectiveness of pollution abatement with swales for a greenfield development in Brisbane.

Table 38: Cost-effectiveness pollution abatement with swales.

TSS		Cost-effectiveness ($/tonne)
Maximum load reduction	95%	317.0
Compliance reduction for Best Practice	80%	37.6
Assumed maximum cost-effective application	76%	29.7
TOTAL PHOSPHORUS		Greater Brisbane
Maximum load reduction	67%	173518
Compliance reduction for Best Practice	60%	48440
Assumed maximum cost-effective application	54%	21690
TOTAL NITROGEN		Greater Brisbane
Maximum load reduction	18%	1211011
Compliance reduction for Best Practice	45%	
Assumed maximum cost-effective application	14%	454129

5.4. Stormwater Harvesting and Use

5.4.1. Capital and Operating Costs

Constructed wetlands for urban stormwater have the potential to reduce diffuse urban water pollution as well as provide a number of benefits such as a possible water source, habitat, aesthetic and recreation values. The land area required may be a constraint in existing urban developments and may be more feasible in peri urban areas. From the perspective of constructed wetlands for nutrient control, the scale of the wetland appears to be an important determinant of the cost-effectiveness (BDAGroup, 2005 – p53, 65). Figure 14 reproduces the relationship between cost and reuse volume for urban stormwater projects based upon case studies in NSW (DEC, 2006).

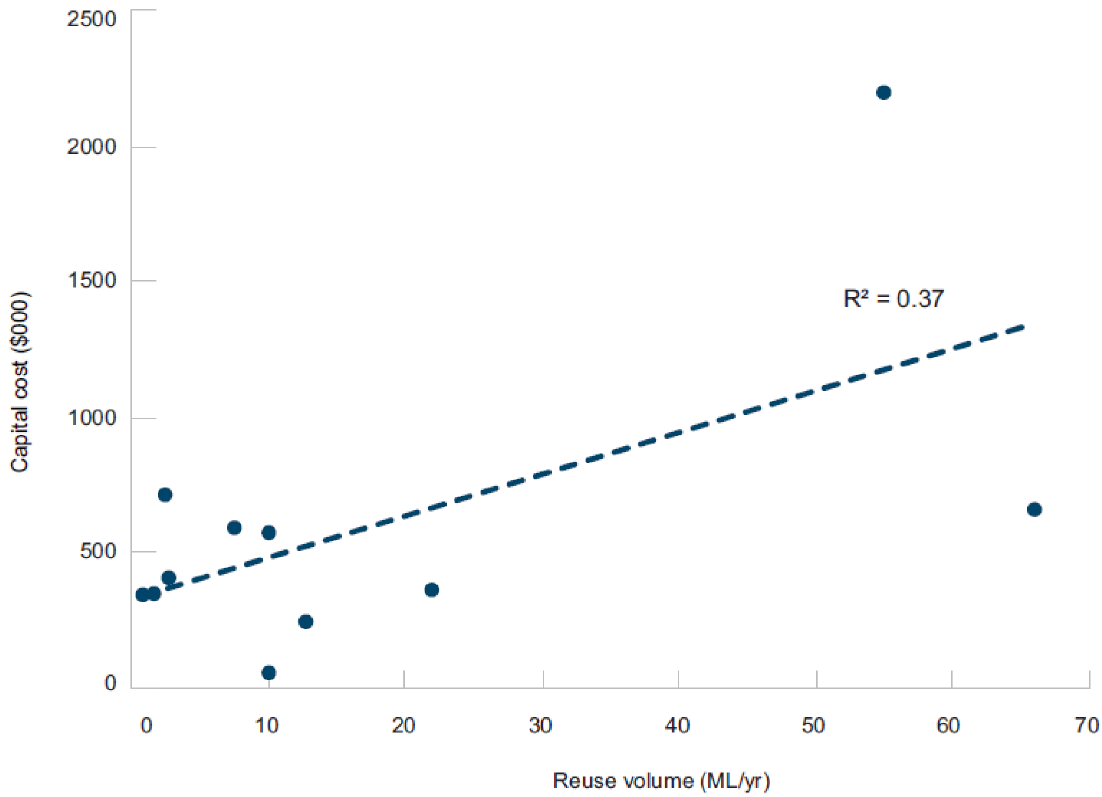

Figure 14: Capital costs for stormwater harvesting and use based on NSW case studies (DEC, 2006 – Fig 8-2).

Table 39 presents the annual operating and maintenance costs relative to capital costs for the series of NSW stormwater harvesting case studies (DEC, 2006). The proportion of annual operating to capital cost ranges from about 3% to 11%, with an average of about 6% and a median of 8% for the case studies. There was also no clear relationship between scale and operating costs. An annual operating to capital cost percentage of 7% was assumed for the cost-effectiveness calculations. This partially considers an outlier effect of the largest capital project given that other estimates in the literature assume a range of approximately 2-6% - (DEC, 2006 – Table D-1).

Table 39: Proportion of annual operating to capital costs for stormwater harvesting.

	Capital cost ($AUD2006)	Operating Cost ($AUD2006)	Percentage of annual operating to capital cost
Riverside Park, Chipping Norton	68000	5700	8%
Black Beach Foreshore Park, Kiama	174900	17000	10%
Hornsby Shire Council Nursery And Parks Depot	329000	28000	9%
Barnwell Park Golf Course, Five Dock	337530	27000	8%
Manly Stormwater Treatment And Use	359780	39000	11%
Powells Creek Reserve, North Strathfield	379183	30000	8%
Solander Park, Erskineville	544798	46000	8%
Scope Creek, Cranebrook	562452	44000	8%
Bexley Municipal Golf Course, Bexley	594197	18000	3%
Sydney Smith Park, Westmead	731827	45000	6%
Taronga Zoo, Mosman	2200000	55000	3%
Average	571061	32245	6%
Standard deviation	573166	14980	3%
Average excluding largest capital project	408167	29970	8%
Standard deviation excluding largest capital project	201770	13640	2%

The value of the water supply from stormwater harvesting was estimated using the price path presented in section 2.1.7. Capital costs show an economy of scale. For example, a 10-fold increase in the size of the scheme from 5 ML to 50 ML only doubles the cost. The cost allocation was also sensitive to the value of the water and the assumed maintenance and operating costs. Although not typical compared to the other case studies presented, the capital and operating of the largest case study combined with the value of water would give pollution abatement at minimal cost.

Table 40: Stormwater harvesting capital and operating costs for pollution abatement.

Stormwater reuse Volume (ML/yr)	Capital cost ($million AUD 2006)	Capital cost ($AUD million 2010)	Operating cost per year at 4% of capital cost ($AUD million 2010)	Operating cost ($AUD million 2010)	Total capital and operation cost excluding value of water ($AUD million 2010)	Present value of water (million $AUD2010)	Total capital and operating including value of water (million $AUD2010)	24% allocation to N (million $AUD2010)	32% allocation to P (million $AUD2010)	44% to TSS (million $AUD2010)
5	0.45	0.51	0.035	0.53	1.03	0.19	0.84	0.20	0.27	0.37
10	0.49	0.55	0.039	0.57	1.13	0.39	0.74	0.18	0.24	0.33
20	0.65	0.73	0.051	0.76	1.49	0.77	0.72	0.17	0.23	0.32
50	1.05	1.18	0.083	1.23	2.41	1.93	0.48	0.12	0.15	0.21

5.4.2. Cost-Effectiveness

The effectiveness of stormwater harvesting for nutrient removal was affected by the concentration of pollutants in the stormwater and the removal efficiency. Table 41 provides a range of concentrations for stormwater constituents. There was an order or magnitude difference between low and high concentrations of phosphorus and nitrogen in stormwater which was captured in the estimate of cost-effectiveness. Table 42 indicates the levels of pollution retention and outflow concentrations for different stormwater treatment measures. The range of nutrient removal was relatively smaller and the average in the range was assumed in the estimate of cost-effectiveness of stormwater harvesting for nitrogen abatement (Table 43) and phosphorus abatement (Table 44).

Table 41: Indicative stormwater, sewage and effluent concentrations (DEC, 2006 - Table C-3).

		Stormwater				
Constituent	**Units**	**Lower**	**Typical**	**Upper**	**Sewage**	**Effluent**
Suspended solids[1]	mg/L	40	140	500	300	n/a
Turbidity[2,3]	NTU	14	60	260		n/a
Total phosphorus[1]	mg/L	0.08	0.25	0.8	12	5.9
Filterable phosphorus[5]	µg/L	18	70	170		
Soluble phosphorus[5, 7]	mg/L	0.0381	0.129	3.52		
Total nitrogen[1]	mg/L	0.7	2	6	55	15.2
Total Kjeldahl nitrogen[5,6]	mg/L	1.73	3.02	4.7		
Ammonia[6]	mg/L	0.15	0.17	0.23		
Nitrate and nitrite[5,6]	mg/L	0.15	0.34	0.34		
Chemical oxygen demand[2,3]	mg/L	35	78	175		n/a
Biochemical oxygen demand[2,3]	mg/L	7	14	26	275	n/a
Total organic carbon[2,3]	mg/L	13	24	40		n/a
Oil and grease[1]	mg/L	3	9.5	30		n/a
pH[2,3]	–	6.3	6.9	7.5		7.9
Total dissolved salts[4]	mg/L	110	160	220		675
Electrical conductivity[4]	dS/m	0.17	0.25	0.34		1.3
Aluminium[7, 8]	mg/L	0.1	1.7	4.9		
Boron[8]	mg/L					289
Cadmium (total)[1]	µg/L	1	4.5	20		0.3
Chloride[7, 9]	mg/L	0.3	2.4	4.5		135
Chromium (total) [2,3]	µg/L	6	20	25		9.4
Copper (total)[1]	µg/L	20	80	300		23.5
Cyanide[7,8]	µg/L	2	33	80		
Iron (total)[2,3]	µg/L	800	2,700	9,000		722
Manganese (total) [2,3]	µg/L	80	230	660		35
Mercury (total)[2,3]	µg/L	0.06	0.22	0.78		0.1
Nickel (total)[2,3]	µg/L	14	24	25		7
Sodium[7, 9]	mg/L	0.18	10.7	21.3		181
Zinc (total)[1]	µg/L	100	300	1,000		48
PAH[7]	µg/L	0.24	0.77	1.3		
MTBE	µg/L		1.6			

Source: stormwater data – 1 Fletcher at al. (2004), 2 Engineers Australia (2005), 3 Duncan (1999), 4 Sharpin (1995), 5 Smullen et al. (1999), 6 SWC (1995), 7 Makepeace et al. (1995), 8 Dannecker et al. (1990). Sewage data – SWC (1998). Effluent data – NRMMC & EPHC (2005)

Note = total dissolved solids (TDS) levels were converted to electrical conductivity using the equation EC (dS/m) x 670 = TDS (mg/L) (ANZECC & ARMCANZ 2000)

PAH: Polycyclic aromatic hydrocarbons

Table 42: Indicative levels of pollution retention and outflow concentrations for different stormwater treatment measures (DEC, 2006 – Table 6-7).

Stormwater treatment measure	Suspended solids	Total phosphorus	Total nitrogen	Turbidity	*E. coli*
Retention					
GPT	0–70%	0–30%	0–15%	0–70%	Negligible
Swale	55–75%	25–35%	5–10%	44–77%	Negligible
Sand filter	60–90%	40–70%	30–50%	55–90%	–25–95% (up to 1.5 log)
Bioretention system	70–90%	50–80%	30–50%	55–90%	–58–90% (up to 1 log)
Pond	50–75%	25–45%	10–20%	35–88%	40–98% (0.5–2 log)
Wetland	50–90%	35–65%	15–30%	10–70%	–5–99% (up to 2 log)
Outflow*					
GPT	42–140	0.18–0.25	1.7–2.0	18–60	9,000
Swale	35–63	0.16–0.18	1.8–1.9	14–34	9,000
Sand filter	14–56	0.08–0.15	1.0–1.4	6–93	500–11,000
Bioretention system	14–42	0.05–0.13	1.0–1.4	6–93	900–15,000
Pond	35–70	0.14–0.19	1.6–1.8	7–81	200–5,000
Wetland	11–67	0.09–0.16	1.4–1.7	19–53	100–9,000

* concentrations in mg/L except for turbidity (NTU) and *E. coli* (cfu/100 mL)

Source of retention data: DEC (2006), Fletcher et al. (2004), Victorian Stormwater Committee (1999).

Table 43: Nitrogen abatement cost-effectiveness for urban stormwater harvesting and use.

Stormwater reuse Volume (ML/yr)	24% allocation to N ($AUD million 2010)	Nitrogen concentration in stormwater (mg/L)			Nitrogen reduction in concentration of 22.5% (mg/L)			Nitrogen load reduction over 20 years (tonnes)			N cost-effectiveness ($AUD million/tonne)		
		low	typical	high	low	typical	high	low	typical	high	low	typical	high
5	0.20	0.7	2	6	0.1575	0.45	1.35	0.01575	0.045	0.135	12.81	4.48	1.49
10	0.18	0.7	2	6	0.1575	0.45	1.35	0.0315	0.09	0.27	5.64	1.97	0.66
20	0.17	0.7	2	6	0.1575	0.45	1.35	0.063	0.18	0.54	2.75	0.96	0.32
50	0.12	0.7	2	6	0.1575	0.45	1.35	0.1575	0.45	1.35	0.73	0.26	0.09

Table 44: Phosphorus abatement cost-effectiveness for urban stormwater harvesting.

Stormwater reuse Volume (ML/yr)	25% allocation to P ($AUD million 2010)	Phosphorus concentration			P Reduction 35-65% (50%)			P load reduction over 20 years (tonnes)			P cost-effectiveness ($AUD millions/tonne)		
		low	typical	high	low	typical	high	low	typical	high	low	typical	high
5	0.27	0.08	0.25	0.8	0.04	0.125	0.4	0.004	0.0125	0.04	67.3	21.53	6.73
10	0.24	0.08	0.25	0.8	0.04	0.125	0.4	0.008	0.025	0.08	29.6	9.47	2.96
20	0.23	0.08	0.25	0.8	0.04	0.125	0.4	0.016	0.05	0.16	14.4	4.61	1.44
50	0.15	0.08	0.25	0.8	0.04	0.125	0.4	0.04	0.125	0.4	3.9	1.23	0.39

The most cost-effective phosphorus abatement for stormwater harvesting was approximately $AUD 0.39 million 2010 per tonne, or $390 per kilogram, for stormwater with a 50 ML/yr reuse and high phosphorus concentration inflows. The least cost-effective phosphorus abatement was $AUD 67 million 2010 per tonne, or $67,000 per kilogram, for stormwater harvesting with a 5 ML/yr reuse and relatively low phosphorus concentration inflows (Table 44). This compares to the source data which estimated the average cost of phosphorus removal from the case studies was $AUD 9,000 in 2006/kg/year, ranging from $AUD 300 to $63,000 in 2006/kg/year (DEC, 2006 - p77). Caution is required for comparisons because of the range of assumptions used in the calculations. Similarly, cost-effectiveness per *kilogram* of nitrogen and phosphorus removed was calculated for urban stormwater projects in Brisbane (BDAGroup, 2005 – Table 23). For example, large regional constructed wetlands had a cost of approximately $3.7 million (AUD2005) per tonne for phosphorus removal and $0.85 million (AUD 2005) per tonne for nitrogen removal.

5.4.3. Uncertainty

The capital costs were based upon a series of projects of various scales. The correlation for the curve fitted to the case study data was low and data was especially limited for larger scale stormwater harvesting options. The data quality was considered poor (more than 30%).

The operating costs were based upon a percentage of capital costs and the data quality was considered as poor (more than +/-30%).

Pollutant removal was based upon a number of studies and low, typical and high concentrations of pollutants in stormwater considered in the estimates. The data accuracy was considered fair (+/- 30%) but the characteristics of the stormwater of a particular area need to be reviewed before applying this data.

5.5. Rainwater Tanks

The primary function of a rainwater tank is to provide an alternative water supply for households. However, rainwater tanks also reduce urban runoff and potentially reduce pollutant loads to waterways. The following analysis explored the pollution abatement provided by rainwater tanks. Important considerations include tank set up, tank yield, value of water and the capital and operation costs of the particular tank configuration.

5.5.1. Capital and Operating Costs

The capital and operating costs were based upon a recent review provided by Water by Design (WaterByDesign, 2010 – Table 12) and are similar to costs presented in earlier reviews in Australia (Marsden Jacob, 2007). However, the average capital costs can vary significantly depending on plumbing and installation requirements (Marsden Jacob, 2007). The average operating costs can also vary significantly depending on tank set-up and the efficiency of the pump (Retamal, Glassmire *et al.*, 2009). Table 45 presents the average costs for rainwater tanks of different sizes and shows that operating costs become less significant relative to capital costs as the size of the tank increases.

Table 45: Rainwater Tank Capital and Operating Costs (WaterByDesign, 2010 – Table 12).

Tank size (kL)	Capital ($AUD2010)	Operating ($AUD2010/year)	Capital and Operating (Net Present Value $AUD2010)
3	2500	90	3839
5	3000	90	4339
9	9000	175	11604
21	6000	175	8604
32	20000	300	24463

Case study 1 in the Water by Design case studies contained 951 rainwater tanks with a 5 kL capacity. Each tank was designed to meet Queensland Development Code (QDC) MP 4.2 Water Savings Targets which requires a 70 kL/year alternative water source (non reticulated potable water) per household. This provided a yield of 66 570 kL per year from the 951 rainwater tanks. However, there is some uncertainty whether a rainwater tank by itself can meet the QDC MP 4.2 water savings target. A recent report suggested that the average yield was approximately 40-50 kL/year for an internally plumbed 5 kL rainwater tank (Beal, Gardner *et al.*, 2011). There was also significant variation in the yield, with the range in average values of 20-95 kL/household/year (kL/hh/yr) and a mean of 50 kL/hh/yr and a range in median values 28-52 kL/hh/yr and a mean of 40 kL/hh/yr. A number of factors were noted for the lower yield including lower overall demand for new houses with efficient fixtures and water restrictions that may have reduced external use. Table 46 illustrates the value of water from the rainwater tank for Water By Design Case study 1, assuming a yield of 70 kL/hh/yr. The value of the water from rainwater tanks was estimated by considering the water supply that was offset by the rainwater tanks; for more details refer to section 2.1.7.

Table 46: Value of rainwater tank water from a case study development

Yield from a 5kL tank (kL/year)	Present Value of Water per Tank Over 20 years ($AUD2010/tank)	Present Value for 951 Tanks Over 20 years (Million$AUD2010)
30	1159	1.1
50	1931	1.8
70	2703	2.6

The capital and operating costs for the residential greenfield rainwater tanks are presented in Table 47 and coupled with the value of water. The change in tank yield has a significant effect on the value of present value of the tanks in a development. The costs were then allocated to each pollutant based upon WSUD stormwater objectives (for more information, refer to section 2.1.7*)*. Rainwater tanks may also reduce the size of stormwater infrastructure in a greenfield residential site. This was estimated to be a cost saving in the order of $0.30/kL to $1.00/kL although only one reference was cited and it was noted that the information was limited (Marsden Jacob, 2007 –p xii). However, this cost saving was not included because of the lack of data and potential double counting with other WSUD devices.

Table 47: Allocated capital and operating costs for a development using 5 kL rainwater tanks.

Yield Scenario (kL/year)	Number of 5 kL rainwater tanks	Capital (MILLION $AUD2010)	Operating Costs (MILLION $AUD2010/year)	Present Value not including water (MILLION $AUD2010/ 20 years)	Present Value including water (MILLION $AUD2010 /20years)	Cost Allocation to Pollutants (MILLION $AUD2010/20years)		
						TSS	TP	TN
30	951	2.85	0.09	4.13	3.02	1.30	0.97	0.73
50	951	2.85	0.09	4.13	2.29	0.98	0.73	0.55
70	951	2.85	0.09	4.13	1.56	0.67	0.50	0.37

5.5.2. Cost-Effectiveness

The following table presents the pollution abatement from rainwater tanks based upon the Water by Design Case study 1. It was assumed that the reduced yield would have a proportionate effect on the pollution mitigation.

Table 48 illustrates a five-fold difference between upper and lower estimates. A change in tank yield has a multiplier effect on cost-effectiveness because it changes both the 'income' from water from the tank as well as the amount of pollution abated.

Table 48: Cost-effectiveness for pollution reduction from a 5kL rainwater tank.

Yield Scenario for a 5kL Tank		Load Reduction (tonne/20 years)			Cost-effectiveness (Million $AUD2010/tonne)		
		TSS	TP	TN	TSS	TP	TN
	Unmitigated (kg/yr)	87000	169	921			
70kL	Average annual load (kg/yr)	84960	159	802			
	Reduction (kg/yr)	2040	10	119			
	Reduction over 20 years (tonnes/20 years)	40.8	0.2	2.38	0.016	2.49	0.16
50kL		29.14	0.14	1.7	0.034	5.13	0.32
30kL		17.49	0.09	1.02	0.074	11.3	0.71

5.5.3. Uncertainty

The capital cost data for rainwater tanks is based upon a review of market price for a relatively common product and was considered as good (+/-15%).

Operating costs had a large range due to different possible tank set-ups and demands. The variation was captured with a range and was considered fair (+/- 30%). In addition the tank yield was uncertain which affects the value of water gained from the tank and the net present value.

Effectiveness was based upon modelling for a case study. However, the uncertainty in the yield has about a five fold effect on the cost-effectiveness due to changes in both income and pollution abated. This was captured with the range in values and the data was considered fair (+/- 30%).

5.6. Riparian Revegetation – Planting and Fencing

5.6.1. Capital and Operating Costs

The capital and operating costs for riparian revegetation appear to reflect the scope of work and considerations rather than the unit costs for planting or fencing. Riparian 'revegetation' can simply include planting new vegetation using on-site labour or can include fencing, stock crossings, off-stream watering, bank stabilisation works and opportunity costs for lost production from farms. The cost was also reported as being significantly different for grass as opposed to vegetated riparian zones. For example, a recent SEQ study (Rolfe, Donaghy *et al.*, 2005 - p51) reviewed the costs for abatement of diffuse pollution from agriculture. It was estimated that grassed riparian filter strips with fencing and off-stream watering points had capital costs of $5000/km and $200/km maintenance. Riparian rehabilitation strips were much more expensive. In a review of a number of studies, it was estimated that the cost for riparian rehabilitation was $25 000/km inclusive of maintenance costs for three years. Higher costs of $45 000/km were noted where 'some big bank stabilisation projects were constructed' for coastal Moreton Bay. It was estimated that approximately 5000 km of grassed riparian strips and 2700 km of riparian rehabilitation were required in SEQ. It was also noted that the average for all of SEQ from Moreton Bay Waterways and Catchment Partnerships was $20 000/km inclusive of revegetation, weed and erosion control, fencing and maintenance. Table 49 presents the capital and operating costs as well as opportunity costs for riparian revegetation from a number of SEQ references as well as a study for the Hawkesbury Nepean catchment in NSW and a study from the Myponga Catchment in SA. Data from the original studies were expressed in 2010 dollar terms. Unless noted in the original study, the width of the riparian strip was assumed to be 20 m based upon the average width of riparian buffer strips in a series of restoration works undertaken in the Myponga Catchment (Bradford, Finney *et al.*, 2008). The width of the buffer strip can vary from about 10-30 m and is important for translating costs into area rates.

Table 49: Review of riparian revegetation costs.

Reference	Capital ($AUD2010/ha)	Operating ($AUD2010/ha)	Present Value ($AUD ha/20 years)	Total Capital and Operating Costs ($AUD/ha/20year)	Note
(Kandulu and Bryan, 2009)	8858				Included planting and fencing. The SA study that also notes annual farm maintenance costs of $309, equivalent to about 5% of capital costs.
(Olley, Saxton *et al.*)	8742				Authors estimate for order of magnitude calculation for catchment-wide SEQ program budget.
(Rolfe, Donaghy *et al.*, 2005 – p51)	14491				Includes planting and fencing Based upon Moreton Bay Waterways and Catchment Partnership data, Mary River data and includes the first three years of maintenance. A 20m wide riparian vegetation strip was assumed based upon (Bradford *et al.*, 2008).
(Cuddy, Marston *et al.*, 1994 – p7)	10495	1845			Capital includes fencing, gates and stock watering. The NSW study notes uncertainty of approximately +/- 30% for capital and capital includes tanks, pipes and pumps and backhoe hire. Operation includes replacement of fencing every 5 years due to flooding and about $320/ha/yr (AUD2010) for weeds and pests.
Average	10647				
SD	2685				
Assumed Value	10647	498	7409	18056	The average capital cost was assumed. An annual maintenance cost of 5% of capital cost was assumed based upon maintenance costs from (Kandulu and Bryan, 2009) and (Cuddy *et al.*, 1994) and mostly covers weed and pest maintenance. Costs are similar to annual maintenance reported for bioretention systems (DEC, 2006 – Table D-1). The opportunity costs from Lockyer Valley grazing was assumed.

The cost of sediment abatement also varies depending on the type of agriculture and the anticipated effect on production. For example, it was estimated that total suspended solids could be reduced by up to 35% at no additional cost for sugar cane producers near the Great Barrier Reef. The reductions were achieved mainly by soil conservation practices such as reduced and zero tillage. However, reductions in total suspended solids from the grazing industry were calculated to have a significant cost to the industry. In this case the reductions were achieved by a combination of reduction in grazing areas with reduced stocking rates (Roebeling, van Grieken *et al.*, 2009).

The opportunity cost was calculated for grazing based upon the gross margin/hectare multiplied by the area taken out of production for the buffer zone. For example, if it was assumed that farm income from grazing was about $100/ha ($AUD2010), then the opportunity cost adds approximately 5-10% to the cost for sediment load reductions. This is a relatively small addition to the cost of pollution abatement but may be significant for farm income. However, it has also been argued that the cost of establishing off-stream drinking sites associated with riparian revegetation is offset by increased sale price of cattle due to cleaner water for stock, reduced disease and increases weight gain by cattle (Kandulu and Bryan, 2009). Figure 15 illustrates that land use associated with grazing was identified as the main source of rural diffuse sediment pollution in SEQ (Attwater, Booth *et al.*, 2002; SEQHWP, 2007).

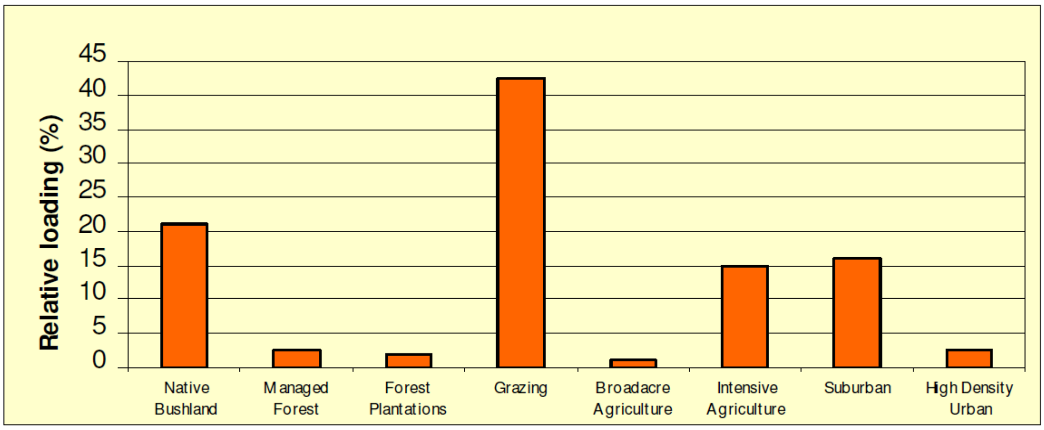

Figure 15: Relative suspended solids loadings to Moreton Bay by land use type.

5.6.2. Cost-Effectiveness for Sediment Reductions

The cost-effectiveness of reducing sediments is affected by the selection of the location and the estimated load reduction. It was estimated that approximately 80% of the sediment delivered to Moreton Bay originates from less than 20% of the catchment area (Olley Aug 27 - Sept 1, 2007).

Figure 16 illustrates a pareto relationship where a relatively small proportion of river revegetation results in a large reduction in sediment export to Moreton Bay. Conversely, to achieve further reductions requires increasingly large amounts of river restoration. Consequently, one of the most important variables for the cost-effectiveness of sediment abatement is the identification of areas to revegetate. Figure 17 depicts the predicted areas within the catchment that are feeding sediment to Moreton Bay. The total load of sediment that was delivered from the Brisbane River to Moreton Bay was estimated at 280 000 tonnes/year.

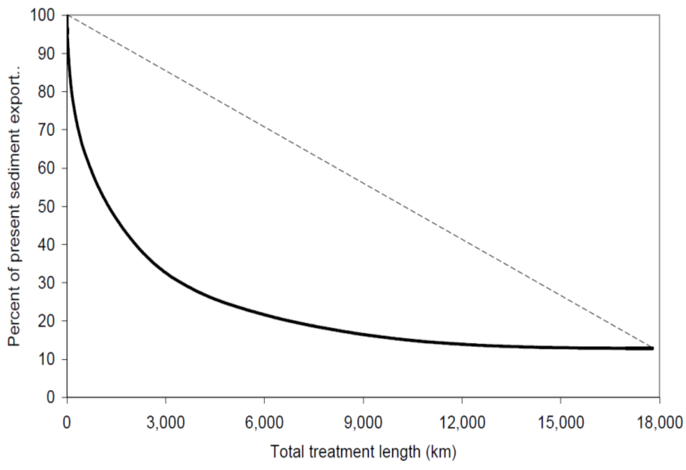

Figure 16: Large reduction in sediment load by relatively small revegetation (Olley Aug 27-Sept 1, 2007).

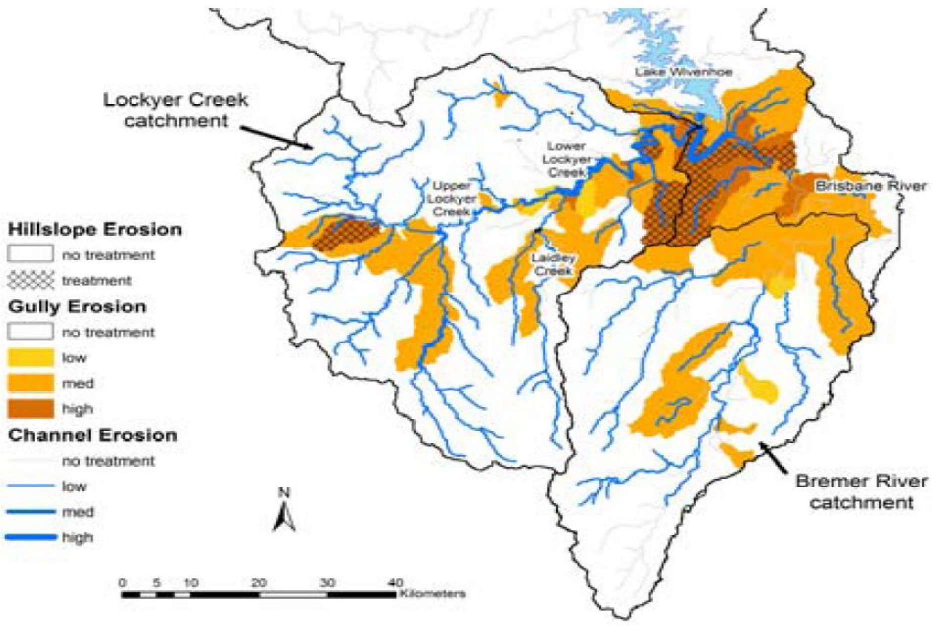

Figure 17: Catchment modelling of sediment sources to Moreton Bay (Olley, Aug 27 - Sept 1, 2007).

Figure 16 suggests that an 80% reduction in sediment requires approximately 6,700km of river revegetation. Assuming the costs presented in Table 49, this gives a cost-effectiveness of approximately $AUD 84/tonne in 2010. This was similar to one of the main SEQ studies considered, which assumed 7,700km of river revegetation with a cost-effectiveness of $AUD64/tonne in 2010 ($54/tonne in 2004) (Rolfe, Donaghy *et al.*, 2005 – Table 3.20). However, the cost-effectiveness for achieving a 50% reduction in sediment load is almost a third of the unit cost for an 80% reduction in sediment load. This suggests that site selection is very important for ensuring that the lowest cost abatement measures are selected first. Table 50 presents the cost-effectiveness for sediment load reduction targets assuming that the most cost-effective sites are prioritised. Detailed studies as part of the *Healthy Country* (Olley, Saxton *et al.*) program are required to identify priority areas and further define the scope of possible sediment reductions as well as accurate maintenance costs.

Table 50: Revegetation cost-effectiveness by sediment load reduction target.

Reduction in current sediment load (%)	Revegetation required (km)	Revegetation (ha)	Sediment reduction over 20 years (tonnes)	Present Value Cost for Riparian Vegetation ($AUD2010/ha/20 years)	Cost for revegetation ($AUD million 2010/area of riparian revegetation to meet target)	Cost-effectiveness over 20 years ($AUD 2010/tonne)
25%	300	900	1400000	18809	17	12
50%	1300	3900	2800000	18809	73	26
70%	3500	10500	3920000	18809	197	50
80%	6700	20100	4480000	18809	378	84
88%	18000	54000	4928000	18809	1016	206

5.6.3. Uncertainty

Capital costs were based upon a number of studies for revegetation and were considered as fair (+/- 30%).

Operating costs were based upon maintenance costs as a percentage of capital costs and were considered to be poor (+/- more than 30%).

Catchment sediment control effectiveness was based upon a strategic assessment of rehabilitation required in SEQ and was considered as fair (+/30%). The actual load reduced and the cost-effectiveness of abatement will depend upon site selection of high erosion areas.

5.6.4. Cost-Effectiveness for Nutrient Abatement

It was assumed that the locations for rural diffuse nutrient abatement would focus on land use types with high nutrient concentrations and loads. These locations may be different to areas targeted for the most cost-effective abatement of rural diffuse sediment load. However, grazing land was also the main source of rural diffuse nutrient pollution in SEQ as illustrated by Figure 18 and Figure 19 (James, 1994). Figure 15 also shows the importance of grazing for suspended solids.

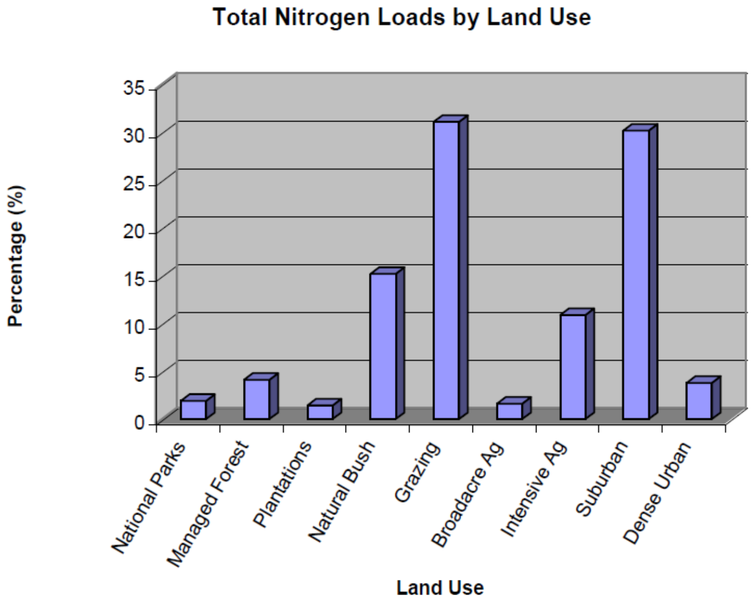

Figure 18: Diffuse Total Nitrogen loads by land use in SEQ (James, 1994).

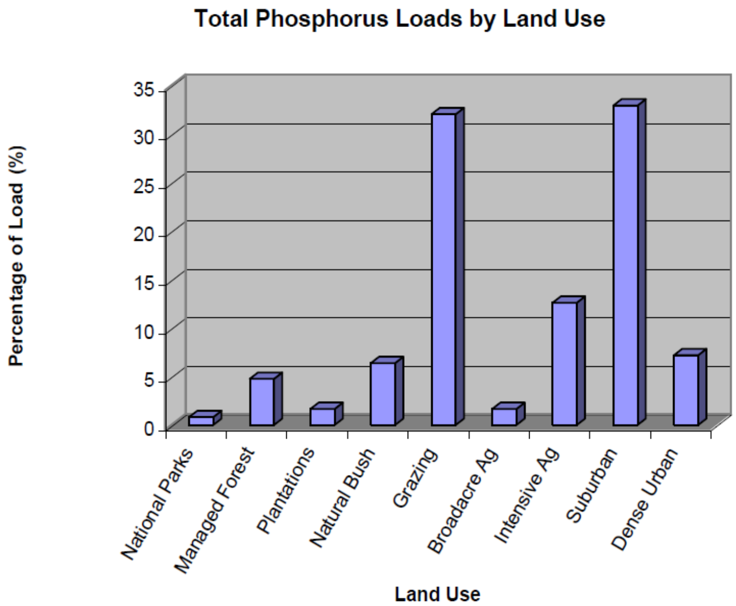

Figure 19: Diffuse Total Phosphorus loads by land Use in SEQ (James, 1994).

The following order of magnitude estimate attempts to make each component of the calculation transparent. The estimate used rainfall and runoff assumptions and runoff concentrations to calculate a load per unit area of agricultural land. The nutrient generation rates were cross checked with reported rates. A load reduction was then calculated for abatement measures using load reduction percentages from the literature. The load reduction was then coupled with unit costs to calculate the cost-effectiveness.

A mean annual rainfall of 1 m was assumed to reflect the rainfall associated with western catchments in SEQ where most of the grazing occurs (SEQHWP, 2007). A mean annual rainfall of 1 m gives a mean annual rainfall volume of 10 000m^3/hectare. Rainfall runoff fractions vary according to catchment characteristics but typical values by surface type range from about 12% for open space and 18% for lawns through to 80% for paving and 90% for roads (EngineersAustralia, 2006). Assuming a rainfall runoff fraction of 12% gives a mean runoff volume of 1 200 m^3 per hectare which is equivalent to 1.2 megalitres (1 200 m^3 per hectare x 1000 litres/m^3).

Table 51 shows the pollutant EMC (Event Mean Concentration) for a number of agricultural land uses from the WBM WaterCast catchment model (BDAGroup, 2005). Table 52 provides pollutant removal efficiencies based upon the report *Diffuse Source Best Management Practices: Review of Efficacy and Costs* (WBM, 2005) as well as (Cuddy, Marston *et al*., 1994). It was noted in a number of references that there was considerable variability in pollutant removal efficiencies with a range of at least +/- 20% (WBM, 2005). The pollutant removal efficiencies can also be affected by factors such as the location in the landscape (need to be located in depressions where flows concentrate), the vegetation type and whether cattle are excluded (BDAGroup, 2007 - Table 7.2).

Table 51: Event Mean Concentrations for grazing, broadacre and intensive agriculture from the WBM WaterCast Model.

	TSS EMC (mg/L)	TN EMC (mg/L)	TP EMC (mg/L)
Grazing	260	2.08	0.3
Broadacre agriculture	300	1.95	0.321
Intensive agriculture	550	5.2	0.449

Table 52: Pollutant removal efficiency for riparian revegetation.

TSS	TN	TP
0.9	0.7	0.8

Table 53 presents a hypothetical grazing, broad acre and intensive agriculture farm in SEQ. The load estimates were similar to estimates based upon nutrient generation rates from the *Nutrient Data Book* for the CMSS model (Marston, Young *et al*., 1995). For example, TP generation rates for grazing and pasture were generally in the range of 0.1-0.4 kg/ha/year which is equal to 2-8 tonne/1000 ha/20 years. It was assumed that approximately 5% of the farm area was required for revegetation and the results are sensitive to this assumption. Table 54 presents the cost-effectiveness for sediment and nutrient abatement using riparian revegetation. The cost-effectiveness calculated for Total Suspended Solids and was found to be similar to the estimate for abatement sites focussed only on sediment control. This was partly due to the allocation of costs to the various pollutants.

Table 53: Load and load reduction estimates by farm type in SEQ.

Farm Type	Farm Area (ha)	Area of Revegetation (ha)	TSS Load (t/farm/20yr)	TN Load (t/farm/20yr)	TP Load (t/farm/20yr)	TSS Load Reduction (t/farm/20ye	TN Load Reduction (tonne/farm/ 20 years)	TP Load Reduction (tonne/farm/ 20 years)
Grazing	1000	50	6240	50	7	5616	35	5.8
Broad Acre Agriculture	1000	50	7200	47	8	6480	33	6.2
Intensive Agriculture	1000	50	13200	125	11	11880	87	8.6

Table 54: Cost-effectiveness for sediment and nutrient abatement using riparian revegetation.

Farm	Present Value# ($)	Allocation of costs ($)*			Cost-effectiveness for Abatement ($/tonne)		
		TSS	**TN**	**TP**	**TSS**	**TN**	**TP**
Grazing	928297	399168	222791	297055	71	6376	51572
Broad Acre Agriculture	928297	399168	222791	297055	62	6801	48198
Intensive Agriculture	928297	399168	222791	297055	34	2550	34458

Present value was based upon unit rates in Table 49 and land area for revegetation in Table 53
*Present value was allocated between the pollutants based upon the weightings of 0.43, 0.24 and 0.32 for TSS, TN and TP respectively

5.6.5. Uncertainty

Capital costs were based upon a number of studies for revegetation and were considered as fair (+/-30%).

Operating costs were based upon maintenance costs as a percentage of capital costs and were considered to be poor (+/- more than 30%).

Catchment nutrient control cost-effectiveness was based upon an approximate calculation of loads for grazing in western catchment of SEQ and a general pollutant removal efficiency from the literature. It was also assumed that approximately 5% of the farm area was required for revegetation. The combination of these factors produces an order of magnitude estimate and the data quality is 'poor' (more than +/30%).

5.7. Efficient Application of Fertilizers in Agriculture

Efficient application of fertilizer has the potential to provide cost savings to farmers as well as reducing nutrient pollution of waterways. The cost (or cost saving) depends upon the type of agriculture and the current rates of application. For example, with reference to sugar cane production and dissolved inorganic nitrogen pollution delivered to the Great Barrier Reef, it was estimated that reductions of up to 50% of dissolved inorganic nitrogen could be achieved at no cost to the industry (Roebeling, van Grieken *et al.*, 2009 - p1155). However, the potential cost saving is dependent on the type of agriculture. For example, the same study found that reductions in dissolved inorganic nitrogen pollution would incur a large cost to the grazing industry.

Tools have also been developed to assist producers manage phosphorus application in parts of Australia (Simpson, Graham *et al.*, 2009). An indicative cost for phosphate fertilizer of $400 per tonne (Simpson, Graham *et al.*, 2009) suggests potential cost savings for efficient use of fertilizers as a means to reduce pollution. Transaction costs for implementing management programs also need to be considered. These costs may vary depending on the policy approach and may be particularly important for land holders. The US EPA *National Management Measures to Control Nonpoint Source Pollution from Agriculture* provides further information on the factors for effectiveness and cost (USEPA, 2003).

6. SUMMARY POLLUTION ABATEMENT COST CURVES

Figures 20, 21 and 22 provide a summary of the cost-effectiveness data reviewed in this study for nitrogen, Total Phosphorus and Total Suspended Solids abatement, respectively. The figures do not capture the amount of pollution abatement available for each abatement option and the process for developing and applying the data to a particular catchment is outlined in Section 9. The range of options was not exhaustive but does capture a number of the main pollution abatement options considered in SEQ for water quality. Note that a log scale was used because the costs increase exponentially from the least to the most expensive abatement option. Differences between options on a log scale (order of magnitude comparisons) are also likely to be greater than the uncertainty in the data. The range in cost-effectiveness for each abatement option indicates the variation from factors such as scale and pollution abatement effectiveness. This range may account for the conflicting comparisons of cost-effectiveness for abatement options in the literature. For example, tertiary filtration is more cost-effective than stormwater harvesting for nitrogen abatement for most scales and abatement effectiveness. However, the reverse is true if the most cost-effective stormwater harvesting is compared with the least cost-effective tertiary filtration. This emphasises the important of considering the context for applying general cost-effectiveness data.

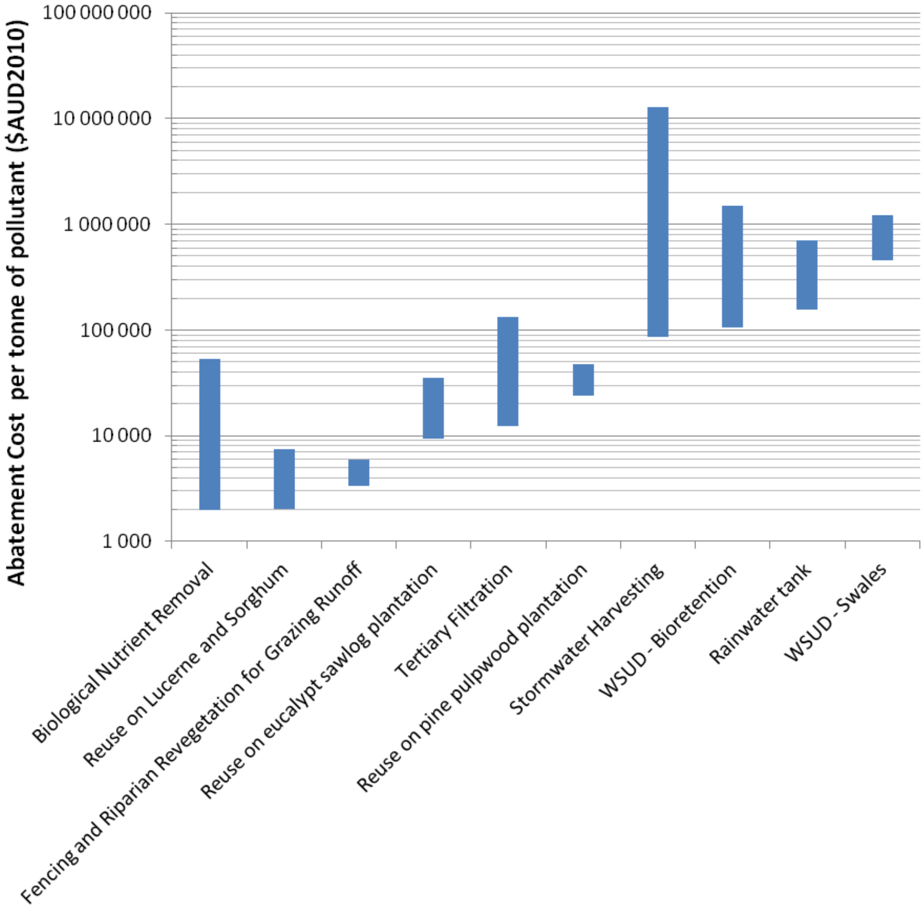

Figure 20: Review of high and low cost estimates for nitrogen abatement (with a 3% discount rate and value of water based upon the bulk water price path).

The application of the data also requires a review of cost allocations, the assumed value of water, environmental equivalences for pollutants and assumptions for the discount rate and period of analysis. Understanding the load profile of a catchment is also important. Some pollutant sources may dominate the load in the catchment and it may not be possible to 'trade' a costly abatement measure for a cheaper one to achieve a pollution reduction target.

Nonetheless, some abatement options are many orders of magnitude more cost-effective than others. For example, nitrogen abatement from large scale Biological Nutrient Removal (which treats most SEQ effluent), reuse of effluent for plantations and riparian revegetation and fencing on grazing land were orders of magnitude more cost-effective than swales, rainwater tanks, bioretention and stormwater harvesting (Figure 20). This conclusion is unlikely to change by considering application in SEQ in more detail.

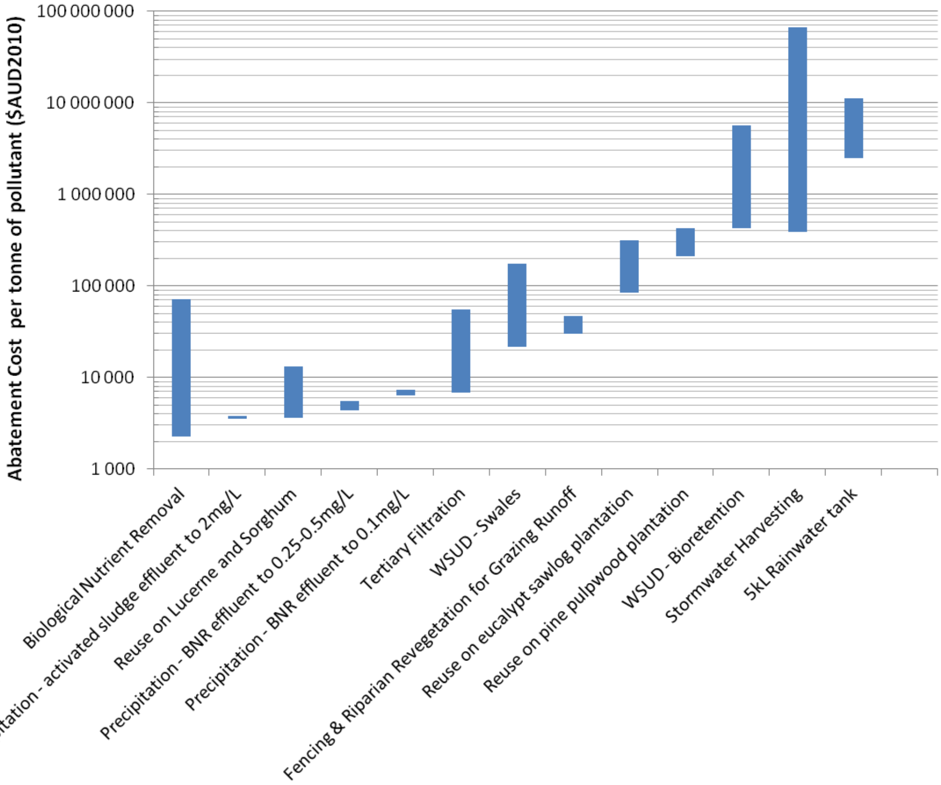

Figure 21: Review of high and low cost estimates for Total Phosphorus abatement (with a 3% discount rate and value of water based upon the bulk water price path).

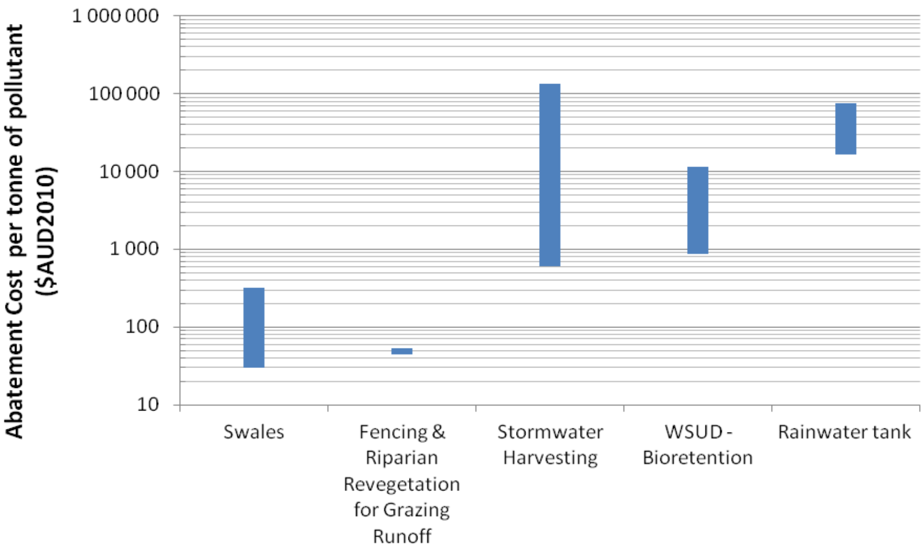

Figure 22: Review of high and low cost estimates for Total Suspended Solids abatement (with a 3% discount rate and value of water based upon the bulk water price path).

7. SENSITIVITY

The sensitivity of the results was explored for a change in the discount rate. The higher discount rate affects the operating costs and changes the cost-effectiveness of the abatement options. A higher discount rate improves the cost-effectiveness of abatement options with relatively high operating costs. For nitrogen abatement, this includes options such as BNR, bioretention and riparian fencing and revegetation. However, the higher discount rate makes the cost-effectiveness worse for options that generate an income over the period of analysis. This includes income from harvesting plantations used for reuse of effluent as well as options that also produce water such as AWTP, stormwater harvesting and rainwater tanks. Table 55 provides the percentage change in cost-effectiveness for nitrogen abatement and a brief explanation from increasing the discount rate from 3 to 5.5%.

Table 55: Percentage difference in cost-effectiveness for nitrogen abatement by increasing the discount rate from 3% to 5.5%.

	Percentage change in low range estimate	Percentage change in high range estimate	Comment
Biological Nutrient Removal	-23%	-23%	Reduces operating cost and becomes more cost-effective
Reuse on Lucerne and Sorghum	21%	12%	Increases cost because income achieved from harvest is further discounted
Fencing and Riparian Revegetation for Grazing Runoff	-3%	-3%	Small reduction because operating costs relatively small
Reuse on eucalypt sawlog plantation	34%	16%	Increases cost because income achieved from harvest is further discounted
Tertiary Filtration	-9%	-8%	Reduces operating cost and becomes more cost-effective
Reuse on pine pulpwood plantation	14%	11%	Increases cost because income achieved from harvest is further discounted
WSUD - Bioretention	-12%	-56%	Reduces operating cost and becomes more cost-effective especially for small systems which had relatively high operating costs.
Stormwater Harvesting	49%	-4%	Small reduction for the high cost estimate because income from water supply was relatively small for small scale systems. Large increase in cost for bigger systems because the value of the water was relatively important and was reduced by discounting.
5kL Rainwater tank	14%	-1%	Small reduction in cost for low yield estimate because the income from water supply relatively small. Moderate increase for higher yield estimate because the value of the water was relatively more important and was reduced by discounting.
WSUD - Swales	-17%	-17%	Reduction in operating costs from higher discount rate.

Figure 23 and Figure 24 provide the cost-effectiveness of nitrogen and phosphorous abatement, respectively, for a 5.5% discount rate. Although there are some large percentage changes in the ranges for the abatement options, there is only minor change to the ranking of options. In most cases, the increase in the discount rate increases the differences already observed for the most and least cost-effective abatement options. Minor changes in ranking include tertiary filtration becoming more cost-effective than reuse of effluent on eucalypt plantations and bioretention becoming more cost-effective than stormwater harvesting. Figure 24 and Figure 25 illustrates similar changes for phosphorus and total suspended solids abatement in response to the change in the discount rate.

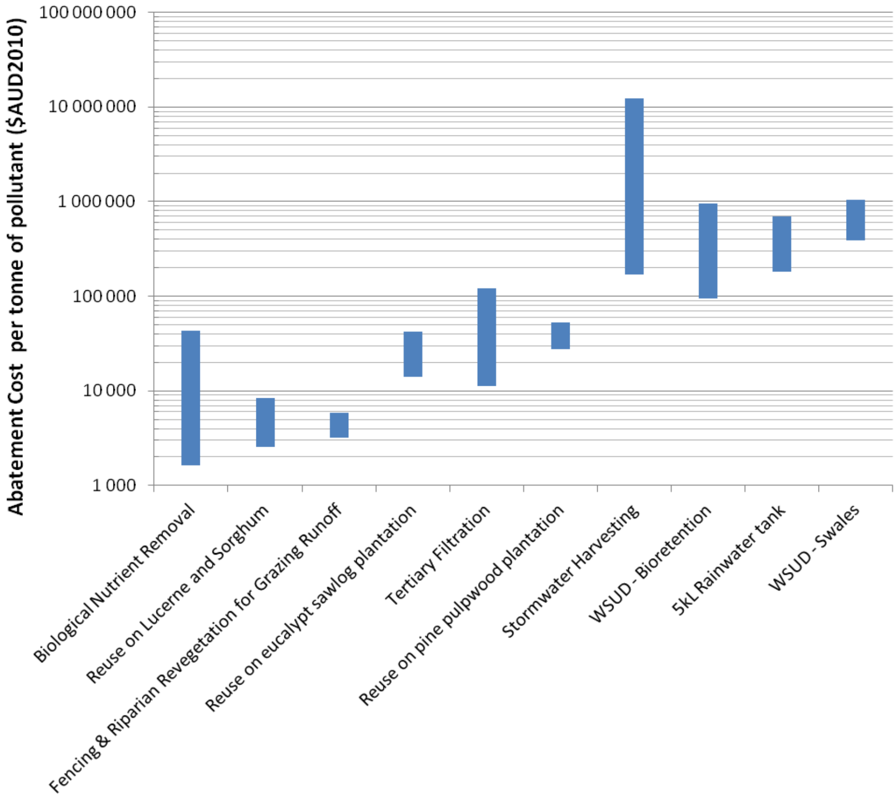

Figure 23: Review of high and low cost estimates for nitrogen abatement (with a 5.5% discount rate and value of water based upon the bulk water price path).

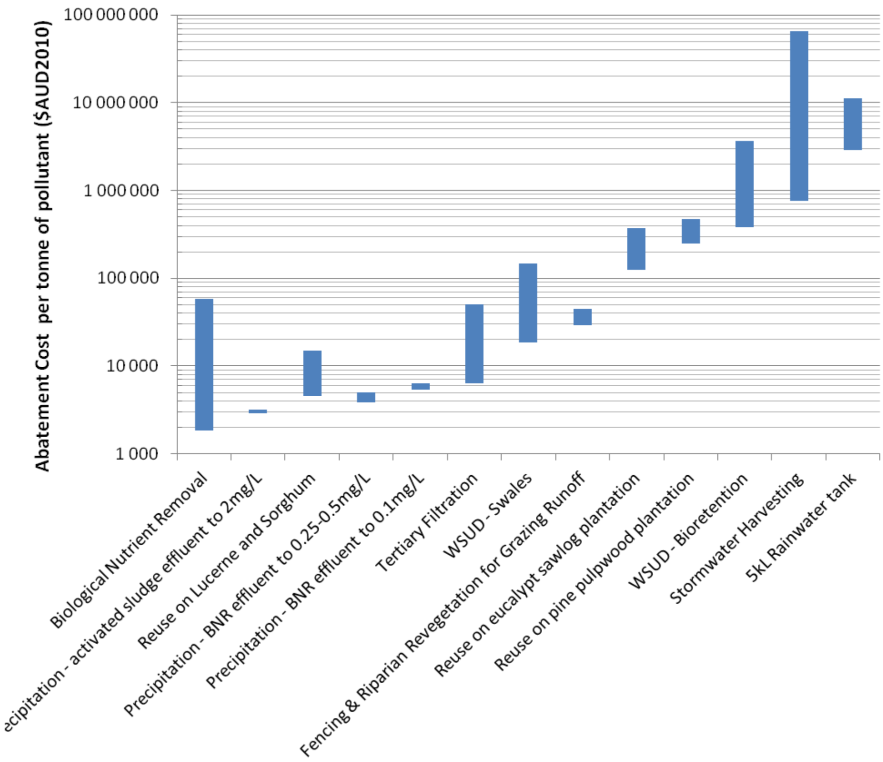

Figure 24: Review of high and low cost estimates for phosphorus abatement (with a 5.5% discount rate and value of water based upon the bulk water price path).

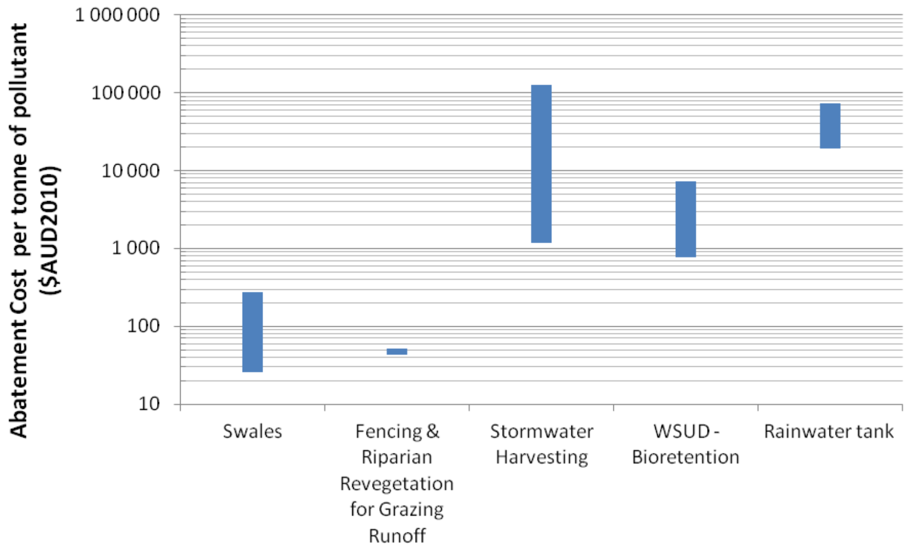

Figure 25: Review of high and low cost estimates for total suspended solids abatement (with a 5.5% discount rate and value of water based upon the bulk water price path).

8. APPLICATION TO TOTAL WATER CYCLE MANAGEMENT PLANNING

The following points provide a process for applying the research to incorporate externalities into TWCMP.

8.1. Check Method

The value of pollution is based upon the cost of abatement options and the amount of pollution to be mitigated. This is similar to the cost of water provision where the value of urban water largely reflects the cost of production. As demand increases, and water costs more to produce (e.g. marginal water supplied by desalination) and the cost (and value) of water increases.

Once a value for the pollution is established then it can be used in cost-effectiveness analysis to evaluate options. The cost curve for pollution abatement can also be used to develop a cost-effective plan for meeting pollution abatement targets.

8.2. Check Assumptions

The following assumptions need to be checked to ensure the cost curve reflects the context. It is relatively easy to update the existing data with new assumptions.

a) Discount rate (currently a 3% discount rate has been assumed with sensitivity analysis of 5.5%);

b) Length of time of the analysis (currently 20 years has been assumed); and

c) Allocation of costs:

- Wastewater treatment plants (currently based on a WSAA report for allocation of costs for wastewater treatment plants);
- Stormwater (currently based on Best Practice Environmental Management Guidelines by DERM); and
- The value of water (currently based on an assumed bulk water price path).

Figure 26 shows a generic cost curve for pollution abatement. These cost curves have been developed for abatement of total phosphorus, total nitrogen, total suspended solids and greenhouse gas emissions.

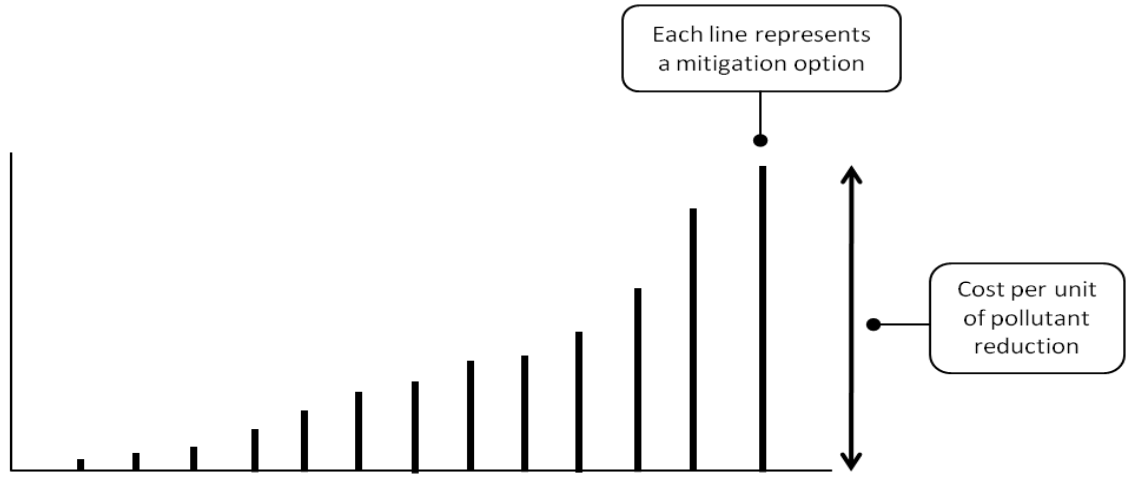

Figure 26: Format of existing cost curve for pollution abatement options.

8.3. Characterise Loads, Catchment Values and the Sustainable Load Targets

The sustainable load target identifies how pollution abatement is required. This includes an understanding of existing ecosystem health and value and the changes to loads that may occur in the future. A starting point may be the load reduction required to maintain the current state of the ecosystems health over the next 20 years. Detailed modelling of sustainable loads can also define the load reduction required.

The breakdown of loads into diffuse and point source is required to align abatement strategies to meet the load reductions. For example, if diffuse source pollutants are the main pollutant load in the catchment then diffuse abatement is likely to be required regardless of its cost-effectiveness compared to point source reductions.

8.4. Identify Relevant Pollution Abatement Strategies and Load Reduction Potential

The amount of pollution abatement for each abatement option needs to be estimated for the catchment. This estimate will draw upon existing plant and stormwater data and likely changes over the coming decades due to development. Figure 27 illustrates the format of the abatement cost curve with abatement quantities considered.

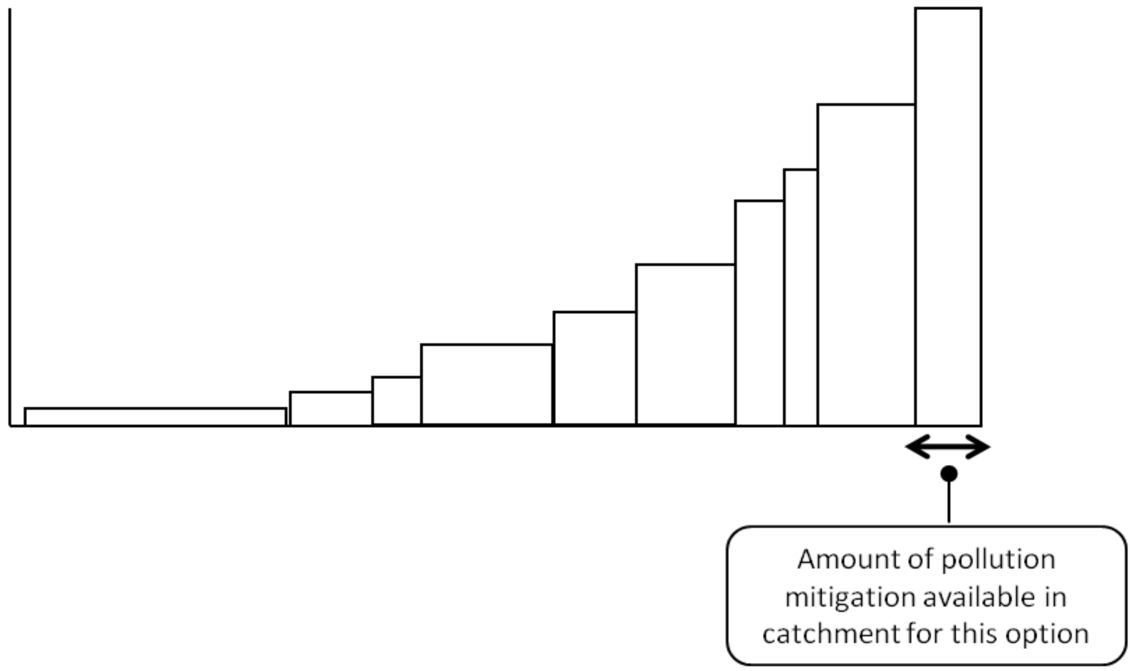

Figure 27: Cost curve with quantities of pollutants mitigated by each option.

Figure 28 illustrates how the cost curve can be used to identify a suite of cost-effective abatement options and to calculate a weighted average price for pollution abatement to meet the load reduction target.

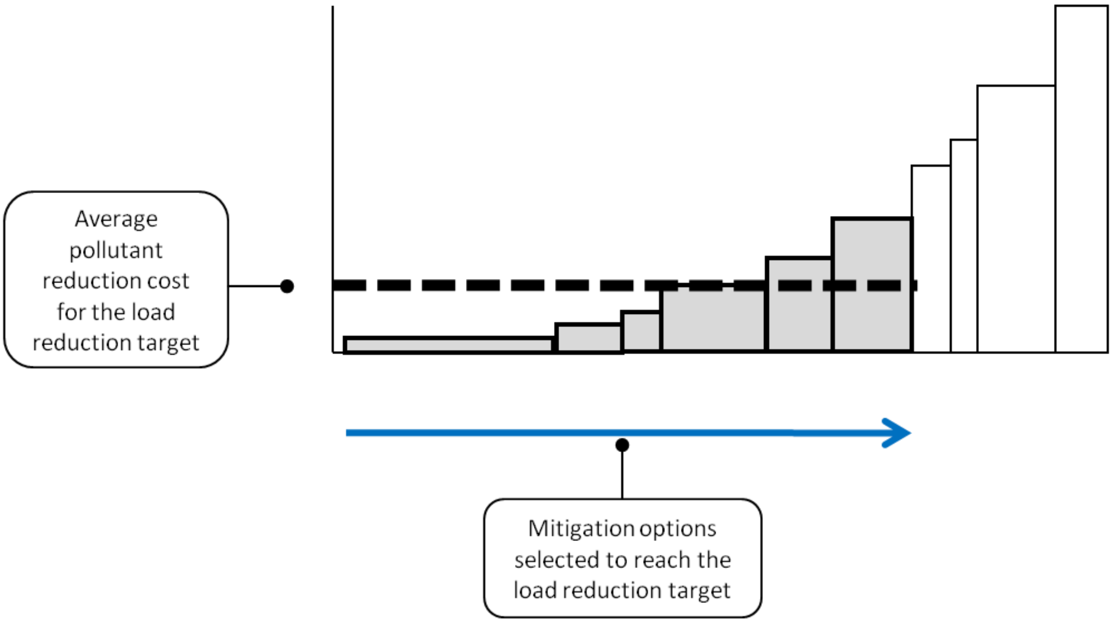

Figure 28: Sustainable load target defines what options are required and price for pollutant.

APPENDIX 1: Additional Literature on MACC, Policy Targets and Systems Analysis

The use of policy targets to define the pollution reduction target can be further explored with respect to literature for ecological economics and systems analysis. For example, in Chesapeake Bay, US, it was noted that pollution abatement costs were lower than damage costs from pollution on environmental services (Birch, Gramig *et al.*, 2011). In this sense, the pollution abatement costs provide a conservative (low) estimate of environmental value. However, it was also noted that damage costs can have large uncertainty especially for productivity, biodiversity and recreation (Compton, Harrison *et al.*, 2011).

The Abatement Cost Method (IDC, 2003 p40; StandardsAustralia, 1999. - p14; Bowers and Young, 2000 - p35) uses pollution abatement costs (neutralisation costs (Bowers and Young, 2000 – p40), or engineering costs for the abatement of a pollutant. The cost of correcting the externality is relevant if avoiding the externality is a constraint on decision making (Bowers and Young, 2000; CommonwealthofAustralia, 2006) . The comparison of options can then consider reduced pollutant loads as a cost saving approach (CommonwealthofAustralia, 1991 – p23). The report *Valuing Externalities: A Methodology for Urban Water Use* was used to define the method and provides further information on the definition of externalities and the application of various methods. This approach has also been clearly articulated and applied in SEQ for assessing cost-effectiveness of water quality interventions in SEQ (Alam, Rolfe *et al.*, 2008).

The method was also considered with respect to literature on the economic valuation of nature. In particular, the use of engineering abatement costs was considered valid for estimating environmental value if: (i) the engineered system provides functions that are equivalent in quality and magnitude to the natural function; (ii) that the human-engineered system is the least cost alternative way of performing this function; and (iii) that individuals in aggregate would in fact be willing to incur these costs if the natural function were no longer available (Bockstael, Freeman *et al.*, 2000). The method satisfies the first condition by focusing on the abatement of a physical quantity of pollution entering the catchment. The pollution is also considered in context of the sensitivity and health of the receiving environment. Conditions (ii) and (iii) are addressed by considering a marginal cost pollution abatement curve. The cost curve identifies the least cost alternatives as well as explores the relationship of the cost to the objectives. As a starting point, the existing 'duty of care' for the environment (Bowers and Young, 2000 – p12) was based upon Government water quality objectives, planning and legislation and was adopted as the 'willingness to incur' the costs.

The Abatement Cost Method for valuing pollution is similar to values which are based upon the cost of production. This provides a similar approach to valuation as applied in engineering project costs as well as costs for resource such as water. This method of valuation is similar to the value placed upon potable urban water. For example, water is currently valued at a couple of dollars per kilolitre and largely reflects its cost of production. The value of water has increased recently due to increased capital and operating costs. The addition of new water sources with relatively high marginal costs, such as desalination, increases the average cost of production and the average value placed upon potable water.

The use of the Abatement Cost Method to value pollutants is also analogous to the expansion of the system boundary to include externalities in methods such as Life Cycle Assessment. In this case, pollutants are brought into the system boundary for the production of a good or service by including the energy and materials required to treat the pollutant. A similar approach is used in Input-Output Analysis by considering the cost and impact on production for reducing a pollutant to a 'tolerated' level (Leontief, 1986 – p249-252). In this study, abatement costs were used to bring externalities into an evaluation of cost-effectiveness for water supply options. The following figure illustrates energy and material flows from an urban water option and the expansion of the system boundary to treat the 'pollutants'. The term 'pollutant' requires some caution as the pollution may also be considered as a resource either via recycling or use in another option.

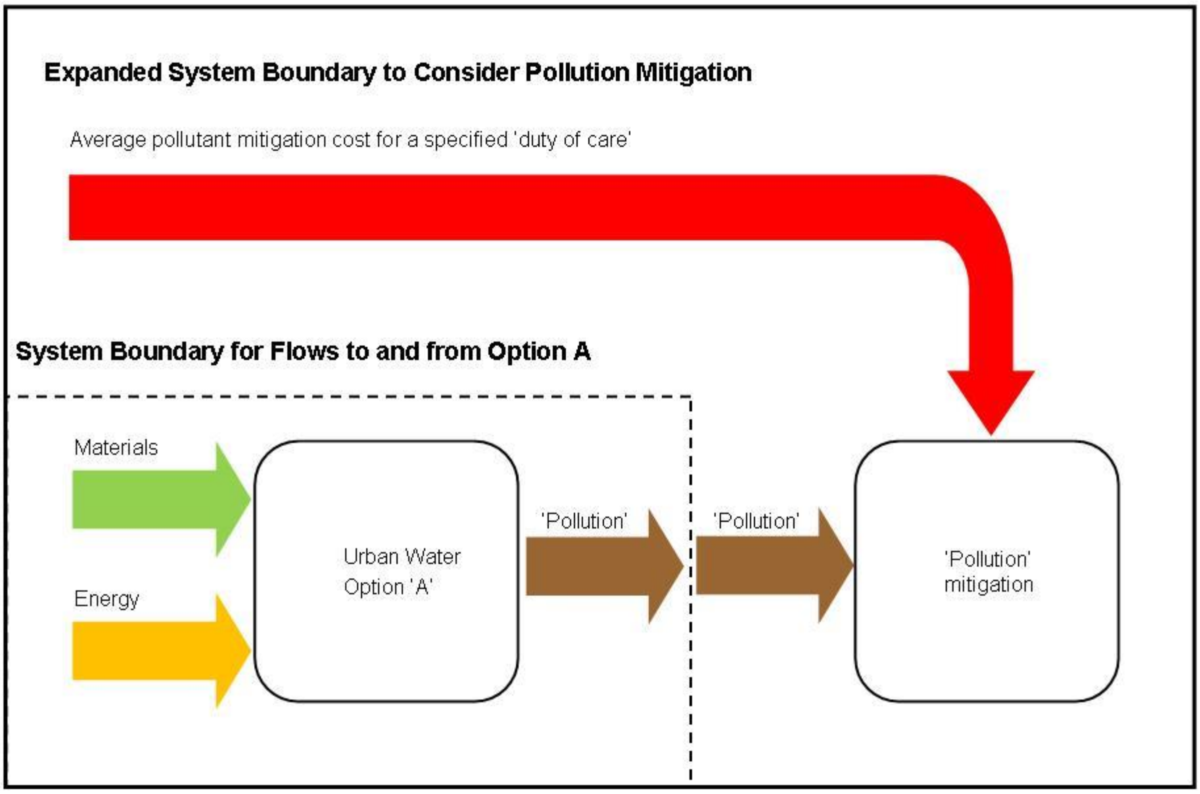

Figure 29: Expansion of System Boundary for Pollution Abatement.

APPENDIX 2: Data from Melbourne Water Used for Calculation of Confidence Intervals for Hydraulic and Water Quality Development Fees for Stormwater

Sample	No.	Locality (Municipality based)	Current Residential Rate - Water Quality ($/ha)
1	9201	Banyule City	8,645
2	9202	Bass Coast Shire	8,317
3	9203	Baw Baw Shire	12,258
4	9204	Bayside City	8,645
5	9205	Boroondara City	8,317
6	9206	Brimbank City	6,347
7	9207	Cardinia Shire (Port Phillip Catchment)	11,271
8	9243	Cardinia Shire (Western Port Catchment)	11,271
9	9208	Casey City (Port Phillip Catchment)	9,139
10	9244	Casey City (Western Port Catchment)	9,139
11	9209	Darebin City	7,824
12	9210	Frankston City (Port Phillip Catchment)	9,301
13	9245	Frankston City (Western Port Catchment)	9,301
14	9211	French Island	7,990
15	9212	Glen Eira City	8,206
16	9213	Golden Plains Shire	5,413
17	9214	Greater Dandenong City	9,084
18	9215	Greater Geelong City	5,413
19	9216	Hepburn Shire	8,097
20	9217	Hobson's Bay City	6,347
21	9218	Hume City	6,839
22	9219	Kingston City	9,191
23	9220	Knox City	9,139
24	9221	Macedon Ranges Shire	6,895
25	9222	Manningham City	9,960
26	9223	Maribyrnong City	6,347
27	9224	Maroondah City	9,794
28	9225	Melbourne City	8,317
29	9226	Melton Shire	5,745
30	9227	Mitchell Shire	6,786
31	9228	Monash City	9,191
32	9229	Moonee Valley City	6,513
33	9230	Moorabool Shire	8,097
34	9231	Moreland City	8,206
35	9232	Mornington Peninsula City (Port Phillip Catchment)	8,593
36	9246	Mornington Peninsula City (Western Port Catchment)	8,593
37	9233	Murrindindi Shire	11,655
38	9234	Nillumbik Shire	8,317

Sample	No.	Locality (Municipality based)	Current Residential Rate - Water Quality ($/ha)
39	9235	Port Phillip City	6,238
40	9236	South Gippsland Shire	11,875
41	9237	Stonnington City	8,427
42	9238	Whitehorse City	9,687
43	9239	Whittlesea City	7,769
44	9240	Wyndham City	5,692
45	9241	Yarra City	7,169
46	9242	Yarra Ranges Shire (Port Phillip Catchment)	12,531
47	9247	Yarra Ranges Shire (Western Port Catchment)	12,531
		Total	**400,422**
		Median	8,317
		Average	8,519
		Standard deviation	1,875
		Confidence	536
		Upper 95% confidence limit	9,055
		Lower 95% confidence limit	7,983

Sample	No.	Greenfield Scheme Name	Current Base Rate (standard residential) ($/ha)	
			Hydraulic	Water Quality
1	8226	Abey Road DS (Preliminary)	92,590	22,530
2	1612	Abrehart Road DS (Interim)	39,000	0
3	4480	Aitken Creek DS	17,915	15,378
4	7701	Alfred Road DS	45,890	24,892
5	8055	Arnolds Creek West Strategy	0	0
6	1202	Ballarto Road DS	25,674	10,731
7	2324	Baxter East DS	21,911	5,959
8	2323	Baxter West DS	27,706	10,945
9	615	Berwick South DS	8,944	9,216
10	616	Berwick Township DS	10,301	13,896
11	4137	Billingham Road Strategy	0	0
12	6071	Black Flat Road DS	17,204	10,863
13	7720	Black Forest Road DS (preliminary)	28,781	14,202
14	6820	Blind Creek Strategy	0	0
15	1124	Bowen Road DS	38,594	11,871
16	1128	Braeside South DS	6,077	24,777
17	4653	Broad Gully DS	13,080	25,585
18	4365	Brodies Creek DS (Interim)	27,000	0
19	8060	Brookfield Creek Strategy	0	0
20	407	Bungalook Creek DS	16,224	15,300
21	2881	Bunyip Township DS (Interim)	39,000	0
22	2883	Bunyip West DS (Preliminary)	52,278	14,970
23	4115	Burgess Street DS	17,625	9,843
24	8510	Cairns Drive DS (interim)	27,000	0
25	1510	Cardinia Industrial DS	39,870	14,352
26	1502	Cardinia Road Drain DS	63,938	24,975
27	1101	Carrum Lowlands DS	16,392	11,113
28	6851	Central Creek DS (interim)	27,000	0
29	1103	Chandler Road DS	17,873	10,271
30	4047	Cheetham Creek DS	12,657	4,431
31	4101	Cherry's Creek DS (Preliminary)	38,465	6,739
32	0705A	Colemans Road DS	12,911	0
33	4642	Collard Drive DS	10,257	8,529
34	2371	Collison Road DS	61,551	3,799
35	703	Cranbourne Central DS	6,961	10,415
36	707	Cranbourne West DS	58,590	12,731
37	2251	Crib Point DS	51,187	18,517
38	8010	Davis Creek DS (interim)	27,000	0
39	8015	Davis Creek East DS (Preliminary)	82,152	6,738
40	1601	Deep Creek South DS	25,013	7,503
41	4100	Derrimut DS	21,904	14,534

Sample	No.	Greenfield Scheme Name	Current Base Rate (standard residential) ($/ha)	
			Hydraulic	Water Quality
42	4088	Dohertys Drain DS	54,550	6,022
43	6551	Donnybrook DS (Interim)	27,000	16,000
44	4681	Doreen DS	34,805	10,038
45	4621	Dry Creek DS	36,054	12,785
46	4139	East Moreton DS	17,275	10,021
47	1603	Eastern Drain DS	37,277	10,759
48	4440	Edgars Creek DS	33,995	6,446
49	1110	Edithvale Road DS	12,279	9,800
50	1316	Edrington Park DS	8,423	21,971
51	206	Essex Park Extension DS	19,945	9,315
52	8230	Exford Road DS (Preliminary)	18,590	13,780
53	8037	Eynesbury Estate Strategy	0	0
54	805	Ferntree Gully South DS	16,267	8,599
55	8228	Ferris Road DS (Preliminary)	28,630	8,730
56	4535	Findon Creek DS	38,674	17,050
57	607	Fitzgerald Road DS	8,786	8,330
58	1119	Five Ways DS	16,263	14,034
59	604	Fordholm Road East DS	16,055	9,982
60	4077	Forsyth Road Drain DS	48,596	32,208
61	408	Garden St DS	12,827	18,617
62	1111	Gartsides North DS	37,796	8,593
63	1107	Gartsides South DS	28,589	8,716
64	706	Glasscocks Road West DS	56,746	5,582
65	629	Golf Links Road DS	41,766	7,211
66	2327	Gomms Road DS	21,904	7,199
67	4615	Gorge Road DS	27,952	12,773
68	1302	Grasmere Creek DS (Interim)	39,000	16,600
69	624	Greaves Road DS	4,163	8,215
70	510	Greens Lane DS	16,133	11,002
71	4380	Greenvale West DS (Preliminary)	27,000	0
72	1114	Haileybury DS	40,579	4,156
73	714	Hall Road DS	68,794	8,314
74	501	Hallam North Road DS	23,468	3,513
75	602	Hallam South DS	24,744	13,805
76	621	Hallam Valley Contour Drain Remodelling	8,352	15,504
77	608	Hallam Valley DS	19,590	3,744
78	605	Hampton Park East Extension DS	15,280	109
79	6814	Harpers Creek DS (Preliminary)	41,504	18,076
80	4174	High Street Melton DS (Preliminary)	26,085	4,461
81	8090	Hogans Road DS	71,965	17,250
82	613	Homestead Road DS	34,401	6,037
83	631	Homestead Road Extension DS	94,770	7,356

Sample	No.	Greenfield Scheme Name	Current Base Rate (standard residential) ($/ha)	
			Hydraulic	Water Quality
84	302	Jacksons Road North DS	9,968	9,798
85	304	Jells Road South DS	15,378	8,390
86	2363	Junction Village DS (Preliminary)	0	0
87	6550	Kalkallo Creek DS (Interim)	27,000	0
88	4090	Kayes Drain DS	33,430	2,262
89	801	Kelletts Road DS	16,195	14,378
90	1604	Kennedy Creek DS	37,277	10,791
91	803	Kent Park DS	11,296	17,397
92	1115	Keysborough South DS	78,540	12,713
93	410	Kilsyth West DS	16,267	15,949
94	2231	Kings Creek DS	22,486	8,376
95	2326	Kinlora DS	30,447	11,561
96	6821	Kismet Creek Strategy	0	0
97	3821	Koo Wee Rup DS	22,949	10,520
98	314	Koomba Road DS	20,635	8,678
99	1204	Langwarrin DS	17,033	8,615
100	2331	Langwarrin South DS	18,566	7,096
101	1230	Lathams Road DS	8,872	16,588
102	6001	Laurimar DS	22,080	23,863
103	4080	Laverton Creek DS	42,983	7,121
104	4077A	Laverton RAAF DS	0	3,044
105	1221	Little Boggy Creek DS	9,014	7,771
106	405	Little Bungalook Creek DS - Part A	7,870	17,292
107	406	Little Bungalook Creek DS - Part B	16,267	17,292
108	1208	Lower Carrum Downs DS	36,950	8,876
109	711	Lyndhurst DS	42,113	4,995
110	705	Lyndhurst North DS	48,742	10,258
111	704	Lyndhurst South DS	24,879	7,987
112	802	Lysterfield West DS	9,771	4,092
113	4490	Malcolm Creek Strategy	0	0
114	7742	Manor Lakes DS (preliminary)	9,592	19,410
115	1301	Manuka Road DS	18,305	13,556
116	8504	Masons Lane DS (interim)	27,000	0
117	1501	McGregor Road DS	21,781	6,667
118	2855	McNamara Road DS (Interim)	39,000	0
119	8229	Melton South DS (Preliminary)	54,000	5,120
120	4635	Mernda Central DS	42,678	6,730
121	6004	Mernda North DS	42,438	19,838
122	4634	Mernda South DS	39,078	24,493
123	413	Merrindale DS	17,814	16,609
124	8051	Minns Road Strategy	0	0
125	701	Monahans Road DS	13,930	7,587

Sample	No.	Greenfield Scheme Name	Current Base Rate (standard residential) ($/ha)	
			Hydraulic	Water Quality
126	4172	Mt Cottrell Road DS (Preliminary)	61,767	0
127	611	Narre Warren Township DS	23,769	8,435
128	6985	New Gisborne DS (interim)	27,000	0
129	1306	O'Neil Road DS	36,850	32,854
130	614	O'Shea's Road DS	19,110	4,493
131	1315	Officer DS (Preliminary)	65,109	6,230
132	201	Ordish Rd North DS	19,929	10,774
133	508	Ordish Road South DS	19,929	15,464
134	1602	Pakenham Creek DS	37,277	10,852
135	1507	Pakenham West DS	9,316	15,364
136	4171	Paynes Road DS (Preliminary)	82,910	0
137	7726	Pedder Street Drain DS	25,889	22,119
138	1140	Perry Road DS	64,084	11,716
139	8075	Point Cook DS	30,647	27,544
140	1222	Potts Road West DS	35,475	9,503
141	601	Princes Domain DS	14,564	13,376
142	6961	Riddells Creek DS (Interim)	27,000	0
143	518	Rockleigh Park DS	17,926	3,902
144	6681	Romsey DS (Interim)	27,000	0
145	303	Rowville DS	13,293	8,236
146	4479	Roxburgh Park (Coopers Road) DS	24,450	1,424
147	4487	Roxburgh Park (Craigieburn South) DS	24,450	1,424
148	4478	Roxburgh Park (Patullos Lane) DS	24,450	1,424
149	4072	Sayers Drain DS	30,861	37,919
150	4173	Shogaki Drive DS (Preliminary)	43,484	5,049
151	4073	Skeleton Creek East DS	3,569	20,675
152	4070	Skeleton Creek South DS	2,758	8,911
153	1205	Skye Road North DS	16,201	12,106
154	2328	Somerville DS	16,901	5,332
155	4628	South Morang DS	19,542	29,534
156	4230	Steele Creek DS	40,113	8,364
157	4141	Stony Hill Creek DS	18,052	15,507
158	4079	Tarneit Creek DS	18,183	15,122
159	6531	Taylors Creek DS	57,201	0
160	6529	Taylors Creek East DS (Preliminary)	24,030	15,810
161	4150	Taylors Hill West DS	28,652	0
162	619	Ti-Tree Creek DS	19,902	7,634
163	610	Troups Creek East Branch DS	16,015	14,406
164	606	Troups Creek West Branch DS	37,932	2,027
165	709	Turf Farm Strategy	0	0
166	1203	Upper Boggy Creek DS	9,014	7,885
167	4381	Upper Brodies Creek DS (Interim)	27,000	0

Sample	No.	Greenfield Scheme Name	Current Base Rate (standard residential) ($/ha)	
			Hydraulic	Water Quality
168	0404-N	Upper Bungalook Creek DS (North)	20,173	15,460
169	0404-S	Upper Bungalook Creek DS (South)	18,441	15,460
170	1209	Upper Carrum Downs DS	30,136	10,324
171	1403	Upper Gum Scrub Creek DS (Preliminary)	46,225	5,493
172	4532	Upper Hendersons Creek DS	36,808	9,923
173	1231	Valley Road DS	32,189	10,506
174	312	Vermont South DS	18,690	9,110
175	6530	Wallan Airfield Strategy	0	0
176	6535	Wallan Street Drainage Strategy	0	0
177	316	Wantirna North DS	15,833	16,224
178	2221	Warringine Creek DS	29,168	6,905
179	2854	Wattletree Road DS (Interim)	39,000	0
180	1210	Wells Road DS	10,527	10,998
181	7716	Werribee West Drain DS	51,027	7,195
182	4622	Wiltonvale Creek DS	42,179	16,847
183	4360	Yuroke Creek DS (Interim)	27,000	0
		median	24,450	8,730
		average	27,076	9,671
		standard deviation	18,604	7,532
		sample	183	183
		95% confidence interval	2,645	1,091
		upper confidence boundary	29,721	10,762
		lower confidence boundary	24,431	8,580

APPENDIX 3: Assumptions from Watcost for Costing of Reuse of Wastewater Effluent for Plantations

CASH FLOW ASSUMPTIONS AND RESULTS	
Variable	
Effluent assumptions	
Effluent available from plant (ML/year)	183
Annual storage evaporation losses (%)	6
Number of months of storage required	4
Annual total loading rate (ML/year)	172.02
Plantation assumptions	
Land purchase cost ($/ha)	$2500
Type of plantation	Euc. saw
Plantation effluent use (ML/ha/year)	6.5
Irrigation method	flood
Irrigation efficiency (%)	70
Effluent loading rate (ML/ha/year)	9.3
Area of plantation (ha)	18.5
Number of seedlings per hectare	1200
Distance from plant to storage pond (m)	1000
Yields	
Euc saw 1st thinnings yield (year 3) (m3/ha)	13
Euc. saw 2nd thinnings yield (year6) (m3/ha)	34
Euc. saw 3rd thinnings yield (year9) (m3/ha)	62
Euc. saw clear fell yield (year15) (m3/ha)	293
Market assumptions	
Harvesting costs-pulp and thinnings ($/m^3)	$20
Harvesting costs-sawlogs ($/m^3)	$15
Distance from plantation to pulplog market (km)	50
Euc pulplog price at mill ($/m^3)	$42
Pine pulplog price at mill ($/m^3)	$37
Distance from plantation to sawlog market (km)	50
Euc sawlog price at mill ($/m^3)	$70
Pine sawlog price at mill ($/m^3)	$70
Financial assumptions	
Discount rate (%)	3.0
Financial indicators	
NPV ($/ha)	-$30,398
Amortised value ($/ha/year)	-$2,546
Amortised value ($/ML)	-$258
Initial capital costs ($)	$335,027

		Enter either	
Rotation		1	Euc. pulp
		2	Euc. saw
	2	3	Pine pulp
		4	Pine saw

		Irrigation method	Establish. costs ($/ha)	Irrigation efficiency (%)
Irrigation	Enter either			
	1	flood [1]	$3,000	70
1	2	sprinkler	$5,000	76
	3	drip	$4,000	85

[1] Euc pulp and Euc saw only

Yields	
Euc. pulp	(m3/ha)
1st coppice, year 5	111
2nd coppice, year 10	111
3rd coppice, year 15	111
Euc. sawlog	
Thinnings, year 3	13
Thinnings, year 6	34
Thinnings, year 9	62
Clear fell, year 15	293
Pine pulp	
1st clear fell, year 7	197
2nd clear fell, year 15	197
Pine sawlog	
Thinnings, year 7	98
Clear fell, year 15	440

CASH FLOW ASSUMPTIONS AND RESULTS

Variable	
Effluent assumptions	
Effluent available from plant (ML/year)	183
Annual storage evaporation losses (%)	6
Number of months of storage required	4
Annual total loading rate (ML/year)	172.02
Plantation assumptions	
Land purchase cost ($/ha)	$2500
Type of plantation	Pine pulp
Plantation effluent use (ML/ha/year)	6.5
Irrigation method	sprinkler
Irrigation efficiency (%)	76
Effluent loading rate (ML/ha/year)	8.6
Area of plantation (ha)	20.1
Number of seedlings per hectare	1200
Distance from plant to storage pond (m)	1000
Yields	
Pine pulp 1st clear fell yield (year 7) (m3/ha)	197
Pine pulp 2nd clear fell yield (year15) (m3/ha)	197
	0
	0
Market assumptions	
Harvesting costs-pulp and thinnings ($/m^3)	$20
Harvesting costs-sawlogs ($/m^3)	$15
Distance from plantation to pulplog market (km)	50
Euc pulplog price at mill ($/m^3)	$42
Pine pulplog price at mill ($/m^3)	$37
Distance from plantation to sawlog market (km)	50
Euc sawlog price at mill ($/m^3)	$70
Pine sawlog price at mill ($/m^3)	$70
Financial assumptions	
Discount rate (%)	3.0
Financial indicators	
NPV ($/ha)	-$39,900
Amortised value ($/ha/year)	-$3,342
Amortised value ($/ML)	-$367
Initial capital costs ($)	$374,989

Rotation	Enter either	
	1	Euc. pulp
	2	Euc. saw
3	3	Pine pulp
	4	Pine saw

Irrigation	Enter either	Irrigation method	Establish. costs ($/ha)	Irrigation efficiency (%)
	1	flood [1]	$3,000	70
2	2	sprinkler	$5,000	76
	3	drip	$4,000	85

[1] Euc pulp and Euc saw only

Yields	(m3/ha)
Euc. pulp	
1st coppice, year 5	111
2nd coppice, year 10	111
3rd coppice, year 15	111
Euc. sawlog	
Thinnings, year 3	13
Thinnings, year 6	34
Thinnings, year 9	62
Clear fell, year 15	293
Pine pulp	
1st clear fell, year 7	197
2nd clear fell, year 15	197
Pine sawlog	
Thinnings, year 7	98
Clear fell, year 15	440

CASH FLOW ASSUMPTIONS AND RESULTS

Lucerne/sorghum production

Effluent assumptions	
Effluent available from plant (ML/yr)	183
Storage evaporation losses (%)	6
Number of months of storage required	4
Annual total loading rate (ML/yr)	172.02
Land use assumptions	
Land purchase cost ($/ha)	$2,500
Pasture/crop effluent use (ML/ha/yr)	8.6
Irrigation efficiency (%)	70
Effluent loading rate (ML/ha/yr)	12.3
Area of crop (ha)	14.0
Distance from STP to plantation (m)	1000
Crop yield assumptions	
- **Lucerne**	
Establishment year	
• first cut (t/ha)	2.00
• later cuts (t/ha)	6.00
Maintenance years (years 2-4)	
• first cut (t/ha)	3.50
• later cuts (t/ha)	10.50
- **Sorghum** (t/ha)	6.50
Price assumptions (@farm gate, net all levies etc)	
- **Lucerne**	
• first cut ($/t)	$90
• later cuts ($/t)	$140
- **Sorghum** (t/ha)	$110
Discount rate (%)	3.0
Financial indicators	
NPV ($/ha)	-$40,228
Amortised value ($/ha/yr)	-$3,370
Amortised value ($/ML)	-$258
Initial capital costs ($)	$302,808

APPENDIX 4: Proposed Method for Cost-Effectiveness for Individual WSUD Devices

The size of an individual WSUD device as a fraction of the catchment depends on the treatment objectives and whether it is coupled together with other WSUD devices. Pollutant removal performance curves illustrate this relationship for individual WSUD devices by showing the increased load reduction for greater application of the device in a catchment (MBWCP, 2006). However, a WSUD device is often placed in a treatment train to make use of the different performance characteristics of different devices and to optimise performance overall. Consequently, the optimal size of an individual WSUD is difficult to determine in isolation of a treatment train. The logarithmic nature of many pollutant removal performance curves suggests that the application of the device past a particular point on the performance curve is unlikely to be cost-effective. For example, it may not be cost-effective to scale an individual WSUD to meet a pollutant reduction objective.

The following example explores the diminishing returns for individual devices and develops a 'rule of thumb' for their application. The example draws upon performance curves in *Water Sensitive Urban Design Technical Design Guidelines for South East Queensland, Version 1*; 2006 (MBWCP, 2006) with a recent study of *An Assessment of Stormwater Treatment Trains for Moreton Bay Regional Council* (van Woerden, 2010).

The performance curves for individual WSUD devices show a diminishing return for pollutant removal with increasing application in the catchment. Figure 30 shows a typical performance curve.

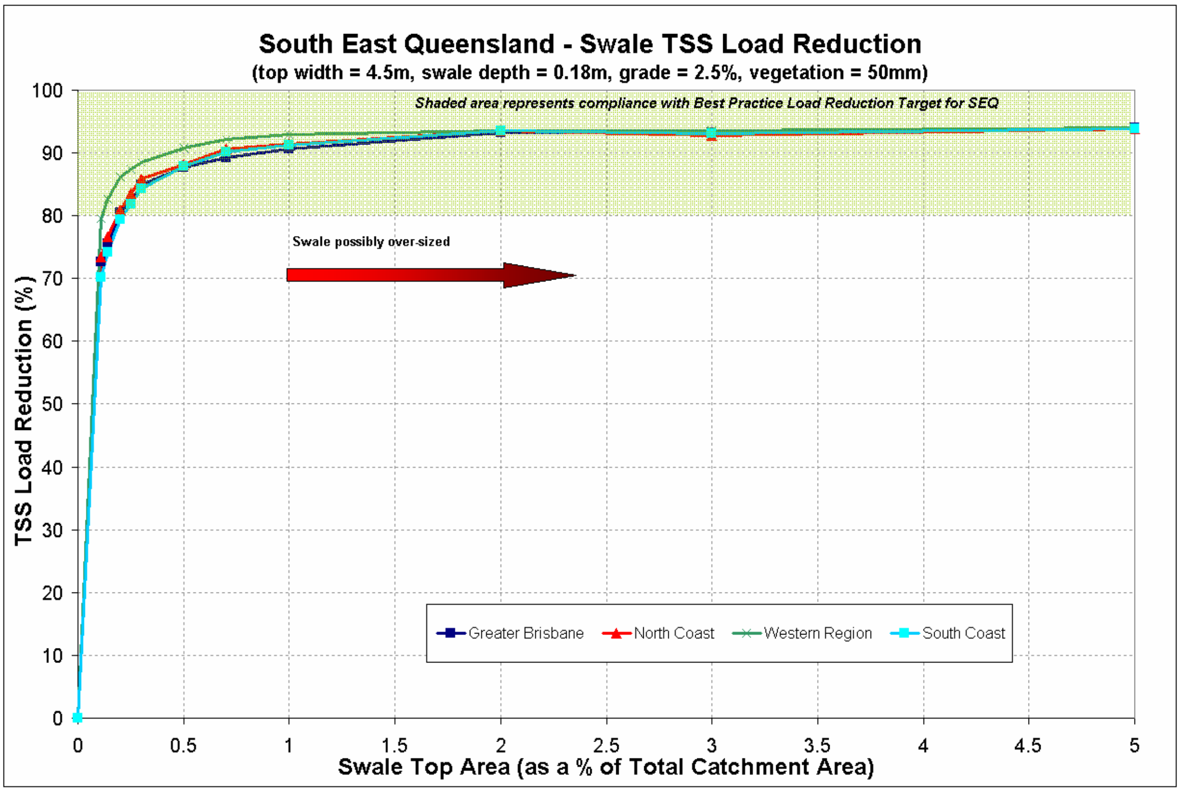

Figure 30: Swale Total Suspended Solids removal performance - p2-8 (MBWCP, 2006).

Figure 30 illustrates that a relatively large amount of pollutants are removed for a relatively small amount of the catchment area that is devoted to the swales. However, the continual addition of more of the WSUD has a diminishing return and reaches a point where further application does not remove much additional pollutants. This type of relationship can be described by log normal distributions. A log normal distribution can be used with pareto analysis for an '80/20 rule' where 80% of the impact is achieved with 20% of the effort and vice versa.

In addition, some devices may never reach a particular pollutant reduction target regardless of the level of application. This means the level of application of a WSUD device for cost-effectiveness is difficult to define by considering each device with a load reduction target. Figure 31 illustrates a WSUD that would require application to most of the area of the catchment to meet the TN load reduction target.

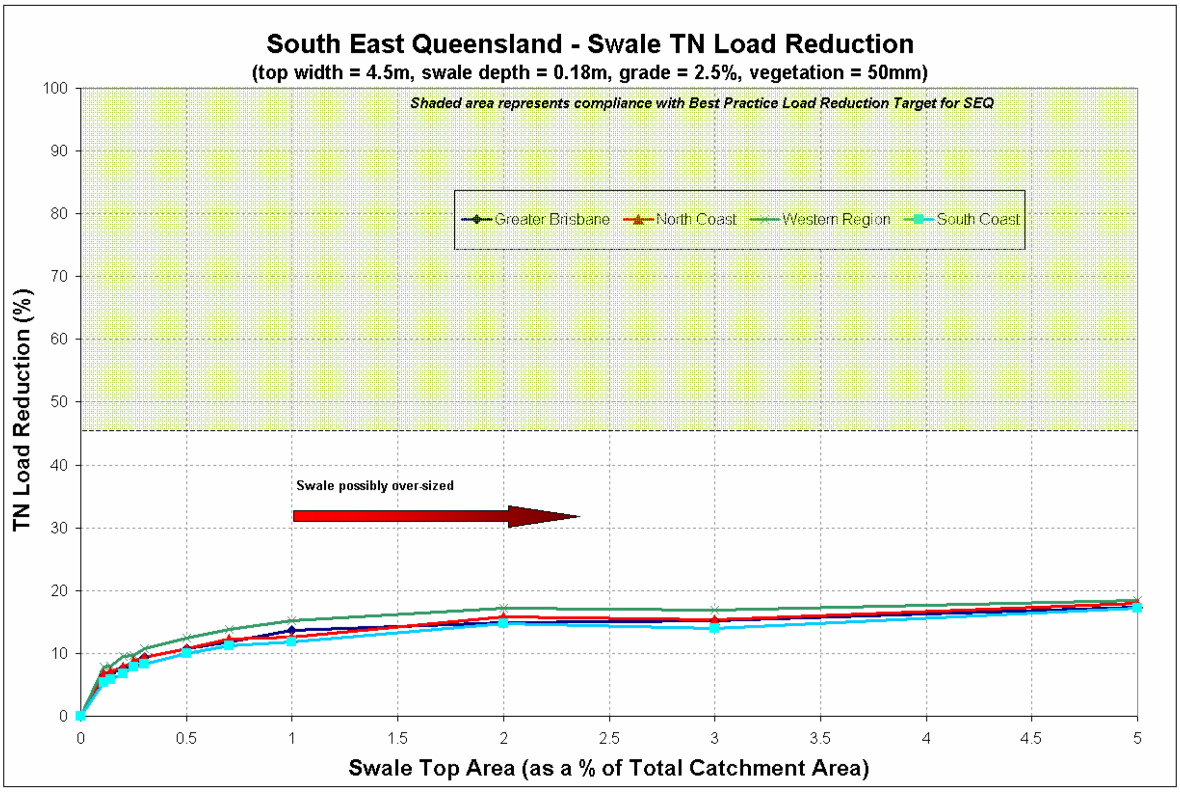

Figure 31: Swale TN removal performance p2-9 [1].

The following approach presents two criteria for defining the cost-effective application of a device based upon a log normal distribution and a 'pareto' 80/20 rule.

Criteria 1

It was assumed that if the distribution appeared to be a log distribution then an '80/20 rule' from pareto analysis would apply. It was assumed that a cost-effective application was limited to up to 80% of the maximum load reduction for the WSUD device. This value is somewhat arbitrary because there are diminishing returns from the onset of application of the device. However, with reference to the WSUD pollutant removal performance curves, this value appeared workable as a starting point for reaching load reduction targets. In the future, the approach could be optimised as part of a portfolio approach for WSUD devices to meet the load reduction target.

The application of Criteria 1 is outlined in the following example which considers the use of a swale to reduce Total Phosphorus (TP) in the North Coast of SEQ, shown in Figure 32 . The curve appears to have a log distribution and the maximum TP load reduction is approximately 67%. This is greater than the 60% TP load reduction required for compliance with the Best Practice load reduction target for SEQ. However, it is argued that it might not be cost-effective to reach compliance by just using a swale due to the diminishing returns for pollutant removal. If the cut off of 80% of the maximum load reduction is applied then only approximately 54% (67% * 0.8) of the TP load reduction would be achieved. The device does not meet the load reduction target by about 6% TP load reduction. However, by applying this condition reduces the land area required for swales by half to approximately 0.25% of the catchment. It is unlikely to be cost-effective to double the cost for a relatively small gain towards the target especially if the device is part of a WSUD treatment train.

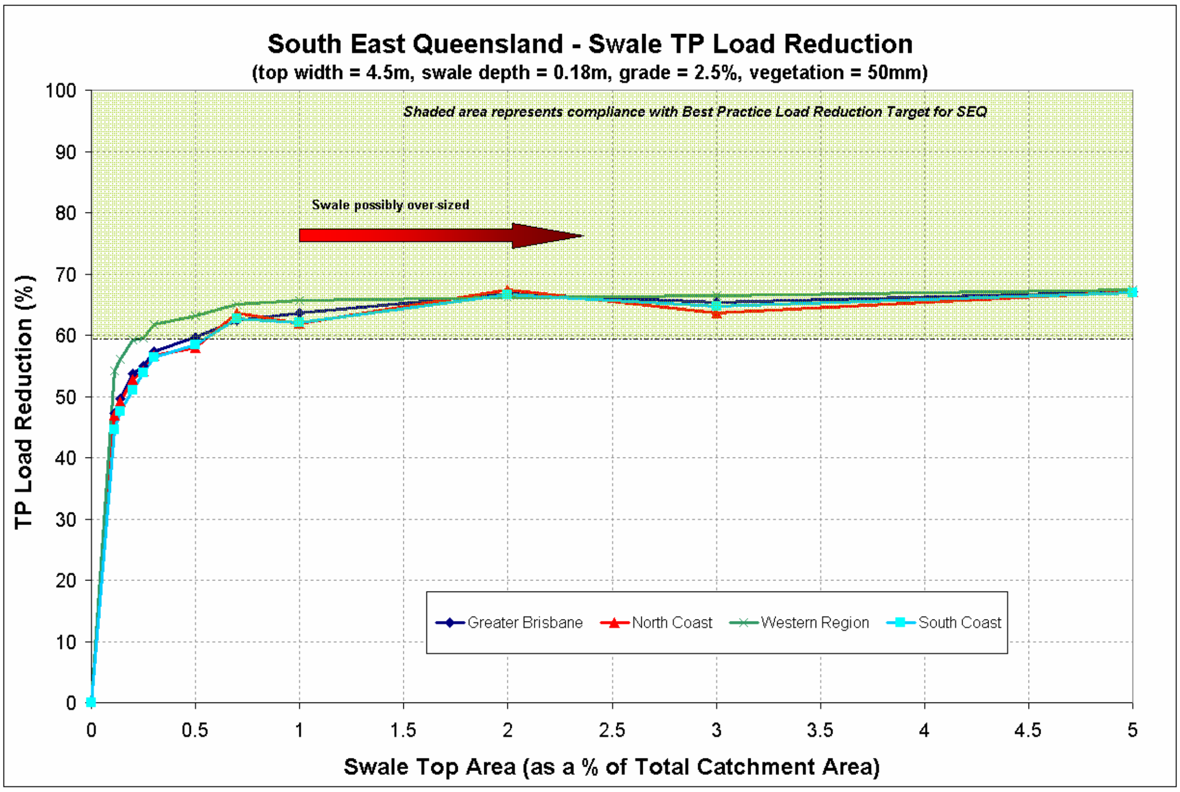

Figure 32: Swale Total Phosphorus removal performance p2-9 [1].

The use of 80% of the maximum load reduction also appeared to be a workable value when applied to a treatment train. Two WSUD treatment trains were developed for Moreton Bay Regional Council (van Woerden, 2010)and the performance is illustrated below in Table 56. Criteria 1 was applied to the performance and compared to compliance with best practice. Treatment Train A exceeded compliance for total nitrogen and Treatment Train B exceeded performance for total phosphorus. However, in all other cases the performance was within 10% of compliance. This has a number of implications. Firstly, a 10% failing is perhaps not significant given the uncertainty of performance curves and the effect of design variables for a particular context. Secondly, the difference in the land area for the two treatment train designs suggests that modifications of the design rather than changing the scale of the same design are likely to be more cost-effective.

This suggests that increasing the scale of the design is probably not cost-effective to meet compliance. For example, Treatment Train A needs to be increased 2-3 times in scale to meet compliance with TP reductions. Consequently, the cost for the treatment train would increase by about 2-3 times for an 8% reduction of TP towards the load reduction target.

Criteria 1 appears to perform a reasonable screen by limiting the effect of diminishing returns for a treatment train design. Consideration of a treatment train also enables compliance to be achieved for a range of pollutants at the lowest possible area of application. This research could be explored further to re-evaluate Criteria 1 for individual WSUD device pollutant ie it would be less than 80% due to the effect of a number of devices to meet compliance levels. However, the main criterion for this research is to limit the effect of diminishing returns on cost-effectiveness calculations and Criteria 1 is a workable approximation. A second criterion was then applied for the case where performance exceeds compliance levels.

Table 56: Load Reduction for two WSUD treatment trains in Moreton Bay Regional Council (van Woerden, 2010 –p 32).

	Mixed Catchment					
	Treatment Train A Pollutant Load Reduction (%)			Treatment Train B Pollutant Load Reduction (%)		
Treatment Area (% of catchment)	**TSS**	**TP**	**TN**	**TSS**	**TP**	**TN**
0.50%	81	52	29	70	42	10
1%	88	58	43	77	52	17
2%	93	62	59	84	62	27
3%	96	64	67	87	69	33
4%	97	64	72	90	73	38
5%	98	65	76	92	77	42
6%	98	65	78	93	80	46
7%	99	66	80	94	82	50
8%	99	66	82	95	84	53
Maximum load reduction (%)	99	66	82	95	84	53
Criteria 1: 80% of maximum load reduction (%)	79.2	52.8	65.6	76	67.2	42.4
Catchment area required Criteria 1. (% of catchment)	0.50%	0.50%	3%	1%	3%	5%
Criteria 2 - Limit application to compliance (%)-(DERM 2009)	80%	60%	45%	80%	60%	45%
Catchment area require for Criteria 2 (% of catchment)			1%		2%	
Catchment area from criteria 1 and 2 (% of catchment)	**0.50%**	**0.50%**	**1%**	**1%**	**2%**	**5%**

Criteria 2 – Limit Application of WSUD to Compliance

The second criteria relates to achieving compliance with the Best Practice Load Reduction Target in SEQ as shown in Table 56 as 'Criteria 2 - Limit application to compliance levels'. Criteria 2 provides a further screen for cost-effectiveness by limiting the application of a WSUD device to compliance levels (ie no further WSUD is applied if compliance has been reached). Criteria 2 also has the effect of harmonising the land area for the treatment train by reducing larger land application for particular pollutants back to compliance levels. This has the effect of making the area of application for the WSUD more consistent across the range of pollutants.

The application of Criteria 2 is illustrated in Table 56. For Treatment Train A, the effect is that Criteria 1 is practically the same as compliance for TSS, falls below by about 7% for TP and exceeds compliance for TN. When Criteria 2 is applied then the catchment area required for TN is reduced to a third of the Criteria 1 value. Treatment Train A appears to be cost-effective for reaching compliance. It also appears to be a relatively balanced design because the land area required is relatively consistent to reach each pollutant reduction target. Some degree of caution is required with the fine tuning of this approach. For example, the performance curves are likely to have an uncertainty range of at least 10%. As a result, if the performance is within 10% of each compliance level then further refinement of the scale and design may be difficult to justify as noted above.

On the other hand Treatment Train B does not appear cost-effective compared to Treatment Train A and requires much greater land for each pollutant reduction. Application of Criteria 2 to Treatment Train B reduces the land area required for TP. However, a large land area is required for TN load reduction for both Criteria 1 and 2 and suggests further work on the design of the treatment train.

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