Agricultural land management strategies to reduce phosphorus loads in the Gippsland Lakes, Australia

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Abstract

A target to reduce phosphorus flows into the Gippsland Lakes in south-eastern Australia by 40 per cent to improve water quality has previously been established by stakeholders. An integrated analysis at the catchment scale is undertaken to assess the agricultural land management changes required to achieve this target, and to evaluate the cost-effectiveness of these changes. It appears technically feasible to achieve a 40 per cent reduction in P load entering the lakes, but the least-costly way of doing so would require around A$1 billion over 20 years, a dramatic increase in the current levels of funding provided for management. On the other hand, a 20 per cent P reduction could be achieved at much lower cost: around $80 million over 20 years and requiring more modest land-management changes. The choice of optimal land-management strategies depends upon whether on-going costs for management maintenance are likely to be available after the initial funding ceased. Reliance on voluntary adoption of 'Current Recommended Practices' (CRPs) is unlikely to deliver changes in management practices at the scale required to have sufficient environmental impacts. Enforcement of existing regulations for the dairy industry would be amongst the most cost-effective management strategies. The major implications of this work for agriculturally induced diffuse-source pollution include the need for feedback between goal setting and program costs, and consideration of factors such as the levels of landholder adoption of new practices that are required, and the feasibility of achieving those adoption levels. Costs, land holder adoption of new practices and socio-political risks appear neglected in the formulation of many water quality programs. The framework used in this study provides a strong basis for discussion and debate about the environmental outcomes that can be achieved with limited budgets and also about the agricultural production and environmental trade-offs required to reduce diffuse-source nutrient pollution. The results are relevant to comparable water-quality programs worldwide.

1. Introduction

Agricultural systems have long been known to reduce the water quality of waterways through loss of sediments, phosphorus (P) and nitrogen (N) (e.g., Logan, 1993; Sharpley et al., 1999). Though not large compared to levels of nutrient application, nutrient losses from agriculture are often sufficient to impair the provision of economic, social and environmental values by waterways through promoting the nuisance growth of algae, leading to eutrophication. Excess P inputs are often the

major cause of eutrophication of surface freshwaters (Sharpley et al., 1999), although N can also contribute to the problem (Stoate et al., 2009). Nitrogen leaching is most often the cause of problems in marine systems (National Research Council, 2008; Stoate et al., 2009) and groundwaters (Maticic, 1999; Kay et al., 2009).

Water bodies affected by nutrient and sediment loss from agriculture include the Great Lakes (León et al., 2004), Chesapeake Bay (Simpson, 2010), the Florida Everglades (Rice et al., 2002), the Gulf of Mexico (National Research Council, 2008) and the Californian Central Coast (Dowd et al., 2008) in North America and the Baltic and Black Seas in Europe (Stoate et al., 2009). New Zealand has nutrient-impaired rivers (Monaghan et al., 2009) and highly-valued lakes, such as Lakes Taupo and Rotorua, are threatened (Connor et al., 2009; Kerr et al., 2007; Monaghan et al., 2007). Agriculturally-induced water quality problems (both P and N) also occur in Australia, threatening highly-valued environmental assets, such as the Great Barrier Reef (Waterhouse et al., 2010), the Peel-Harvey Inlet (Summers et al., 1999) and the Gippsland Lakes (Department of Natural Resources and Environment, 2002).

Point-source nutrient pollution (e.g. sewage treatment works, industrial sites, animal feeding operations) has often being successfully reduced in developed countries (Baker, 1993; National Research Council, 2008; Sharpley et al., 1999). This stems from the fact that nutrient loadings from point sources are measurable and occur from spatially confined areas, making them cost-effective to monitor and regulate. In contrast, regulation of non-point (diffuse) sources is more complicated for several reasons: the emissions are often essentially unmeasurable and highly variable across time (Shortle and Horan, 2001); there are large numbers of farmers involved, increasing the transaction costs of policy engagement; in some policy regimes agriculture receives special treatment – in the USA, for example, the Clean Water Act exempts many agricultural sources of pollutants from regulation (National Research Council, 2008; Ruhl, 2000); and regulation may require cooperation and agreement across different states or nations (National Research Council, 2009), such as for the Great Lakes of North America and many European waterways, lakes and marine ecosystems. Consequently, diffuse pollution is now the major contributor to declining water quality in much of North America, Europe and Australia, with agriculture being the largest contributor (Cherry et al., 2008; Waterhouse et al., 2010; Hancock et al., 2007).

Nutrient reduction programs are usually developed by setting targets aimed at attaining water-quality standards. These include the Water Framework Directive in Europe (Kay et al., 2009) and the total maximum daily load (TMDL) approach in the USA (National Research Council, 2008). Given the complexities of dealing with diffuse-source pollution problems, Gunningham and Sinclair (2002) suggest that policy approaches need to include 1) the removal of perverse market signals; 2) education and training; 3) underpinning regulation; and 4) a systematic approach to rewarding performance. The mix of policy tools differs between countries; for example, regulation is common in Denmark; voluntary approaches are common in the United Kingdom (UK); and Sweden uses a mixture of regulations, subsidies and education/extension (Kyllmar et al., 2006).

The aim of this study is to assess the potential for changed land use and land management practices to achieve nutrient reduction targets for an important Australian environmental asset, the Gippsland Lakes. The analysis identifies least-cost integrated strategies (involving combinations of many practices) to achieve a range of target nutrient reductions. It is an interdisciplinary and participatory study, bringing together researchers, managers, extension agents and various technical

experts. Results provide a trade-off curve between program cost and environmental benefits (Weersink et al., 2002), and sets of least-cost management actions to achieve particular targets.

2. Methods

2.1 Study area – Gippsland Lakes, Victoria, Australia

The Gippsland Lakes, located in south-eastern Australia (Figure 1), are one of the most important environmental assets in the state of Victoria. They consist of a system of coastal lagoons separated from the Tasman Sea by the coastal dunes of the Ninety Mile Beach. The main lakes – Wellington, Victoria and King – cover 340 km^2, with a shoreline of 320 km. There are seven major rivers draining into the Lakes and the size of the catchment is approximately 20,000 km^2. The Gippsland Lakes empty into the ocean through a constructed and dredged entrance at the town of Lakes Entrance. The Lakes and catchment contain a number of sites of national and international significance under the Ramsar Convention, the Japan-Australia Migratory Bird Agreement and the China-Australia Migratory Bird Agreement (Department of Sustainability and Environment, 2003).
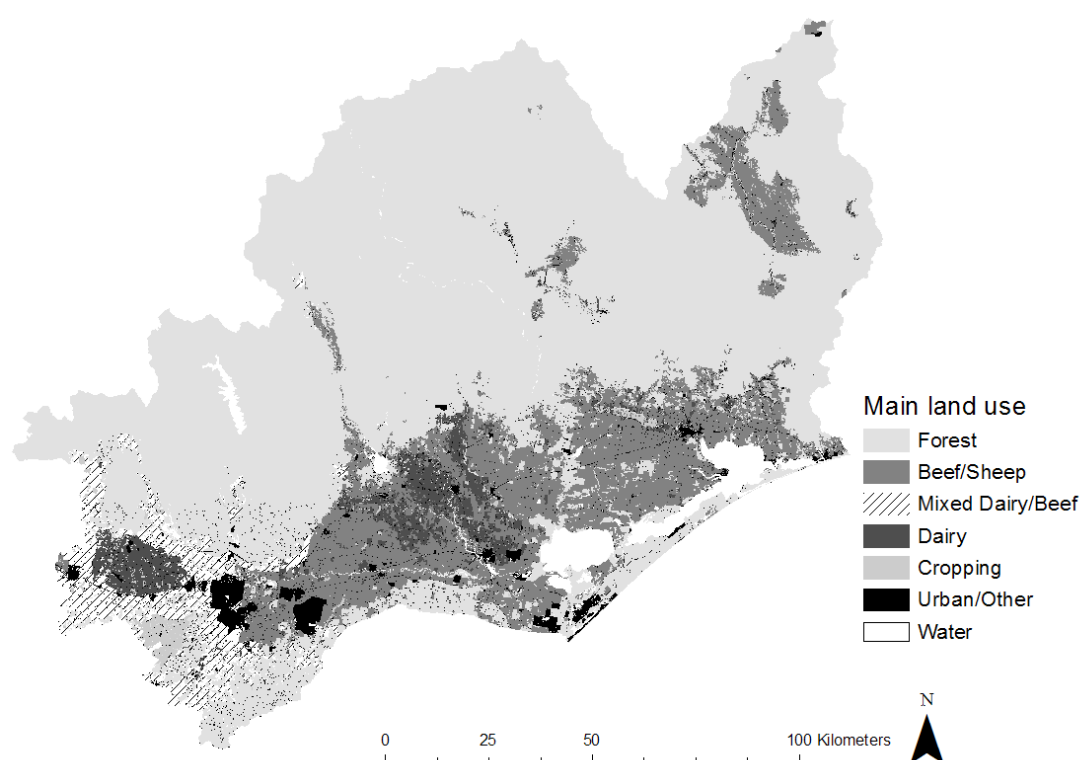
Within the catchment, agriculture and forestry generate over A$1 billion per year of agricultural products (Department of Primary Industries, 2006). The main industries are dryland grazing (beef and sheep), dairy production (irrigated and dryland in different parts of the catchment), forestry and horticultural cropping (potatoes, vegetables) (Figure 2). The Lakes are also important for tourism, being a major recreational boating destination, generating over A$250 million per year in tourism income to the regional economy (Department of Natural Resources and Environment, 2002).

The Gippsland Lakes are threatened by eutrophication (both P and N) (Harris et al., 1998). By Australian standards, there is a strong research basis for understanding major nutrient sources for the Lakes, with modelling studies (Grayson and Argent, 2002; Hancock et al., 2007) having been previously conducted.

A target of 40 per cent reduction in the average annual nutrient (P and N) load entering the lakes over 20 years was agreed to in 2002 (Department of Natural Resources and Environment, 2002) by stakeholders comprising a governing body called the Gippsland Lakes Taskforce (GLTF). The target was set based on an expectation that this would reduce the frequency of algal blooms and improve aquatic habitat.

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Figure 1. Location of the Gippsland Lakes, Victoria, Australia

Main land use
- Forest
- Beef/Sheep
- Mixed Dairy/Beef
- Dairy
- Cropping
- Urban/Other
- Water

Figure 2. Major land uses within the Gippsland Lakes catchment.
The approach for nutrient reduction activities has been largely based on previous identification of major nutrient sources. It has involved fencing off the riparian areas of major rivers to exclude livestock from the river banks, wetland restoration and

provision of small temporary payments to irrigated dairy farmers for adoption of improved management practices. Despite a concerted effort with the available budget, the GLTF has acknowledged that the current approach will fall far short of the target, with achievable P reductions being estimated at approximately 13 per cent (Ladson and Tilleard, 2006). This analysis was therefore highly relevant to the GLTF as it considers its future strategies and directions.

2.2 Stakeholder engagement in the analysis

The analysis was conducted in partnership with the GLTF. This group chose a trusted consultant with knowledge of the institutional context, previous research and implementation projects and people involved. The consultant oversaw the participatory process, and collection of information (previous reports, expert opinion, local knowledge). The major stages of participation were:

- Gaining GLTF support to conduct the analysis, and developing lines of communication.
- An inception workshop of key stakeholders to agree on the definition of the asset, its condition, main land-use types (irrigated dairy, dryland dairy, mixed dairy and beef, dryland beef/sheep grazing, forestry, horticulture), suggested scenarios for analysis and some parameters for the later analysis.
- A workshop of technical experts to identify previous research conducted in which there was sufficient confidence for it to be usable in this analysis, identify major knowledge gaps and agree on improved land management options for major land uses. For each nominated land-management option, implemented at a particular scale, experts were asked to estimate their effectiveness in reducing nutrient loss, as a percentage of the total nutrient load entering the lakes. From this, it became clear that there was less confidence about N impacts on the Lakes than for P. Management practices whose effectiveness in P reduction could not be estimated using expert opinion were discarded. In addition it was clear that land management practices which had been emphasised by the GLTF to date had been focussed on P-load reduction. For this reason, and due to lack of knowledge about impacts of management, N was excluded from the current analysis.
- A workshop of local extension staff. For each land-management option, extension staff were asked to identify the likely percentage levels of adoption by farmers under three levels of funding: zero public funding, current standard payment levels (if any), and the full opportunity cost to farmers of adopting the practice.
- Follow-up discussions with individual technical experts and extension staff for details of CRP effectiveness and cost estimates or clarification of information.
- Opportunities for technical experts and extension staff to review/change assumptions based on additional evidence.
- Discussions with the Executive Officer and Chair of the GLTF when decisions were required or when concerns arose.
- Presentation of an interim progress report to the GLTF. It became clear part way through the process that the costs to achieve the existing 40 per cent P reduction target would be prohibitively expensive. There was concern that results could be disregarded due to their lack of palatability. A decision was made to include additional scenarios based on politically achievable budgets and to assess what could be achieved by focussing on particular land

management practices. This was also viewed as a way of providing a range of options for the GLTF as the basis for developing a business case for further public funding. The scenarios developed are outlined in Table 1. Three of these (scenarios 16-18) were included to test how sensitive the results were to changing some of the important factors in the analysis.

- Visual presentation of the results to the GLTF for consideration, during which there was a high level of engagement and active discussion, including agreement on the need for additional time to consider how best to use the information.
- External review of the science and knowledge base used for the analysis and scrutiny about the assumptions used to underpin the analysis.

2.3 Use and modification of existing data, products and reports

Many reports providing data and information for the analysis were available. However, only a minority of information was in a form useful for the analysis. Some information was excessively complex (e.g. information about the land management options), and other information was not sufficiently detailed (e.g. the available land-use information on which previous modelling had been based). Without modification, reliance on existing information would have reduced the credibility of the analysis. As a result of these issues two products were developed:

- An updated land-use map that reduced inconsistencies in land-use classifications between the eastern and western parts of the catchment, relative to previous maps (Anonymous, 2007; Sargant, 2009), and that separated extensive beef cattle grazing and non-irrigated dairy farming systems (two new land classes of dryland dairy and mixed-dairy beef). The total P load from the generic dryland grazing systems estimated by previous modelling work was retained, but was apportioned between the new land classes (dryland dairy, mixed dairy-beef) assuming that P export rates (per ha) of dryland dairying land were three times that of dryland beef-sheep systems, based on the known large P surpluses generated by dairy systems (Gourley et al., 2007).
- A model integrating the available information to estimate the P-load reductions in the Gippsland Lakes in response to changes in land use or land management. This spreadsheet tool was based on a previous tool (Ladson and Tilleard, 2006), modelling results (Grayson, 2006) and information collected from the various stakeholder meetings and workshops. The model is described further below, and assumptions are outlined in an online report (http://cyllene.uwa.edu.au/~dpannell/archive/gippsland.pdf) and provided in detail in the spreadsheet tool itself, available online at (http://cyllene.uwa.edu.au/~dpannell/archive/gippsland_tool27.xls).

Data and assumptions are too voluminous to present here. To illustrate the model and its assumptions, consider the case of irrigated dairy. There are approximately 46,000 ha of irrigated dairy production in the catchment, and this land includes 1,400 km of small streams. Total P exports to the Lakes in the absence of additional interventions are estimated at 65 tonnes per year.

Table 1. Scenarios analysed for the Gippsland Lakes INFFER analysis.

Scenario no.	Description
1	40% P reduction by 2030, based on the 10 year average load to the Lakes
2	30% P reduction by 2030, based on the 10 year average load to the Lakes
3	20% P reduction by 2030, based on the 10 year average load to the Lakes
4	10% P reduction by 2030, based on the 10 year average load to the Lakes
5	$2 million/year for 5 years, followed by funding to maintain works
6	$5 million/year for 5 years, followed by funding to maintain works
7	$10 million/year for 5 years, followed by funding to maintain works
8	$2 million/year for 5 years, followed by no on-going funding
9	$5 million/year for 5 years, followed by no on-going funding
10	$10 million/year for 5 years, followed by no on-going funding
11	Payments for all CRPs at current rates for all industries, and including management of riparian areas for rivers and smaller streams
12	As for scenario 11, but excluding riparian management of smaller streams
13	Current incentive rates for irrigated-dairy CRPs, full enforcement of effluent management, no riparian management
14	Enforcement of farm effluent management only
15	Riverbank protection (riparian management on rivers) – full costs assuming 50% effectiveness in P reduction
16	Riverbank protection (riparian management on rivers) – full costs assuming 20% effectiveness in P reduction
17	As for scenario 1, but a 10-fold increase in valuation of the Lakes
18	As for scenario 3, but halving the valuation of the Lakes

Management options which have an impact on nutrient reduction for irrigated dairy (agreed to by participating stakeholders) are: on-farm re-use systems for irrigation tailwater, conversion to pressurized irrigation, irrigation automation, effluent management, irrigation farm plans, and management of riparian areas (fencing to exclude livestock). The first option, tailwater re-use, is potentially applicable to 40 per cent of irrigated dairy land. If adopted, it would reduce total P exports from that area by 80 per cent. The level of adoption of tailwater re-use is predicted to be 10 per cent in the absence of incentive payments, 18 per cent with existing modest incentive payment rates, and 90 per cent if payments cover the full opportunity cost of adoption. Total P export reductions from these three payment levels are calculated to be 5.2, 8.0 and 19.2 tonnes per year, respectively. The cost is $3,800 per ha up front with no ongoing maintenance cost.

Similar assumptions were required for each of the other management options for irrigated dairy, and repeated for each of the other land uses in the catchment: dryland (non-irrigated) dairy and mixed dairy-beef, dryland beef-sheep, forest, and river riparian areas (which span a variety of agricultural industries).

2.4 Economic optimisation

The spreadsheet P-load model described above was used as a basis for an optimisation analysis to estimate P-load reductions for scenarios 1-10 (Table 1). The optimisation analysis aimed to select the least-cost combinations of practices to achieve P-load reduction targets of 40, 30, 20, and 10 per cent (scenarios 1-4), as well as to assess the management actions that would maximise P-load reductions for fixed budgets ($2 million, $5 million and $10 million per year for 5 years) (scenarios 5-10). Scenarios 5-7 also allowed for on-going annual maintenance funding, whereas scenarios 8-10 assumed no ongoing funding is available.

Decision variables for the optimisation were: the area allocated to each land use, the levels of payments made to landholders to encourage adoption of Current Recommended Practices (CRPs), the percentage of relevant land over which these payments were offered, and the areas of land changed from current agricultural or forestry industries to non-commercial forests (with no nutrient inputs). The payment levels could be one of several values: zero, current low levels of payment, or higher payments calculated to cover the farmers' full opportunity costs. The percentage of land over which payments were offered was constrained so that it could not exceed the maximum area of land over which the CRP could be used minus the level of current adoption.

The number of potential combinations of management actions in the model is enormous. Optimisation was used to search for least-cost strategies. The interdependency of integer and continuous decision variables prevented the application of standard mathematical programming optimisation methods, so a genetic algorithm (GA) (Mitchell, 1996) was used. The GA was part of the Premium Solver Platform version 9.5 (Frontline Systems, 2009) package used for optimisation in Microsoft Excel.

When using GA, the solutions found are generally near-optimal but may not be absolutely optimal. To get solutions as close to optimal as possible, the following strategies were used:
1. The termination conditions for the search algorithm were tightly defined to prevent premature completion of the search process.
2. The GA was run more than three times for each problem instance to see whether superior solutions could be identified.
3. Solutions were used as starting strategies for subsequent optimisations.
4. The model was also coded in the General Algebraic Modeling System (GAMS) (Brooke et al., 2008) and optimised using alternative global optimisation algorithms (i.e. BARON, LindoGlobal).
5. Trial and error was used to test whether solutions could be feasibly improved.
These measures ensured that the identified solutions are of very high quality.

2.5 Investment Framework for Environmental Resources (INFFER)

The analysis made use of the INFFER framework (Pannell et al., 2009, www.inffer.org). INFFER is designed to assist with decision making about public investment in the environment and natural resources, such as natural habitat, rivers,

wetlands, threatened species, agricultural land, lakes, parks and reserves. INFFER is used to evaluate and compare alternative environmental projects on the basis of environmental benefits per dollar spent. For each potential project, it elicits and integrates the following information:

- The significance or importance of each environmental asset,
- Threats currently affecting or likely to affect the asset,
- A SMART (Specific Measureable, Achievable, Relevant, Time-bound) goal for the project,
- Proposed works/actions in the project,
- Time lags between undertaking works/actions and generating benefits,
- The effectiveness of proposed works/actions, and technical risks,
- Spin-off benefits and costs from the project,
- The adoptability of proposed works/actions by target users,
- Delivery mechanisms/policy mechanisms to be used in the project,
- Project costs,
- A variety of risks,
- Knowledge gaps and information quality.

Users are guided through a process to develop projects that are internally consistent, in that they will deliver the required on-ground action to achieve a specific, measurable, time-bound goal. The Public: Private Benefits Framework (Pannell, 2008) is used to guide the choice of delivery mechanisms (e.g. positive incentive mechanisms, negative incentive mechanisms, extension, technology development, no action).

An output from the INFFER process is a Benefit: Cost Index (BCI). The higher the BCI, the more attractive is investment in the project. The BCI is closely consistent with a standard Benefit:Cost Ratio used by economists, except that it usually does not require dollar values to be placed on environmental assets (e.g. using non-market valuation), instead relying on a scoring system for significance or value of the asset in good condition. Other variables in the BCI formula adjust the score to reflect the expected difference that the project will make to level of degradation of the asset. This study, involves comparison of alternative projects for the same environmental asset, so the asset score is the same in each case (with the exception of scenarios 17 and 18 which show the sensitivity of the asset value to the result), while the impact of the project on asset degradation varies. Thus, the asset significance score makes no difference to the ranking of projects in this study for scenarios 1 to 16 – it is the same for all projects. If it is assumed that an asset significance score of 100 corresponds to an asset value of A\$2 billion, then a BCI figure of greater than 1 is desirable, indicating that benefits exceed costs. Further details on the BCI, including the parameters used in this analysis, are provided in an Appendix.

3. Results

Table 2 shows core results for each of the scenarios described in Table 1. The second column shows the percentage reduction in P load entering the Lakes. For scenarios 1 to 4 and 17 to 18, the percentage reduction is defined as part of the scenario. For scenarios 1 to 10, the reduction is calculated by the optimisation algorithm as being

the greatest reduction that is possible for the available budget. Scenarios 11-16 were added as scenarios of interest to the GLTF or the research team itself. The third column shows the present value of project costs over 20 years. Project costs include up-front costs and annual maintenance costs, to which a real discount rate of 5 per cent was applied. Project administration costs were also included, assumed to be 5 per cent of upfront and annual maintenance costs. The fourth column shows the Benefit: Cost Index for each scenario.

In Table 3, land management strategies are presented for scenarios 1-10.

Based on the data collected from stakeholders, experts and existing reports, a 4 per cent reduction in P load was calculated as being achieveable at no public cost. This reduction results from low levels of voluntary adoption of CRPs by landholders who are environmentally motivated.

3.1 Least cost P-load reduction (scenarios 1-4)

Using the available science and assumptions used, the existing official target of a 40 per cent reduction in P load entering the Lakes (scenario 1) is technically feasible but very costly, requiring public expenditure of at least $994 million (present value) over 20 years (Table 2). This vastly exceeds current public expenditure on management of the Lakes. Least-cost actions to achieve this target consist of paying full opportunity costs to farmers to achieve maximum adoption of all CRPs, enforcement of effluent regulations in both irrigated and dryland dairy farming, management of riparian areas for both rivers and streams, works on forest roads to minimise erosion and conversion of 2,415 ha of irrigated dairy land to non-commercial native forests (Table 3). The estimated Benefit: Cost Index for this strategy is 0.02, indicating that benefits are likely to be much smaller than costs.

The 30 per cent P reduction target (scenario 2) is also expensive ($223 million over 20 years, Table 2), although much less so than the 40 per cent target. It also has a low BCI of 0.25. The 30 per cent target appears achievable without land-use change away from agriculture but still requires a diverse package of other measures (Table 3).

Twenty percent P reduction (scenario 3) can be achieved at a present value cost of $80.2 million (Table 2). For this target, the land-management changes are more modest, including paying full costs for riverbank protection (but not riparian management of smaller streams), payments for irrigated dairying farm plans and re-use systems, and enforcement of existing dairy regulations (Table 3). The BCI for this target is 1.0, indicating that benefits would be sufficient to approximately offset costs.

The cost of the 10 per cent P reduction target is relatively modest ($16.5 million), involving a much less extensive package of actions. It results in a favourable BCI of 2.6, indicating that benefits would significantly outweigh costs.

Table 2: Percentage P reduction load achieved and the associated Benefit:Cost Index for each investment scenario in the Gippsland Lakes.

Scenario	% P reduction	Present value ($ million) over 20 years	Benefit:Cost Index
1. 40% P reduction	40	994	0.02
2. 30% P reduction	30	223	0.2
3. 20% P reduction	20	80	1.0
4. 10% P reduction	10	16	2.6
5. $2m/yr for 5 yrs, with on-going annual costs	9	23	2.0
6. $5m/yr for 5 yrs, with on-going annual costs	18	114	0.6
7. $10m/year, for 5 yrs, on-going annual costs	22	142	0.6
8. $2m/yr for 5 yrs, no on-going funding	7	10	4.4
9. $5m/yr for 5 yrs, no on-going funding	7	25	1.7
10. $10m/yr for 5 yrs, no on-going funding	10	49	0.9
11. Current payments including rivers and smaller streams	17	192	0.4
12. As for 11, minus smaller streams	13	30	1.8
13. Current payments irrigated dairy + effluent enforcement, no riparian management for rivers or smaller streams	9	25	1.7
14. Effluent enforcement	6	16	2.8
15. Riverbank protection (50% effective, full costs)	16	61	1.1
16. Riverbank protection (20% effective, full costs)	9	61	0.7
17. As for 1, but V = 1000	40	994	0.2
18. As for 3 but V=50	20	80	0.5

Table 3. Optimal land management strategies to achieve P reduction scenarios in the Gippsland Lakes.

Scenario	Cost ($million)	Strategy
1. 40% P	Upfront costs: $584m Ongoing costs: $38m Present value (PV) over 20 years: $994m	Full costs of CRPs in all dairy and dryland beef-sheep, full costs riverbank protection and forest roads, enforcement of effluent management. Land retirement of 2,415 ha of irrigated dairy land.
2. 30% P	Upfront costs: $117m Ongoing costs: $10m PV: $223m	Full cost, tailwater re-use, over 30% of the relevant area; current payments, pressurised irrigation, 40%; enforcement, effluent management, 80%; current payments, irrigation farm plans, 98%; current payments, irrigated dairy riparian buffering, 82%; full cost, groundcover above 70%, 41%; full cost, riverbank protection, 99%.
3. 20% P	Upfront costs: $54m Ongoing costs: $2.5m PV: $80m	Current payments, tailwater re-use, 30%; current payments, pressurised irrigation, 40%; enforcement, effluent management, 80%; current payments, irrigation farm plans, 98%; full cost, riverbank protection, 90%.
4. 10% P	Upfront costs: $10m Ongoing costs:$0.6m PV: $16m	Current payments, pressurised irrigation conversion, 40%; current payments, riverbank protection, 73%.
5. 9%P	Upfront costs: $9.9m Ongoing costs: $1.2m PV: $23m	Current payments, pressurised irrigation, 40%; enforcement effluent management 80%; current payments, riverbank protection, 23%.
6. 18% P	Upfront costs: $25m Ongoing costs: $8.4m PV: $114m	Current payments, pressurised irrigation, 40%; enforcement effluent management 80%; full costs, groundcover above 70%, 40%; current payments, riverbank protection, 99%.
7. 22% P	Upfront costs: $50m Ongoing costs: $8.6m PV: $142m	Current payments, pressurised irrigation, 40%; enforcement effluent management 80%; full costs, groundcover above 70%, 38%; full costs riverbank protection, 82%.
8. 6.6% P	Upfront costs: $9.8m Ongoing costs: $0 PV:$9.8m	Current payments, tailwater re-use, 30%; current payments, pressurised irrigation, 40%; full cost, irrigation automation, 1%; current payments, irrigation farm plans, 98%.
9. 7.4% P	Upfront costs: $25m Ongoing costs: $0 PV: $25m	Full cost, tailwater re-use, 12%; current payments, pressurised irrigation, 40%; current payments, irrigation farm plans, 98%.
10. 10% P	Upfront costs: $49m Ongoing costs: $0 PV:$49m	Full cost, tailwater re-use, 27%; current payments, pressurised irrigation, 40%; current payments, irrigation farm plans, 98%.

Figure 3 shows the trade-off between P reductions and costs. The relationship is highly non-linear, with costs escalating as the environmental target becomes more stringent.
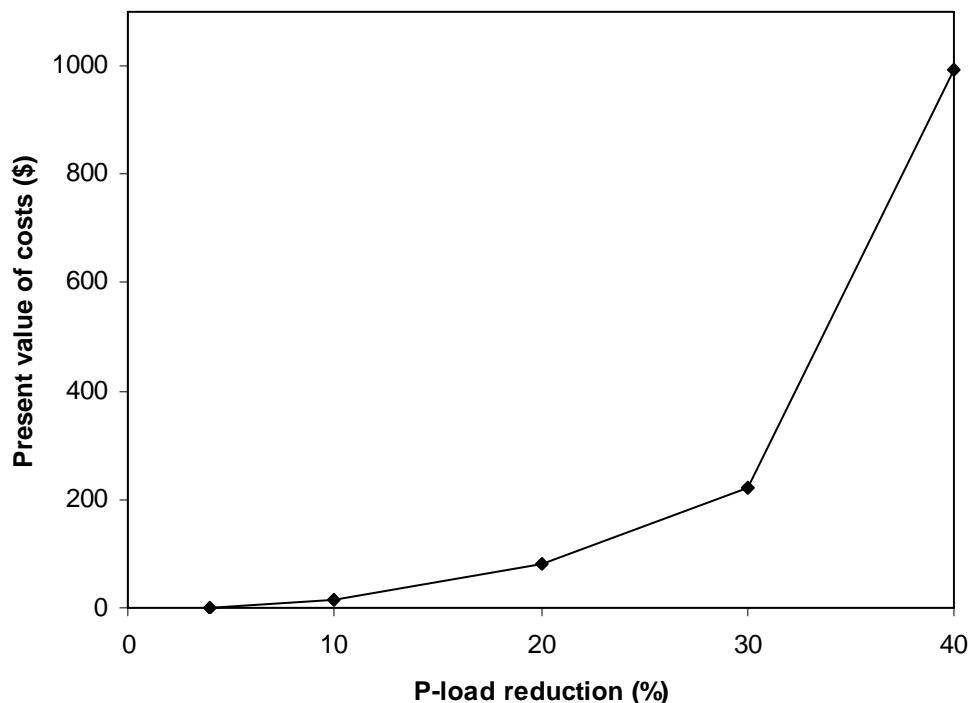
Figure 3. Trade-off between reductions in P-load into the Lakes and present value of costs of abatement (present value over 20 years using 5% real discount rate).

3.2 P reductions achievable for different budgets (scenarios 5-10)

Scenarios 5-7 show results that maximise P reductions for budgets of $2 million, $5 million and $10 million per year for five years, followed by additional ongoing maintenance costs. The levels of maintenance costs are not constrained. Scenarios 8-10 are for the same budgets for the first five years, but are constrained to actions that do not require on-going funding beyond the initial five-year phase. With ongoing funding available, higher P reductions are possible, but the additional costs are such that the BCIs are lower (Table 2). For example, a $2 million/year budget for five years with ongoing maintenance funding (scenario 5) could achieve 9 per cent P reduction with a BCI of 2.0. In contrast, where no maintenance funding is allocated beyond 5 years (scenario 8) it is only possible to achieve 7 per cent P reduction but the BCI rises to 4.4.

3.3 Payments for Current Recommended Practices, and effluent enforcement (scenarios 11-14)

The BCI results are sensitive to which CRPs are financially supported and whether effluent regulation is enforced. Scenarios 11-14 illustrate some of the possible choices. For example, scenario 11 includes payments at current standard levels for all

CRPs for farm and riparian management (both rivers and streams) across the dairy (both irrigated and dryland) and beef-sheep industries. This could achieve a P reduction target of 17 per cent (Table 2), but at large cost ($192 million) and low BCI (0.4). This is much more costly than scenario 3, which achieves a higher P reduction, illustrating the cost of constraining the choice of interventions. Excluding riparian management (scenario 12) reduces the P load to 13 per cent, and the cost to $30.1 million, but produces a more favourable BCI (1.8). Simply enforcing existing effluent regulations (scenario 14) has only a modest effect on P emissions but is very cost-effective (BCI 2.8).

3.4 Riverbank protection (scenarios 15-16)

Based on previous work (Hancock et al., 2007), the effectiveness of riparian management on rivers was assumed to be 50 per cent, meaning that total P exports from this land area would be reduced by this amount. If this strategy is used on its own (scenario 15), it achieves a significant P reduction (16 per cent) and appears cost-effective (BCI = 1.1). Given that there are very limited data on the rates of riverbank erosion (Hancock et al., 2007), a scenario of low effectiveness (20 per cent) was tested (Table 2). This reduced the P reduction to 9 per cent and the BCI to 0.7.

3.5 Sensitivity of the BCI to valuation of the Lakes

One area of uncertainty and debate is the value score (*V*) assigned to the asset. Scenarios 1-16 have been based on a *V* score of 100. Some may argue that the Gippsland Lakes would be worth much more than $2 billion, and use this as an argument to disregard low BCI results. In scenario 17 the assumed value of the Lakes was increased by 10-fold, keeping other parameters the same as for scenario 1. Even with such a large increase, the 40 per cent P reduction target is still not cost-effective (BCI 0.2, Table 2). Conversely, it could be argued that the score of 100 is too high. Keeping all other factors as for scenario 3 (20 per cent P least cost target) but reducing *V* to 50 (corresponding to a value of $1 billion), reduces the BCI from 1.0 to 0.5.

4. Discussion

One of the most significant findings is that the cost of the current 40 per cent P reduction target (at least A$1 billion) is far beyond the reach of existing environmental budgets. Over the period 2002-2009, A$18.8 million funding was allocated to improvement of water quality for the Gippsland Lakes (www.gippslandlakestaskforce.vic.gov.au). Further, increasing the budget sufficiently to achieve 40 per cent would not be cost-effective and would involve substantial social, economic and political challenges associated with land-use changes away from agriculture. The analysis points to the need for managers and policy makers to reconsider the 40 per cent P reduction target.

The results for P reduction scenarios up to 20 per cent provide a basis to develop a stronger business case for higher public investment in the Lakes. Up to 20 per cent P reduction could be achieved cost-effectively. The decision about whether 20 per cent P reduction will give sufficient environmental benefit is one that needs further discussion by governments, agencies and stakeholders.

The analysis indicates that achievement of sufficient changes in land management to deliver substantial reductions in emissions is very challenging and expensive. This reinforces previous conclusions of Monaghan et al. (2007) that reliance on voluntary adoption of CRPs is unlikely to deliver changes in management practices at the scale required to have large environmental impacts without large payments or effective enforcement of regulations. (Here we estimate that the level of reduction in P exports that is possible at minimal public expense is only 4 per cent.) A large number of impediments to adoption of new conservation practices by land managers have been identified (Pannell et al., 2006; Botha and Parminter, 2006) including cost, compatibility with the current farming systems, complexity of new systems, difficulties in trialling new systems and uncertainty about their performance. These impediments explain the high difficulty and expense of achieving substantial practice change for environmental benefits. The study highlights the importance of explicitly considering likely adoption of management changes at an early stage in the decision making process for environmental projects (as emphasised by Pannell et al., 2006). In practice, this rarely occurs (e.g. Simpson, 2010; Stoate et al., 2009).

Another key factor considered in this analysis that is often neglected when establishing environmental targets and developing projects is the relationship between the works to be undertaken and environmental outcomes. This, together with the adoptability of the required works, determines the realistic level of project costs. Environmental goals for the Gippsland Lakes (40 per cent nutrient reductions) were set on a partial biophysical basis without consideration of costs. The same could be said for many other natural assets including, for example, Chesapeake Bay – also with a target 40 per cent nutrient reductions (Simpson, 2010), the Great Barrier Reef – 50 per cent (State of Queensland, 2009), the Waikato River, New Zealand – 50 per cent (Vant and Petch, 2006) and Lake Rotorua, New Zealand – 70 per cent (Kerr et al., 2007). For the Gippsland Lakes, the upfront cost of achieving official targets for this one asset (close to A\$600 million) greatly exceeds total national annual expenditure on this type of program for all assets. We suspect that other assets with similarly large catchments (e.g., Chesapeake Bay - 165,000 km^2 catchment area, Great Barrier Reef - 424,000 km^2 catchment area) also have targets that are highly inconsistent with current budgets. This lack of financial realism in setting environmental targets makes it impossible to implement a rational approach to prioritising environmental investment, and likely reduces the environmental outcomes that are achievable for the available budget. Consideration of realistically costed scenarios would enable a more informed dialogue about what is environmentally acceptable, technically feasible, politically realistic and cost-effective to achieve.

As well as adoption, feasibility and cost, other commonly neglected factors that have been considered in the determination of cost-effectiveness in this analysis include: socio-political risks, the requirement for long-term maintenance funding beyond the initial project phase, the risk of not obtaining the required long-term funding, and time lags between undertaking works and environmental benefits. These are all included as standard elements of the INFFER process for assessing the cost-effectiveness of environmental projects. INFFER's facilities to test the internal consistency of environmental projects and to advise on appropriate delivery mechanisms for a project were also valuable in this study.

Based on this experience, we believe that a similar comprehensive analytical approach would be very valuable for other large, complex environmental assets affected by poor water quality. For example, decisions about the Great Barrier Reef have lacked a comprehensive analysis of the agricultural and environmental trade-offs that are required to reduce sediment and nutrient inputs sufficiently to halt the Reef ecosystem decline. It is recognised that adoption needs to be high across all industries operating in its catchment, and that resources need to be targeted at priority areas and actions to achieve environmental outcomes cost-effectively (Waterhouse et al., 2010), but analysis to underpin policy and project design has been very partial.

The Chesapeake Bay Program in the USA is one of the world's leading nutrient reduction programs. It has an ambitious nutrient reduction goal of 40 per cent and requires co-operation from six states, Washington D.C. and the Federal government (Simpson, 2010). The program has set quantitative goals, underpinned by water quality modelling to identify major contributing areas and developed quantitative CRP tracking and crediting, monitoring and reporting on progress (Simpson, 2010). It has placed much less emphasis on outlining costs associated with landholder adoption at the scale required to have an impact. It has yet to consider socio-political risks, including the power of the farm lobby and increasing urbanisation. In each of these cases, a comprehensive INFFER-style analysis would provide a stronger basis for more informed debate about appropriate targets and budgets.

The analysis for the Gippsland Lakes highlighted a number of important knowledge gaps, including: lack of data on the effectiveness of CRPs, which were only estimated using expert opinion; poorly understood connectivity between farm scale nutrient surpluses and delivery to the Lakes; and general lack of scientific information about nitrogen's impacts and management. Experience from the Chesapeake Bay Program suggests that estimates of CRP effectiveness derived from workshops of technical experts (as we have done) are likely to be overly-optimistic (Simpson, 2010). If so, results for cost per unit of P reduction will be understated.

It was important that the analysis was done in partnership with the peak management body (GLTF), local stakeholders and experts, for reasons including credibility, management of political sensitivities, access to information (both published and unpublished) and "ownership" or acceptance of the results. The technical report from the study has been released on the GLTF website with comment that 'The Taskforce believes this work provides the most transparent and robust framework to justify future cost-effective public investment that has been done to date and looks forward to further improving components of the analysis in the future'.

5. Conclusion

A 40 per cent P reduction goal for the Gippsland Lakes, whilst appearing technically feasible, is well beyond the reach of existing environmental budgets (at a cost of close to $1 billion over 20 years) and far from being cost-effective. Achieving a 20 per cent P reduction appears cost effective, requiring only modest levels of change within agricultural systems. The major implications of this work for agriculturally-induced diffuse-source pollution problems include the need to be clear about what environmental assets are being protected, the need for feedback between goal setting and program costs, and consideration of factors such as the impacts of works on

environmental condition, the levels of landholder adoption of changed land-management practices required to achieve particular environmental targets and the costs of achieving those level of adoption. The analysis provides a basis for more informed discussion about the environmental outcomes that can be achieved with limited budgets and also about agricultural production and environmental trade-offs involved in reducing diffuse-source nutrient pollution. Results are relevant to comparable water-quality programs worldwide.

Acknowledgements

References

Anonymous, (2007). Soil erosion management plan for the West Gippsland CMA region. Department of Primary Industries, Bendigo.

Baker, D.B., 1993., The Lake Erie Agroecosystem Program: water quality assessments. Agriculture Ecosystems and Environment 46, 197-215.

Botha, N., Parminter, T., 2006. The suitability of available technologies and response to government policies for improving waterway quality. 12th International Symposium on Society and Resource Management. Global challenges and local responses. University of British Columbia, 3-8 July, Vancouver, BC, Canada, page 26.

Brooke, A., Kendrick, D., Meeraus, A., Raman, R., 2008. GAMS—A user's guide. GAMS Development Corporation, Washington D. C.

Cherry, K.A., Shepherd, M., Withers, P.J.A., Mooney, S.J., 2008. Assessing the effectiveness of actions to reduce nutrient loss from agriculture: a review of methods. Science of the Total Environment 406, 1-23.

Connor, J., Hatton MacDonald, D., Morrison, M., Cast, A., 2009. Evaluating policy options for managing diffuse source water quality in Lake Taupo, New Zealand. Environmentalist 29(4), 348-359.

Department of Natural Resources and Environment, 2002. Gippsland Lakes Future Directions and Action Plan. The State of Victoria. www.gippslandlakestaskforce.vic.gov.au/glfdap/future1.pdf (accessed 30 April 2010).

Department of Primary Industries, 2006. Gippsland region. http://www.dpi.vic.gov.au/dpi/nrenti.nsf/93a98744f6ec41bd4a256c8e00013aa9/7f8481a23f3f51bbca257602001a9ab9/$FILE/ATTLILLB/Gippsland.pdf (accessed 25 May 2010).

Department of Sustainability and Environment, 2003. Gippsland Lakes Ramsar site: Strategic Management Plan. Department of Sustainability and Environment, Melbourne.

Dowd, B.M., Press, D., Los Huertos, M., 2008. Agricultural nonpoint source water pollution policy: The case of California's Central Coast. Agriculture, Ecosystems and Environment 128, 151-161.

Frontline Systems, 2009. Risk Solver Platform version 9.5. Frontline Systems, Incline Village.

Gourley, C.J.P., Powell, J.M., Dougherty, W.J., Weaver, D.M., 2007. Nutrient budgeting as an approach to improving nutrient management on Australian dairy farms. Australian Journal of Experimental Agriculture 47, 1064-1074.

Grayson, R., 2006. Prioritising nutrient reduction for the Gippsland Lakes and catchments. Part 1 – Loads and sources. Catchment to Sea. Pty. Ltd. www.gippslandlakestaskforce.vic.gov.au/publications/inlakeresearch/CSIRO1998.pdf (accessed 4 May 2010).

Grayson, R.B., Argent, R.M., 2002. A tool for investigating broad-scale nutrient and sediment sources from the catchments of the Gippsland Lakes. CEAH Report 1/02 45p. www.gippslandlakestaskforce.vic.gov.au/publications/inlakeresearch/CSIRO1998.pdf (accessed 4 May 2010).

Gunningham, N., Sinclair, D., 2002. Leaders and laggards. Next generation environmental regulation. Greenleaf Publishing Ltd. Sheffield, U.K. 224pp.

Hancock, G., Wilkinson, S. and Read, A., 2007. Sources of sediment and nutrients to the Gippsland Lakes assessed using catchment modelling and sediment tracers. CSIRO Land and Water Science Report 70/07. ISSN 1883-4563. www.gippslandlakestaskforce.vic.gov.au/publications/inlakeresearch/CSIRO1998.pdf (accessed 4 May 2010).

Harris, G., Batley, G., Webster, I., Molloy, R., Fox, D., 1998. Gippsland Lakes Environmental Audit: Review of water quality and status of the aquatic ecosystems of the Gippsland Lakes. CSIRO Environmental Projects Office. www.gippslandlakestaskforce.vic.gov.au/publications/inlakeresearch/CSIRO1998.pdf (accessed 4 May 2010).

Kay, P., Edwards, A.C., Foulger, M., 2009. A review of the efficacy of contemporary agricultural stewardship measures for ameliorating water pollution problems of key concern to the UK water industry. Agricultural Systems 99, 67-75.

Kerr, S, Rutherford, K., Lock, K., 2007. Nutrient trading in Lake Rotorua: goals and trading caps. Motu Working Paper 07-08, Wellington, New Zealand.

Kyllmar, K., Carlsson, A., Gustafson, A., Ulén, B., Johnsson, H., 2006. Nutrient discharge from small agricultural catchments in Sweden, characterisation and trends. Agriculture, Ecosystems and Environment 115, 15-26.

Ladson, A., Tilleard, J., 2006. BMPs for reducing phosphorus load to the Gippsland Lakes. Report on findings from expert panel. Monash University/Moroka Pty. Ltd. www.gippslandlakestaskforce.vic.gov.au/publications/inlakeresearch/CSIRO1998.pdf (accessed 4 May 2010).

León, L.F., Booty, W.G., Bowen, G.S., Lam, D.C.L., 2004. Validation of an agricultural non-point source model in a watershed in southern Ontario. Agricultural Water Management 65, 59-75.

Logan, T.J., 1993. Agricultural best management practices for water pollution control: current issues. Agriculture, Ecosystems and Environment 46, 223-231.

Maticic, B., 1999. The impact of agriculture on ground water quality in Slovenia: standards and strategy. Agricultural Water Management 40, 235-247.

Mitchell, M., (1996). *An introduction to genetic algorithms*, MIT Press, Cambridge.

Monaghan, R.M., Hedley, M.J., Di, H.J., McDowell, R.W., Cameron, K.C., Ledgard, S.F., 2007. Nutrient management in New Zealand pastures – recent developments and future issues. New Zealand Journal of Agricultural Research 50, 181-201.

Monaghan, R.M., Carey, P.L., Wilcock, R.J., Drewry, J.J., Houlbrooke, D.J., Quinn, J.M., Thorrold, B.S., 2009. Linkages between land management activities and stream water quality in a border dyke-irrigated pastoral catchment. Agriculture, Ecosystems and Environment 129, 201-211.

National Research Council, 2008. Mississipi river water quality and the clean water act, progress, challenges and opportunities. National Academies Press, Washington D.C.

National Research Council, 2009. Nutrient control actions for improving water quality in the Mississipi river basin and the northern Gulf of Mexico. National Academies Press, Washington D.C.

Pannell, D.J., 2008. Public benefits, private benefits and policy intervention for land-use change for environmental benefits. Land Economics 84, 225-240.

Pannell, D.J., Marshall, G.R., Barr, N., Curtis, A., Vanclay, F., Wilkinson, R., (2006). Understanding and promoting adoption of conservation practices by rural landholders. Australian Journal of Experimental Agriculture 46, 1407-1424.

Pannell, D.J., Roberts, A.M., Park, G., Curatolo, A., Marsh, S., Alexander, J., 2009. INFFER (Investment Framework For Environmental Resources), INFFER Working Paper 0901, University of Western Australia, Perth. http://cyllene.uwa.edu.au/~dpannell/dp0901.htm (accessed 25 July 2010).

Rice, R.W., Izuno, F.T., Garcia, R.M., 2002. Phosphorus load reductions under best management practices for sugarcane cropping systems in the Everglades agricultural area. Agricultural Water Management 56, 17-39.

Ruhl, J.B., 2000. Farms, their environmental harms, and environmental law. Ecology Law Quarterly 27(2), 263-349.

Sargant, J., (2009). East Gippsland soil erosion management plan. Department of Primary Industries, Bairnsdale. ISBN 978 1 74217 190 6.

Sharpley, A.N., Gburek, W.J., Folmar, G., Pionke, H.B., 1999. Sources of phosphorus exported from an agricultural watershed in Pennsylvania. Agricultural Water Management 41, 77-89.

Shortle, J.S., Horan, R.D., 2001. The economics of nonpoint pollution control. Journal of Economic Surveys 15, 255–289.

Simpson, T., 2010. Chesapeake Bay catchment management: lessons learned from a collaborative, science-based approach to water quality restoration. Chapter 18. Handbook of Catchment Management, 1st Edition. Eds. R.C. Ferrier and A. Jenkins. Blackwell Publishing.

State of Queensland, 2009. Reef Water Quality Protection Plan. Department of the Premier and Cabinet. Reef Water Quality Plan Protection Secretariat, 100 George St Brisbane (www.reefplan.qld.gov.au, accessed May 2010)

Stoate, C., Báldi, A., Beja, P., Boatman, N.D., Herzon, I., van Doorn, A., de Snoo, G.R., Rakosy, L., Ramwell, C., 2009. Ecological impacts of early 21st century agricultural change in Europe – a review. Journal of Environmental Management 91, 22-46.

Summers, R.N., Van Gool, D., Guise, N.R., Heady, G.J., Allen, T., 1999. The phosphorus content in the run-off from the coastal catchment of the Peel Inlet and

the Harvey Estuary and its associations with land characteristics. Agriculture, Ecosystems and Environment 73, 271-279.

Vant, W., Petch, T., 2006. Nitrogen and phosphorus in our freshwaters: effects and targets. Environment Waikato, Hamilton, New Zealand.

Waterhouse, J., Grundy, M., Gordon, I., Brodie, J., Eberhard, R., Yorkston, H., 2010. Managing the catchments of the Great Barrier Reef. Chapter 15. Handbook of Catchment Management, 1st Edition. Eds. R.C. Ferrier and A. Jenkins. Blackwell Publishing. pp. 351-375.

Weersink, A., Jeffrey, S., Pannell, D., 2002. Farm-level modelling for bigger issues. Review of Agricultural Economics 24, 123-140.

Appendix: The Benefit: Cost Index

Consistent with a Benfit: Cost Ratio, the broad design of the Benefit: Cost Index is as follows:

$$BCI = \frac{(\text{Asset value or significance}) \times (\text{Proportional impact of project on that value})}{\text{Cost}} \qquad (A.1)$$

The formula is designed to allow comparison of projects of different types, scales, and durations. It facilitates this because, respectively, it expresses benefits in a common unit of measure (a score standardised against an asset of high national significance), it divides benefits by costs to allow comparison of relative cost-effectiveness, and it discounts future benefits and costs to calculate their present values. The higher the value of the BCI, the higher the priority of the project.

Specifically, the BCI formula is as follows:

$$BCI = \frac{V \times W \times F \times A \times B \times P \times G \times DF_B(L) \times 20}{C + PV(M)} \qquad (A.2)$$

where

V = value of the asset
W = multiplier for impact of works
F = multiplier for technical feasibility risk
A = multiplier for adoption
B = multiplier for adverse adoption
P = multiplier for socio-political risk
G = multiplier for long-term funding risk
DF_B = discount factor function for benefits, which depends on L
L = lag until benefits occur (years)
C = short-term cost of project
PV = present value function
M = annual cost of maintaining outcomes from the project in the longer term.

Details about each of the variables is provided below.

Asset value (*V*): *V* is estimated as a score out of 100 that represents the value of this asset, assuming that the asset is in good condition. The scoring range is calibrated such that a score of 100 corresponds to an asset of high national significance (such as the Gippsland Lakes).

Impact of works (*W*): *W* represents the proportional increase in future asset value that would result if the project was fully implemented (i.e. assuming that it is fully adopted) compare to if it wasn't. *W* is measured as a proportion of the total value of the asset (in good condition).

Technical feasibility (*F*): *F* is the probability that the benefits generated would be as large as specified in *W*. In other words, it is the probability that benefits will not be significantly less than *W*. Like all the probabilities included in the formula, *F* is expressed as the probability of success, rather than of failure, so that the formula provides the expected value of benefits.

Private adoption of works and actions (*A*): *A* is the probability that the on-ground works and actions specified in the project will actually be adopted, assuming that the project is fully funded and the project's delivery mechanisms are implemented.

Preventing adoption of adverse practices (*B*): *B* is the probability that the project will not fail due to adoption of adverse works or actions, despite efforts by the project to prevent that adoption from occurring.

Socio-political risks (*P*): *P* represents the probability that other socio-political factors will *not* derail the project. This includes the risk of non-cooperation by other organisations and the impacts of social, administrative or political constraints. The latter can include resistance to the project at the political level, bureaucratic approvals that would be needed, or opposition by local government.

Long-term funding risks (*G*): *G* represents the probability that essential long-term funding will be available to continue to maintain the benefits generated by this project, or to complete the essential works commenced by this project.

Time lag to benefits (*L*): *L* is the expected time lag in years until the desired bio-physical outcomes would be achieved. It represents the earliest time when a large proportion of the benefits will occur.

Discount factor ($DF_B(L)$): Benefits that occur further into the future are a lower priority than similar benefits that occur rapidly. This is captured through the use of discounting. The discount factor is calculated as follows:

$$DF_B(L) = 1/(1.05)^L \tag{A.3}$$

This assumes that the real discount rate (net of inflation) is 0.05.

Up-front costs (*C*): *C* is the sum of direct costs that will be incurred within the immediate time frame of this project – assumed to be three to five years.

Ongoing or maintenance costs (*PV*(*M*)): Some costs may be incurred each year in the long term, such as monitoring and evaluation, or enforcement costs, or ongoing compensation payments. The annual total of these maintenance costs is *M*. To make them comparable to the up-front costs, we need to express them as a discounted present value.

The BCI formula is designed so that it behaves similarly to a BCR, in that a BCI exceeding 1.0 is desirable. This is achieved by including the 20 factor at the end of equation (2) to scale the results appropriately, based on an assumption that a *V* score of 100 corresponds to a total dollar value of $2 billion.

Table A.1 outlines each of the parameters used for the BCI calculation in each scenario.

Table A.1: Parameters used to calculate the INFFER Benefit: Cost Index (BCI).

Parameter[A] Scenario	*V*	*W*	*F*	*A*	*B*	*P*	*G*	*DF$_B$*	*L*	*C*	*M*
1	100	0.50	0.82	0.4	1	0.37	0.5	0.38	20	584.1	38.3
2		0.38	0.85	0.6				0.48	15	116.8	9.9
3	100	0.25	0.89	0.7	1	0.50	0.6	0.61	10	54.0	2.5
4	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	10.3	0.6
5	100	0.13	0.91	0.7	1	0.62	0.7	0.61	10	9.9	1.2
6	100	0.23	0.89	0.7	1	0.62	0.7	0.61	10	24.9	8.4
7	100	0.28	0.88	0.7	1	0.62	0.7	0.61	10	50.0	8.6
8	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	9.8	0
9	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	24.7	0
10	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	49.3	0
11	100	0.21	0.89	0.7	1	0.62	0.7	0.61	10	103.2	8.3
12	100	0.16	0.91	0.7	1	0.62	0.7	0.61	10	20.9	0.87
13	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	14.1	1.03
14	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	5.2	1.03
15	100	0.20	0.90	0.7	1	0.62	0.7	0.61	10	43.9	1.57
16	100	0.13	0.92	0.7	1	0.62	0.7	0.61	10	43.9	1.57
17	500	0.50	0.82	0.4	1	0.37	0.5	0.38	20	584.1	38.3
18	50	0.25	0.89	0.7	1	0.62	0.7	0.61	10	54.0	2.5