The Sustainability and Cost-Effectiveness of Water Storage Projects on Canterbury Rivers: The Opihi River Case

Edward J. S. Hearnshaw
Lecturer in Ecological Economics, Faculty of Commerce, Lincoln University
e-mail: edward.hearnshaw2@lincolnuni.ac.nz

Ross Cullen
Professor of Resource Economics, Faculty of Commerce, Lincoln University

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Edward J. S. Hearnshaw[1] and Ross Cullen[2]
[1]Lecturer in Ecological Economics, Faculty of Commerce, Lincoln University
Email: edward.hearnshaw2@lincolnuni.ac.nz
[2]Professor of Resource Economics, Faculty of Commerce, Lincoln University

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Water is the classic common property resource.
No one really owns the problem.
Therefore, no one really owns the solution.

Ban Ki-moon

Abstract:

There is an increasing demand for water resources in the Canterbury region. The impact of this demand has lead to unacceptable minimum river flows, which has resulted in adverse affects to river ecology. In an effort to resolve this problem water storage projects have gained considerable attention. However, in order to consider all values of the impact of water storage projects, a systematic way of implementing an ecosystem services approach is developed. This ecosystem services approach coupled with various appropriate analytical methods are developed for the purposes of evaluating the cost-effectiveness of water storage projects and the sustainability of river systems impacted by water storage projects. For the purposes of evaluating the cost-effectiveness of water storage projects it is argued that cost utility analysis should be applied through an ecosystem services index, which is constructed from the aggregation of normalized indicators that represent each ecosystem service and preferential weights for each ecosystem service. The evaluation of sustainability is considered both according to its weak and strong definitions. Weak sustainability is evaluated by a non-declining ecosystem services index over time. Strong sustainability is evaluated by the elicitation of threshold levels or safe minimum standards where an ecosystem service, as represented by an indicator, should not pass below. These analytical methods developed are subsequently applied to the Opihi River, which is a river system located in Canterbury that has been hydrologically modified and impounded by the Opuha Dam scheme. The application of the analytical methods to the Opihi River provides a few preliminary results. Further data collection is required to fully determine the cost-effectiveness of the Opuha Dam and the sustainability of the Opihi River impacted by the dam scheme.

Keywords: Cost utility analysis, ecosystem services, ecosystem services index, indicators, sustainability, water storage projects.

1.0 Introduction

In recent times, there has been an increased demand from the agricultural sector in New Zealand for abstracting water resources for irrigation. This demand is particularly strong in the Canterbury region. This is understandable as Canterbury experiences high levels of evaporation through dry summers and yet has the potential for its agricultural land to be extensively irrigated. Nevertheless, the supply of water is scarce, making it essential that cost-effective management and the sustainable use of water resources is appropriately considered.

Irrigation enables farmers to intensify their agricultural operations. Irrigated farms can generate three times the farm income of non-irrigated farms (Harris Consulting, 2006). Intensification can result in improved profitability either by greater levels of agricultural production through increased stocking rates with existing land use practised or a change by farmers toward more productive land uses (*e.g.* sheep farming/mixed cropping to dairy farming/vegetable production). The effects of irrigation through the abstraction of water from rivers and groundwater aquifers in Canterbury are increasingly evident as much land use intensification has occurred over the past 20 years (Parkyn & Wilcox, 2004). Accordingly, today irrigation is viewed as a vital component of the region's land-based economy. However, to meet ever-increasing (or seemingly insatiable) demand for freshwater to irrigate agricultural land, it is necessary to ensure a reliable (and increased) freshwater supply is sustained for the region. The reliability of freshwater supply is important. The less reliable the freshwater supply the less viable land use intensification becomes. Hence, an unreliable freshwater supply can cause increasing uncertainty in the agricultural planning of farmers and the subsequent adoption of conservative and potentially inefficient agricultural practices (Canterbury Regional Council, 1995).

While much irrigation in Canterbury uses run-of-river surface water management schemes, there is a realization that much of this water has reached its maximum allocation limits while retaining acceptable minimum river flows needed to sustain aquatic health (Canterbury Mayoral Forum, 2009; Harris Consulting, 2009). In fact, the degradation of river systems resulting in unacceptable minimum river flows represents one of the most serious environmental problems worldwide (Food and Agricultural Organization, 2000). This realization has led to increased interest in water storage projects, and in particular schemes that impound rivers by way of the construction of dams. Dams make it possible to regulate, stabilize and augment minimum river flows downstream of the dam scheme and store water upstream through the creation of artificial lakes and reservoirs (Graf, 2006). With the potential of reliable and increased freshwater supply resultant from these water storage projects, it is possible for farmers to irrigate their farms and intensify land use in an attempt to maximize farm production and profitability. This potential for improved profitability on agricultural land can lead to further (indirect) benefits to rural communities through greater opportunities for employment (World Commission on Dams, 2000; Harris Consulting, 2009).

However, while the impoundment of rivers through dam construction can result in significant benefits to farmers and rural communities, it also can come at a 'cost', especially to river ecology. For example, Losos *et al.* (1995) found that dams and other water storage projects have resulted in more degradation of threatened species and their habitats than any other activity utilizing environmental resources. Indeed, from a historical perspective, it appears that the impetus for constructing water storage projects has been, by and large, with short-term economic returns and not in improving the actual functioning and aquatic health of river systems (Dyson *et al*., 2003). Moreover, in addition to the problems of dams, scientists have long recognized the negative impact of land use intensification on rivers. Land use intensification, especially the conversion of low-intensity sheep farming to high-intensity dairy farming, often leads to a substantial increase in the application of fertilizers (Harris Consulting, 2006). The increased levels of nutrients (*e.g*. nitrates) applied with intensified agricultural practices can, through surface runoff, pollute rivers. This increased concentration of pollutants in rivers can degrade water quality and the ecology of the river through the excessive primary production of algae. But, even if water quality is not degraded following land use intensification, the abstraction of water can still degrade aquatic health if they do not sustain acceptable minimum river flows (Dyson *et al*., 2003).

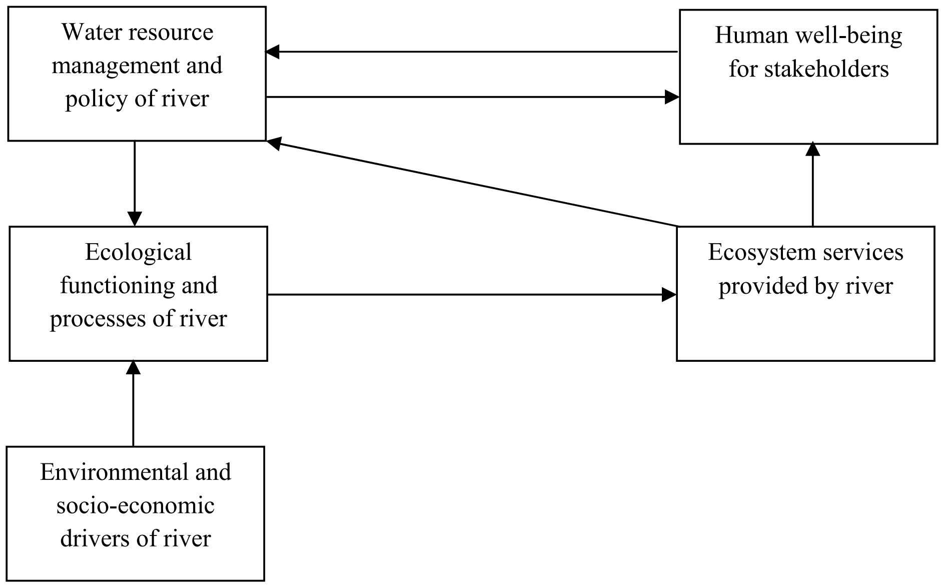
Given the potential positive and negative impacts from water abstraction and river impoundment, water availability and water storage are now critical issues for local and regional government in the Canterbury region (Canterbury Mayoral Forum, 2009). The 'wickedness' (Rayner, 2006; Frame & Russell, 2009) or complexity of Canterbury's water allocation and water storage problems has in recent times become very apparent and resulted in a fundamental shift in water resource management towards an integrated assessment, where the complex nature of water resources to human well-being is inherently recognized (Food & Agricultural Organization, 2000). This is achieved as integrated water resource management entails that all the values provided by river systems should be considered through including the perspectives of all stakeholders (Bouwer, 2000), and that issues of sustainability and cost-effectiveness be evaluated when considering management schemes.

Scarce water resources today are valued for a multitude of reasons including highly important non-use values, such as conservation (*e.g*. biodiversity) and cultural values (*e.g*. Māori spirituality) (Canterbury Mayoral Forum, 2009). For example, when considering the perspectives of all stakeholders, a fisherman might value a river by the abundance of trout fish, a Ngāi Tahu member by its abundance of mahika kai (*i.e*. a term for food resources gathered by Māori using traditional methods), a farmer by its capacity to abstract water for irrigation purposes, an environmentalist by the presence of a threatened bird species, and a water supply company by the treatment costs required to produce safe drinking water. Consequently, for water resource management to be integrated it is no longer acceptable to only consider those tangible use values that relate to direct water consumption (Cortner & Moote, 1994; Jewitt, 2002; Frame & Russell, 2009). Quite simply, conservation and cultural values cannot be left out or disproportionately represented relative to economic development aspirations (*e.g*. irrigation).

The need to consider and integrate all values has lead to the consideration of evaluating water storage projects using an ecosystem services approach, as this approach offers considerable transparency to all values provided by river systems. The ecosystem services approach has been popularized by some notable studies (*e.g*. Costanza *et al*., 1997), including the landmark *Millennium Ecosystem Assessment* (Capistrano *et al*., 2006). Specifically, ecosystem services are the collection of goods and services provided by ecosystems that benefit human well-being (Daily, 1997;

National Research Council, 2005). Consequently, ecosystem services are, in effect, the connection between humans and ecosystems (Wilson *et al*., 2005; Kontogianni *et al*., 2010). *Figure 1* indicates the pivotal role ecosystems services play in connecting human well-being and the ecological functioning of the river system.

Figure 1: Ecosystem services act as the connection between human well-being and ecological functioning (adapted from Wilson *et al*., 2005).



To date, while many researchers have recognized the potential of the ecosystem services approach for evaluating the multiple values provided by ecosystems, the relevant literature reveals that only some ecosystem services are regularly considered (*e.g*. **Recreational Values**) (de Groot *et al*., 2009). Moreover, despite the recognition of the ecosystem services approach for apt project evaluation, there are few studies that have considered the impact and change in ecosystem services provided after the construction of a water storage project (Hoeinghaus *et al*., 2009). One underlying reason for the uneven distribution of research into ecosystem services and their limited use for project evaluation is that there is still much debate on how to apply and implement the approach. For example, one critical debate yet to be fully resolved is how best to define the set of ecosystem services and make a clear distinction between ecological functioning, ecosystem services and benefits to human well-being (Boyd & Banzhaf, 2007; Balmford *et al*., 2008; Fisher *et al*., 2009). There have been various classifications of ecosystem services developed, each with arguments to support the use of the particular classification devised (Capistrano *et al*., 2006; Barkmann *et al*., 2008). However, despite the various classifications available and concerns about their implementability, according to Raymond *et al*. (2009) the set of ecosystem services established by the *Millennium Ecosystem Assessment* remains the most recognizable and well-developed. As such, the classification developed in the *Millennium Ecosystem Assessment* is applied in this paper.

Specifically, in the *Millennium Ecosystem Assessment* classification, there is a taxonomy of four classes of ecosystem services. However, only three of these four classes are directly considered in this paper. This is because one class referred to as ‘supporting ecosystem services’ (*e.g.* **Nutrient Cycling**, **Photosynthesis**, **Primary Production, Pollination**) are services that reflect ecological processes that produce other ecosystem services rather than provide benefit to human well-being directly (Barkmann *et al*., 2008; Layke, 2009). For example, **Photosynthesis** is an ecological process that allows for plant growth. It is not an ecosystem service that provides direct benefit to human well-being. Rather, **Photosynthesis** supports and produces various other ecosystem services that do provide benefit to human well-being including the provisioning ecosystem services **Food** and **Fibre** (Boyd & Banzhaf, 2007). The three classes examined are ‘provisioning ecosystem services’ which provide use benefits through goods that are obtained from the ecosystem, ‘regulating ecosystem services’ which provide benefits through controlling and regulating various ecological functions, and ‘cultural ecosystem services’ which provide non-material benefits including non-use benefits (*Table 1*). Despite that supporting ecosystem services are not evaluated in this paper, it is important to recognize that of the various classifications devised only the *Millennium Ecosystem Assessment* classification and one other similar classification developed by de Groot (2006) acknowledge that some ecosystem services ‘support’ and produce other ecosystem services. Not recognizing this aspect can lead, amongst other things, to double counting when the ecosystem services approach is implemented for project evaluation (Hein *et al*., 2006; Dominati *et al*., 2010).

Table 1: The set of ecosystem services that an ecosystem may provide (adapted from Capistrano *et al*., 2006).

Class	Ecosystem services	Description of ecosystem service
Provisioning ecosystem services	**Food**	Ecosystem supplies food produce (*e.g.* fish, grains, wild game, fruits)
	Fibre	Ecosystem supplies extractable renewable raw materials for fuel & fibre (*e.g.* fuelwood, logs, fodder)
	Freshwater Supply	Ecosystem supplies freshwater for use & storage
	Biological Products	Ecosystem supplies biological resources that can be developed into biochemicals for medicinal/commercial use
	Abiotic Products	Ecosystem supplies extractable non-renewable raw materials such as metals and stones for commercial use
Regulating ecosystem Services	**Climate Regulation**	Ecosystem regulates air temperature and precipitation and acts as a source of and sink for greenhouse gases
	Disease Regulation	Ecosystem regulates the abundance of pathogens
	Water Regulation	Ecosystem regulates hydrological flows (*i.e.* surface water runoff, groundwater recharge/discharge)
	Water Purification	Ecosystem purifies & breaks down excess nutrients in water
	Pest Regulation	Ecosystem regulates abundance of invasive or pest species
	Erosion Control	Ecosystem controls potential biological catastrophes & stabilizes against erosion, thus, retaining soils
	Natural Hazard Regulation	Ecosystem regulates and protects against extreme natural events (*i.e.* floods or droughts)
Cultural ecosystem services	**Educational Values**	Ecosystem provides opportunities for non-commercial uses (*e.g.* archaeological values, knowledge systems).
	Conservation Values	Ecosystem provides existence values for species including important values relating to biodiversity
	Aesthetic Values	Ecosystem provides aesthetic qualities
	Spiritual Values	Ecosystem provides spiritual and inspirational qualities
	Recreational Values	Ecosystem provides opportunities for recreational uses

Given the significance of water storage projects and usefulness of the ecosystem services approach for integrated water resource management, in this paper an *ex-post* evaluation of a water storage project is undertaken that considers the cost-effectiveness of the water storage project and the sustainability of the river system impacted by the management scheme. Specifically, the river system investigated is the Opihi River located in South Canterbury, which is ideal as it has been hydrologically modified and impounded by the Opuha Dam, which was commissioned to store water and augment minimum river flows on the Opihi River primarily for irrigation purposes. The remainder of the paper is structured as follows. In *Section 2* the appropriate analytical methods for evaluating water storage projects is discussed that can overcome difficulties of pricing ecosystem services. This leads to the recognition that cost utility analysis is appropriate, as it is able to integrate the set of ecosystem services provided into a (non-monetary) utility index. Then, it is demonstrated how the ecosystem services index can be used to determine the cost-effectiveness of water storage projects and the sustainability of river systems impacted by water storage projects. With these methodological developments considered, *Section 3* provides an overview of the Opihi River and the Opuha Dam scheme. In *Section 4* a few preliminary results are shown from the analytical methods advocated for the determination of the cost-effectiveness of the Opuha Dam and the sustainability of the Opihi River impacted by the dam scheme. In *Section 5* a discussion is developed that considers the potential for evaluating water storage projects *ex-ante* using the ecosystem services approach devised.

2.0 Evaluations of Water Storage Projects

Despite the need for (*ex-post*) evaluations of water storage projects, such evaluations of their impacts and performance are rarely undertaken despite that such projects are costly (World Commission on Dams, 2000). This signals an obvious failure of not adequately monitoring the performance, impacts and return on investment of water storage projects, which is critical for investigating the cost-effectiveness of water storage projects and the sustainability of river systems impacted by these management schemes. In recognizing the limited efforts to evaluate water storage projects, the World Commission on Dams (2000) performed some simple economic analysis and found that dams often fail to reach their projected economic estimates. For example, cost analysis was performed on various dam schemes and it was found that the actual construction costs were, on average, 56 per cent over projected construction costs. Similar observations have also been recognized by Scudder (2005) and Young (2005), which suggest a systematic bias of underestimating the costs involved with water storage projects.

A focus on cost analysis is, of course, insufficient as it does not consider the benefits available to human well-being. In an effort to include these benefits, cost benefit analysis is conventionally offered as the appropriate analytical method for project evaluation by neoclassical economists. Specifically, cost benefit analysis evaluates projects by collapsing aggregated costs and benefits into a single monetary metric. Accordingly, water storage projects that are deemed economically justifiable are those with a positive net benefit or cost-benefit ratio less than one. However, an immediate difficulty with cost benefit analysis is that it often fails to integrate all values provided by ecosystems as many project evaluations have considered only those tangible use values supplied (*e.g.* **Food** and **Freshwater Supply** ecosystem service) that are easily captured and quantified in monetary terms through the availability of market prices (Young *et al*., 2004; Farber *et al*., 2006). However, more than 80 per cent of ecosystem services lack functioning markets so that their value is not captured by market prices (de Groot *et al*., 2002; Swinton *et al*., 2007). For all intents and purposes, missing markets for these less tangible ecosystem services (*e.g*. **Erosion Control**, **Spiritual Values**) often leave them undervalued or erroneously given an implicit value of zero (Loomis *et al*., 2000; Navrud, 2001; Dyson *et al*., 2003; National Research Council, 2005; Barkmann *et al*., 2008).

The presence of market externalities when evaluating river ecosystems and water storage projects, as a result of missing markets for many ecosystem services, inevitably results in inefficiencies and the misallocation of scarce water resources (Swanson, 1995; National Research Council, 2005). In order to address this problem, environmental economists have devised a number of non-market valuation methods (*e.g*. contingent valuation, choice modelling) designed to overcome pricing difficulties in the absence of actual markets for less tangible values. However, while these non-market valuation methods are theoretically advanced, they usually require a painstaking amount of effort in gathering information about the ecosystem from a large sample population of affected stakeholders. This makes non-market valuation methods costly, labour intensive and time-consuming to undertake (Gowdy, 1997; Baskaran *et al*., 2010).

In an effort to reduce these impracticalities of non-market valuation, the benefit transfer method is often employed. Specifically, this method uses monetary values obtained from previous non-market valuations and applies these values obtained for the purposes of valuing less tangible values for the ecosystem evaluated. However, in using the benefit transfer method, it is inherently assumed that ecosystems can be treated as if they are much alike. But, evidence suggests that maintaining such an assumption only leads to inaccurate monetary values being calculated and the potential degradation of ecological functioning and the subsequent loss of ecosystem services (Carpenter & Brock, 2004; Spash & Vatn, 2006). This makes sense because ecosystem services derive from ecological processes through the “complex interactions between biotic and abiotic components of ecosystems” (De Groot *et al*., 2002). Hence, ecosystems are complex systems, which make them highly variable and unpredictable systems with dynamics that are path dependent and location-specific, so that their unique environmental, socio-economic and management history matters (Naidoo *et al*., 2009). Consequently, it is often not possible to compare monetary values obtained from one ecosystem and apply these to other ecosystems even if they are of similar type (National Research Council, 2005). This recognition of the complexity of ecological processes involved in deriving ecosystem services further underscores the need for efforts to be placed in measuring ecosystem services that are directly connected to human well-being (*i.e*. provisioning, regulating and cultural ecosystem services) and not to ecological functions that are rarely understood, difficult to measure and less relevant for policy making (Kremen & Ostfeld, 2005; Kontogianni *et al*., 2010).

The potential for the meaningful monetization of ecosystem services is impractical and can lead to the erroneous measurement of some less tangible values. In fact, given the difficulties of monetization of benefits with cost benefit analysis, water resource managers and policy makers may choose not to value benefits altogether. Instead, they may resign themselves to water resource management governed in an unsystematic way with little transparency or simply resort back to the use of cost analysis (Prato, 1999). And indeed, evidence suggests that (water resource) managers and policy makers do not attempt to monetize benefits (Gowan *et al*., 2006). Rather, for the most part, representatives of stakeholders in a water resource management setting continue to employ a combination of cost analysis and impact analysis (Stephenson & Shabman, 2001). However, while these analytical methods are no doubt useful, they lack the capacity to indicate the cost-effectiveness of water storage projects or appropriately integrate values for effective policy making, which is a primary task for integrated water resource management.

Fortunately, while not well-known, there are alternative analytical methods to cost benefit analysis (or cost analysis) that are practical and useful (Cullen *et al*., 2001). One of these analytical methods is cost utility analysis, which allows the construction of an outcome (or benefits) function at low cost through the application of expert judgements. Cost utility analysis is attractive because it provides a definitive means of evaluating and aggregating multiple attributes (or values) into a single non-monetary metric as a utility index. The development of a utility index for the outcome function makes cost utility analysis synonymous with multi-criteria analysis. Multi-criteria analysis is an overarching term depicting the set of analytical methods capable of weighting and aggregating multiple attributes together (Munda *et al*., 1994). The capacity of using a utility index for economic analysis is significant as it generally allows all (or almost all) ecosystem services to be considered for project evaluation without having to monetize less tangible values that are more appropriately captured and left in their own terms (Wainger *et al*., 2010).

By cost utility analysis developing a utility index for its analysis, it allows a measure that accounts for both ecosystem status and the preferences for the various ecosystem services provided. This is significant, as cost benefit analysis only considers the latter, yet when evaluating water storage projects it is important to know how the set of ecosystem services have been impacted and what ecosystem services are preferred by all stakeholders involved (Farber *et al*., 2006). Despite the apparent usefulness of cost utility analysis, some neoclassical economists argue that employing expert judgements for the construction of a utility index to be the greatest difficulty with this analytical method (Brent, 2003). Indeed, some neoclassical economists maintain that the determination of social benefits cannot be represented by the 'biased' preferences of a supposedly 'qualified' few. But, recent research suggests that using the judgements and preferences of experts is a legitimate and reasonable approximation of the preferences of all stakeholders (Colombo *et al*., 2009). In fact, expert judgements may outperform the elicitation of the preferences from all stakeholders as affected stakeholders may neither possess sufficient understanding of the 'wickedness' of the water resource problems faced, nor an adequate grasp of the analytical methods employed (Alvarez-Farizo & Hanley, 2006; Barkmann *et al*., 2008).

2.1 Indicators & Ecosystem Services Index

In order to establish a utility index that considers the impacts of ecosystem services from a water storage project, there is a need to measure quantitative changes in ecosystem services over time. To quantitatively measure these changes on ecosystem services, indicators are sometimes used, especially given that most ecosystem services are difficult to measure directly (World Resources Institute, 2008). The use of indicators for investigating the complexity of ecosystems and the services they provide is significant, as indicators are able to "summarize complex information of value to the observer. They condense … complexity to a manageable amount of meaningful information … informing ... and directing our [policy] actions" (Bossel, 1999; p. 8). Despite the usefulness of indicators for representing ecosystem services and informing water resource managers and policy makers, they remain underdeveloped. There are no indicators that are fully agreed upon for the monitoring of each ecosystem service no matter the classification used, though de Groot *et al*. (2009) recently provided a suggested list of indicators that could be employed for representing many of the ecosystem services. One reason why no well-defined list of indicators have been established for the set of ecosystem services is that ecosystem services can be difficult to capture by a single indicator (Layke, 2009). Despite this most economists, when using indicators to represent ecosystem services, have used a single environmental (*i.e*. biophysical) indicator (Lamb *et al*., 2009). This difficulty of equating a single indicator for each ecosystem service is, in part, because while an indicator makes understanding an ecosystem service more manageable, it also can often lead to overly reductionistic interpretations with no connection apparent between ecological functioning and human well-being (Kontogianni *et al*., 2010). Where this is so, the result is that the ecosystem service provided is poorly captured and that policies are directed towards positively influencing only the chosen indicator, thereby potentially negatively influencing other aspects of the ecosystem service that are not accounted for (Functowicz *et al*., 2001).

One means of effectively capturing an adequate description of an ecosystem service is the use of multiple indicators from both environmental and socio-economic perspectives. Indeed, by gathering both environmental and socio-economic indicators the objective ecosystem dimension and subjective (human) value dimension of an ecosystem service can be considered together. This is significant as a single monetary metric usually sought after by neoclassical economists fails to reveal information about the actual status of ecosystems. Moreover, when conducting an evaluation by environmental indicators alone socio-economic realities are inherently ignored (Straton, 2006; Winkler, 2006).

Despite the noted difficulty of establishing a comprehensive set of indicators to represent each ecosystem service, recently Hearnshaw *et al*. (2010) undertook a kind of impact analysis using various available indicators to determine the changes of a water storage project on many of the ecosystem services provided by a Canterbury river system. Accordingly, these indicators that represent the various ecosystem services can, in part, be utilized in a way to allow the construction of an ecosystem services index, as a utility index of the aggregated set of ecosystem services provided (Boyd & Banzhaf, 2007). Nevertheless, even with a list of indicators that go some way to representing the various ecosystem services provided, there remain other problematic issues to consider. For example, there is a need to ensure that a sufficiently long and uninterrupted series of data is available for each indicator and that the sampling methods used to collect the data are scientifically defensible (Ehrlich, 1996; Carpenter *et al*., 2006). The lack of sufficient and scientifically defensible data in a number of indicators has been noted by a number of researchers (Layke, 2009; Hearnshaw *et al*., 2010).

A (comprehensive) set of indicators that represent the ecosystem services provided by the river system investigated allows for the potential to construct an ecosystem services index. The construction of an ecosystem services index from the set of indicators can be undertaken in three steps. First, there is a need to normalize the quantitative output of each indicator on a 1-to-100 scale. For many indicators 1 and 100 would represent the historical minimum point and the preferred historical maximum point, respectively. However, for some indicators the historical maximum point would not be the preferred, and thus each indicator has to be normalized appropriately according to the optimal conditions of quantitative output for that indicator. Furthermore, those indicators with a considerable time-series of data will provide more accurate historical points as the extremities of the data are likely to be known with greater certainty. The second step involves the establishment of the present normalized score of the indicator on the 1-to-100 scale. The final step involves the set of ecosystem services and their associated class of ecosystem service (*i.e.* cultural ecosystem services) having preferential weights given to them by experts, which can be used to estimate the societal preference for the set of ecosystem services provided. Once preferential weights for ecosystem services are quantified, an aggregated ecosystem services index can be estimated by multiplying preferential weights with the normalized scores for each indicator and then summing (or multiplying) these together (*Equation 1*). (By multiplying preferential weights and normalized scores of indicators together, some of the interrelatedness of ecosystem services can be accounted for.)

$$\text{ESI} = \sum w_n s_{in} \text{ or } \prod w_n s_{in} \tag{1}$$

Here ESI is the ecosystem services index;
w_n is the preferential weight w for ecosystem service n; and
s_{in} is the normalized score s of indicator i that represents ecosystem service n.

2.2 Cost-Effectiveness & Sustainability

Despite the detailed account of an ecosystem services index, its construction can be readily critiqued as a unitless index has no natural anchor, lacks meaning and may be perceived to provide little benefit to water resource managers and policy makers (Kuik & Gilbert, 1999). However, the ecosystem services index, as a normalized aggregation of the set of ecosystem services provided, allows for the determination of the cost-effectiveness of a water storage project and the sustainability of the river system impacted by the water storage project. The determination of the cost-effectiveness of a water storage project requires, in addition to the ecosystem services index, data on construction costs and ongoing management costs of the water storage project. This cost data coupled with the ecosystem services index allows the use of cost utility analysis, where cost-effectiveness would be indicated by a cost-utility ratio (*i.e.* ratio of costs to ecosystem services index) after the water storage project being less than prior to its construction. This cost-utility ratio can be determined according to *Equation 2*.

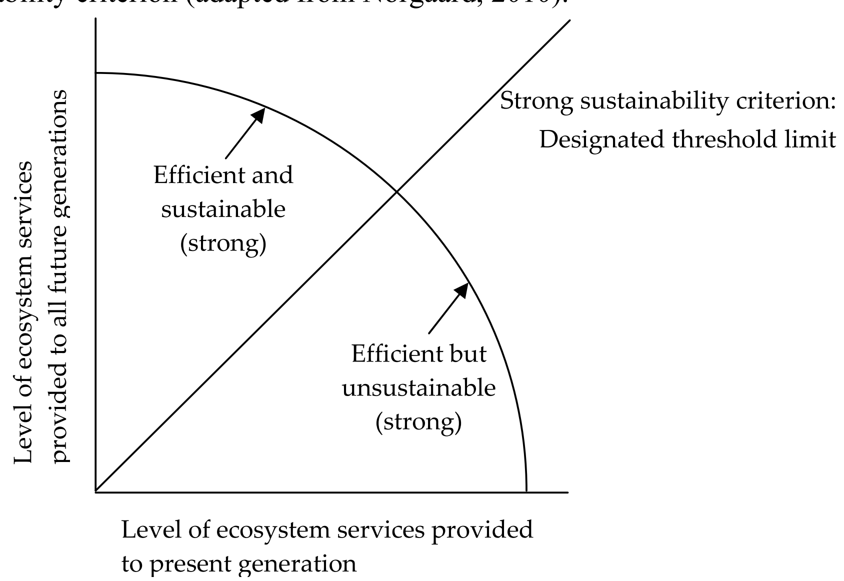
$$\text{Cost per ESI} = \sum_{0}^{T} \frac{c_j / \left(\sum w_n s_{in}\right)}{\left(1+r\right)^t} \tag{2}$$

Here c_j is the monetary cost c of management scheme j; and
r is the rate of time preference;

The ecosystem services index can measure progress towards sustainability as a result of the water storage project, where sustainability is defined as aggregated welfare (*i.e.* human well-being) that is non-declining over the long-term (Pearce *et al*., 1990; Neumayer, 2003). With welfare being non-declining it ensures that future generations are provided with at least the same (overall) welfare from ecosystem services provided as present generations. Hence, if the ecosystem services index is non-declining over the long-term then the ecosystem can be considered as either 'sustainable' or, at least, progressing towards sustainability. This determination of sustainability reflects the definition of 'weak sustainability' because it assumes that all ecosystem services can be compensated with each other and therefore commensurable and reducible to a single metric (*i.e.* ecosystem services index). For example, an ecosystem services index implies that a high scoring **Recreational Values** ecosystem service can compensate a low scoring **Water Regulation** ecosystem service.

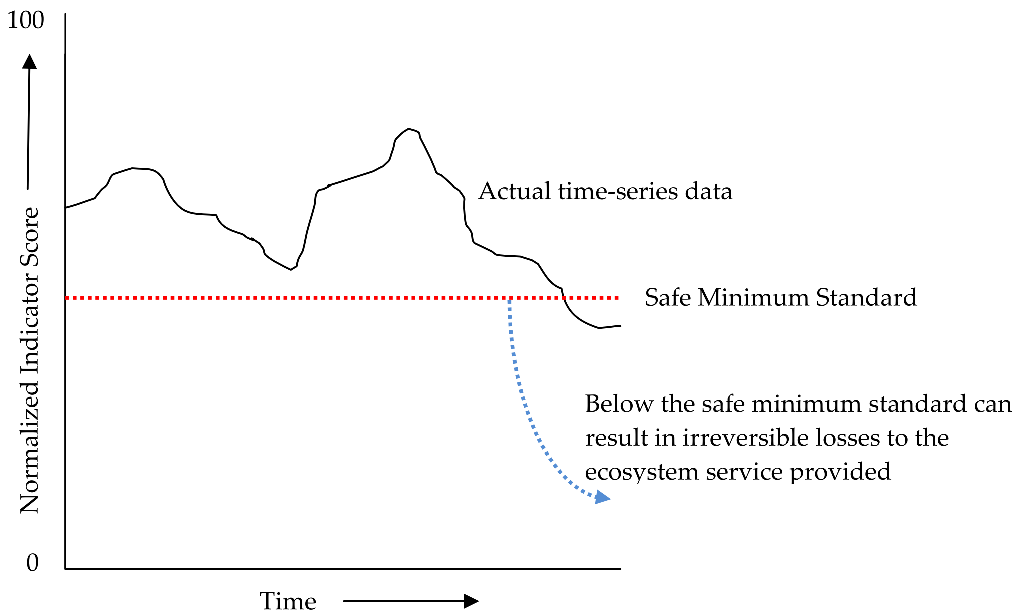
In allowing for compensation, the ecosystem services index developed as a single metric, neither is able to consider who explicitly gains and losses from the water storage project nor is able to consider definitions of 'strong sustainability' where ecosystem services would be considered non-compensatory with each other (Faucheux & O'Connor, 1998). Strong sustainability indicates that measuring the actual delivery of an ecosystem service alone does not necessarily indicate whether the ecosystem service is sustainable in the long-term (Mooney *et al*., 2005). The specific argument for strong sustainability is that there is a "... minimum quantity of ecosystem ... processes... required to maintain a well-functioning ecosystem capable of supplying [ecosystem] services" (Fisher *et al*., 2009; p. 2053). Hence, strong sustainability recognizes that ecological functioning and ecosystem services have threshold limits that need to be preserved to maintain the benefits to human well-being for both present and future generations. Indeed, if these threshold limits are passed for a certain length of time, then the possibility of the irreversible loss of ecosystem services provided by the ecosystem becomes a high likelihood (*Figure 2*) (Costanza *et al*., 2001; Norgaard, 2010).

Figure 2: Designated threshold limits are required to represent the strong sustainability criterion (adapted from Norgaard, 2010).



The difficulty with the definition of strong sustainability has been in making it operational and practicable (Prato, 2007). However, Costanza (1991) has maintained that strong sustainability can be made operational by translating designated threshold limits into the concept of the 'safe minimum standard', which was first introduced by Ciriacy-Wantrup (1952). Specifically, the safe minimum standard indicates a constraint or level of provision of an ecosystem service should pass below (or above if the safe minimum standard is a maximum), as below this designated threshold level (which may include a period of time below this level) it is believed that the ecosystem service provided will become vulnerable to irreversible losses to its sustainable supply (*Figure 3*). This concept of a safe minimum standard should be well-known to water resource managers and policy makers as it is observed in the application of acceptable minimum river flows to sustain aquatic health.

Figure 3: The concept of safe minimum standard for indicating the criterion of strong sustainability.



In applying the safe minimum standard, strong sustainability in its most complete form would be observed where safe minimum standards of all indicators have been met. It is, of course, unlikely that complete strong sustainability will be demonstrated for most ecosystems. Where this is the case, there are several methods of determining whether a water storage project has improved the strong sustainability of the river system and its delivery of ecosystem services. One method is the checklist approach where a simple count is determined for the number of safe minimum standards passed with the water storage project relative to the number passed prior to its construction. The difficulty of this method is that it assumes that all ecosystem services are equally preferred. However, given the elicitation of preferential weights for the construction of the ecosystem services index, it is likely that the set of ecosystem services provided will have a hierarchy of preferential weights. Accordingly, the preferential weights elicited for ecosystem services can be used to determine a hierarchical ranking of the set of ecosystem services provided.

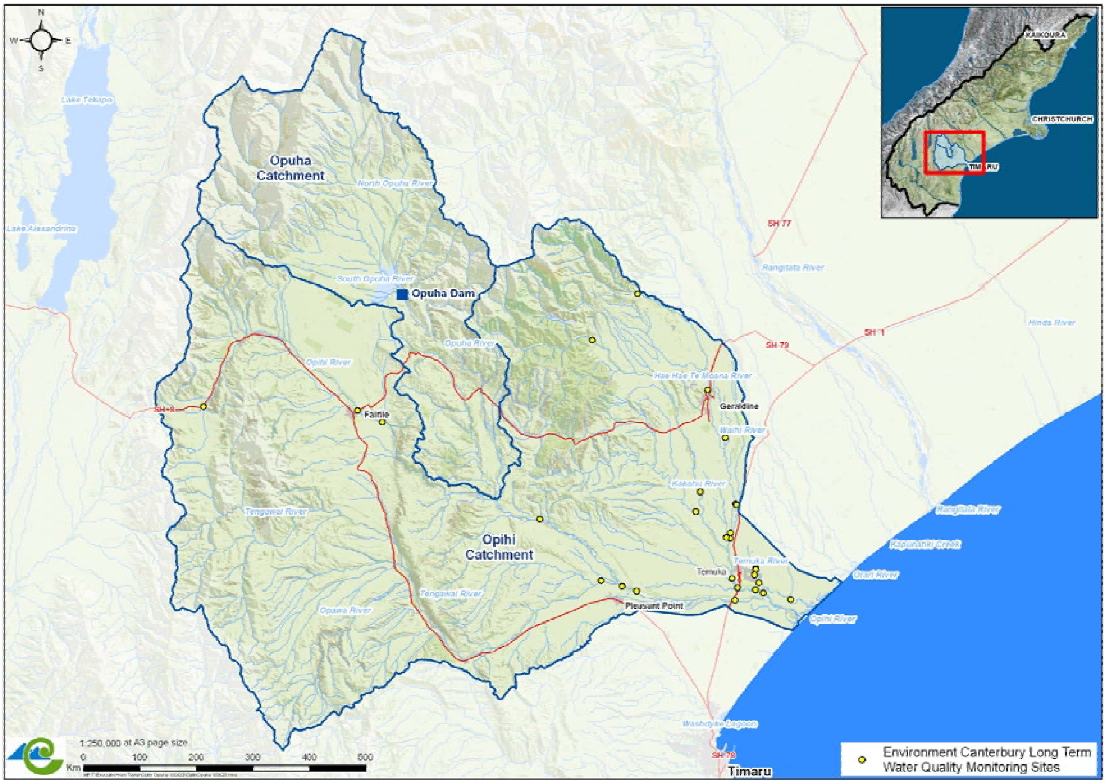
In establishing a hierarchical ranking of ecosystem services a lexicographic method termed the characteristic filtering rule can be applied to assess the (strong) sustainability of river systems impacted by water storage projects. This lexicographic method uses the hierarchical ranking of ecosystem services to act as a filter on each ecosystem service, so that each ecosystem service can be evaluated in order of preference so to determine the greater (strong) sustainability of a management scheme and resolve conflicts where compensation between ecosystem services is not appropriate. The characteristic filtering rule, specifically, considers the safe minimum standard of the highest ranked ecosystem service and establishes whether this limit has been passed or not both with and without the (water storage) project evaluated (Earl, 1986; Lockwood, 1996). Where in both cases, there is a same result (*i.e.* both pass or both fail the safe minimum standards), then the next highest ranked ecosystem service is subsequently considered. This process continues until safe minimum standards for an ecosystem service are passed by one management scheme, but not by the other. When this happens it indicates which management scheme provides the greater (strong) sustainability. Where there is no differentiation in all ecosystem services then weak sustainability alone as indicated by the ecosystem services index can be used to determine the sustainability of the water storage project evaluated.

Where a safe minimum standard fails, it is possible to determine the degree of failure by applying the concept of the (strong) sustainability gap (Ekins *et al*., 2003). Specifically, the sustainability gap is the difference between the present indicator level with the designated safe minimum standard. This difference can be used to indicate and compare the degradation of the set of ecosystem services provided by having the sustainability gaps for each ecosystem service determined from the normalized scores of indicators. An indication of unsustainability can also be indicated even where it is difficult to establish a safe minimum standard for a particular indicator. This is because instead of a safe minimum standard the trend in the indicator can be investigated, where a negative trend would provide evidence that the ecosystem service represented by the indicator may be progressing towards an unsustainable state (Ekins *et al*., 2003).

3.0 The Opihi River & the Opuha Dam

In this section the general environmental description and management history of the Opihi River and its catchment is given. In establishing the history and describing the catchment of the Opihi River it depicts broadly the spatio-temporal boundaries of the river system to be investigated in the project evaluation. The headwaters of the Opihi River are found in the foothills of the Southern Alps at elevations of up to 2200 metres (de Joux, 1982). From these headwaters the river flows through the Timaru downlands and over the Canterbury Plains (*i.e.* including the Levels Plains area) to the coast. The entire catchment of the Opihi River is made of three additional rivers or tributaries (*Figure 4*). These tributaries are the Tengawai River, the Opuha River and the Temuka River. The Opuha River and the Temuka River also have tributaries. These are the North Opuha and South Opuha Rivers on the Opuha River and the Waihi, Hae Hae Te Moana and Kakahu Rivers on the Temuka River. Despite these many tributaries the primary concern of this paper is the ecosystem services provided by Opihi River and, in particular, the part of the Opihi River from the confluence where the Opihi River and the Opuha River converge to the coast.

Figure 4: An outline of the catchment of the Opihi River and its tributaries.



The total catchment area of the Opihi River and its tributaries is approximately 245,000 hectares. Within this catchment area there are a range of land uses. These land uses include extensive grazing on the foothills, intensive dairy farming and cropping on the downlands and Levels Plains. The various rivers that make up the catchment of the Opihi River are all rain-fed. Peak river flows normally occur during the winter months. The average rainfall in the catchment is approximately 860 millimetres. However, rainfall ranges from 1400 millimetres in the foothills to 550 millimetres at the coastal river mouth. The average rainfall during the irrigation season (September to April) is approximately 700 millimetres in the foothills and only 420 millimetres at the coast (Canterbury Regional Council, 1990).

The strong winds and low annual rainfall during summer months results in the catchment being prone to drought conditions. These drought conditions severely impact on the productive use of the agricultural land. In an effort to overcome these droughts and soil water deficiencies for agricultural production, the Levels Plains Irrigation Scheme was developed in 1936. However, despite this irrigation scheme, the demand for water often exceeded its supply. This was especially the case during the numerous drought conditions experienced during the 1980s (Canterbury Regional Council, 1990). The result of these dry summer months coupled with the excessive abstraction of water from the Levels Plain Irrigation Scheme lead to the Opihi River often having very low river flows. In fact, sometimes the Opihi River dried up completely (Scarf, 1984; Worrall, 2007). Consequently, various ecosystem services have become lost or degraded.

Apart from the degradation of the ecosystem service **Freshwater Supply** used for irrigation, the adverse consequences of unacceptable minimum river flows in the Opihi River resulted in a decline in water quality evident through increasing water temperatures, decreasing dissolved oxygen levels and a reduction in the capacity of the river to assimilate pollutants (Canterbury Regional Council, 1990). The poor water quality in the Opihi River in turn resulted in the degradation of various ecosystem services that are associated with the loss of habitat for fish and other aquatic life (*e.g*. the ecosystem services **Food**, **Recreational Values**). Furthermore, the unacceptable minimum river flows were unable to keep the river mouth to the sea open for extensive periods of time. While the closure of the river mouth on the Opihi River is a natural feature, low flows resultant from water abstraction increase the likelihood of this occurring. This inability of the river mouth to open, exacerbated problems of poor water quality in the neighbouring Opihi Lagoon and prevented game and native fish migrating out to sea. The limited fish passage to the sea and poor water quality of the Opihi River and its lagoon were key factors in the declining population of fish and availability of mahika kai (Dacker, 1990; Scarf, 2009; pers. comm.). Mahika kai was once abundant in the Opihi River prior to intensive agricultural operations in the catchment (Waaka-Home, 2010; pers. comm.).

With the noticeable degradation of some ecosystem services, the idea of constructing a dam for the Opihi River in an effort to store water and augment minimum river flows was reconsidered during the early 1990s. It was maintained that a dam would through water storage provide a reliable and increased supply of freshwater for the purposes of irrigation downstream of the dam. In addition, the augmented minimum river flows were foreseen to allow improvements to some ecosystem services including improving the degraded recreational fishery that was once of national importance. The *ex-ante* impact assessment of a dam, to be located on the Opuha River, provided strong indications that the proposed Opuha Dam scheme would generate many economic and environmental benefits. A number of negative impacts were also anticipated. These included the limited number of flushing flows downstream of the dam resulting potentially in the increased likelihood of algal growth and the loss of natural character on the Opihi River (Canterbury Regional Council, 1995). However, despite these negative impacts it was believed that the benefits would outweigh the few environmental costs (Worrall, 2007). Consequently, the commission tasked with reviewing the *ex-ante* evaluation of the proposed Opuha Dam scheme gave its consent. Dam construction went ahead in 1996, and despite a devastating and unexpected dam breach during construction after a period of heavy rain in 1997 the dam was fully operational by the end of 1998 (Worrall, 2007).

4.0 Ecosystem Services Analysis

A question remains over a decade after the construction of the Opuha Dam scheme, as to whether these claims of economic and environmental benefits from the Opihi River are indeed accurate? In order to determine the validity of the performance claims of the Opuha Dam as a cost-effective and sustainable investment, an (*ex-post*) evaluation of the cost-effectiveness of the dam scheme and the sustainability of the Opihi River impacted by the dam are undertaken herein. The initial step in undertaking such analysis using an ecosystem services approach is the determination of the set of ecosystem services provided by the Opihi River (World Resources Institute, 2008). The determination of the set of ecosystem services provided requires systematically considering whether each ecosystem service from *Table 1* is (or has been previously provided) by the Opihi River.

The result of this task is that all ecosystem services except **Biological Products** and **Climate Regulation** are provided by the Opihi River. In the case of **Climate Regulation** it is recognized that only ecosystems that have a large carbon sink (*e.g.* forests) are likely to provide a delivery of this service. With regards to **Biological Products** it was established that this service is not yet provided. However, this ecosystem service could have some quasi-option value, in that future technological progress may attribute value to biological products derived from species that inhabit the Opihi River. *Table 2* indicates the set of ecosystem services provided by the Opihi River and examples of the benefits to human-well-being derived from this set.

Table 2: The set of ecosystem services provided by the Opihi River.

Class	Ecosystem service	Examples of ecosystem service
Provisioning ecosystem services	**Food**	Fisheries (*e.g.* salmon, trout)
		Mahika kai (*e.g.* eel, whitebait, flounder)
	Fibre	Flax, driftwood
	Freshwater supply	Irrigation
		Hydroelectric production
		Municipal water supply
		Industrial water supply
		Stock water supply
	Biological products	*Not applicable*
	Abiotic products	Gravel extraction for road chip and concrete
Regulating ecosystem services	**Climate regulation**	*Not applicable*
	Disease regulation	Parasite and toxic algae regulation
	Water regulation	River flow regulation (*e.g.* minimum river flows)
	Water purification	Removal of pollutants
	Erosion control	Stabilization of river banks
	Pest regulation	Invasive non-native species (*e.g.* algae, willows, gorse)
	Natural hazard regulation	Flood and drought protection
Cultural ecosystem services	**Conservation values**	Native biodiversity and habitat
		Endangered native species (*e.g.* black-billed gull)
		Significant ecological landscapes (*e.g.* Opihi Lagoon)
	Educational values	Historical/archaeological values & knowledge systems
	Aesthetic values	Perceived beauty
	Spiritual values	Māori values (*e.g.* natural character, mauri)
	Recreational values	Boating (*e.g.* sailing, rowing, kayaking)
		Fishing
		Hunting (*e.g.* duck hunting)
		Picnicking (*e.g.* Opihi Lagoon)
		Swimming
		Walking

4.1 Indicator Evaluation

With the set of ecosystem services provided recognized, available environmental and socio-economic indicators were associated with each ecosystem service in order to represent the state of the ecosystem service. Where possible, ecosystem services were represented by multiple indicators from environmental and socio-economic perspectives in order to capture the full extent of each ecosystem service. However, in undertaking this exercise it was recognized that indicators that represent ecosystem services are underdeveloped. Many ecosystem services do not have sufficient numbers of environmental and socio-economic indicators available that allow the ecosystem service to be fully captured. In particular, regulating and cultural ecosystem services appear to have a less comprehensive representation of indicators available than provisioning ecosystem services. This conclusion was also surmised in a recent study undertaken by Layke (2009). Hence, a critical research requirement for the ecosystem services approach is the development of scientifically sound indicators for many ecosystem services, especially those found in the regulating and cultural ecosystem service classes. *Table 3a* (provisioning ecosystem services) and *Table 3b* (regulating and cultural ecosystem services) indicate the various indicators considered for representing the set of ecosystem services provided by the Opihi River.

Table 3a: Indicators considered to represent provisioning ecosystem services provided by the Opihi River.

Class	Ecosystem service	Indicators	Indicator type
Provisioning ecosystem services	**Food**	Number of Days River Mouth Closed	Environmental
		Spawning Numbers	Environmental
		Water temperature	Environmental
		Dissolved oxygen levels	Environmental
		Annual periphyton cover	Environmental
		Commercial fishery employment	Socio-economic
		Fish taste	Socio-economic
		Sedimentation levels	Environmental
		Number of mahika kai species available	Environmental
		Cultural health index	Socio-economic
		Number of salmon caught	Environmental
	Fibre	Number of fibrous species available	Environmental
	Freshwater supply	Irrigated area	Environmental
		Agricultural production	Environmental
		Nitrogen fertilizer application	Environmental
		Economic impact over irrigated area	Socio-economic
		Full time employment	Socio-economic
		Hydroelectric hours produced	Socio-economic
		Total rate of water abstraction	Environmental
		E. coli levels/faecal coliform levels	Socio-economic
		Cost of water treatment	Socio-economic
		Cryptosporidium levels	Environmental
	Abiotic products	Volume of gravel extracted per year	Environmental
		Profitability of gravel resource	Socio-economic

Table 3b: Indicators considered to represent regulating and cultural ecosystem services provided by the Opihi River.

Class	Ecosystem service	Indicators	Indicator type
Regulating ecosystem services	**Disease regulation**	Annual periphyton cover	Environmental
		Number of fish kills	Environmental
	Water regulation	Minimum river flows	Environmental
		Number of days river mouth closed	Environmental
		Number of flushing flows	Environmental
		Number of flood flows	Environmental
		Instantaneous annual flood peaks	Environmental
	Water purification	Total nitrogen concentration	Environmental
		Total phosphorus concentration	Environmental
		Nitrate concentration	Environmental
		Dissolved oxygen phosphorus concentration	Environmental
		pH levels	Environmental
		Annual periphyton cover	Environmental
		Macroinvertebrate community index (MCI)	Environmental
		EPT%	Environmental
	Erosion control	Cost of willow planting	Socio-economic
		Sedimentation levels	Environmental
		Total suspended solids	Environmental
		Turbidity	Environmental
	Pest regulation	Area covered by non-native vegetation	Environmental
	Natural hazard regulation	Number of flood flows	Environmental
		Total economic cost of flood event	Socio-economic
		Number of fatalities of flood event	Socio-economic
		Irrigated area	Environmental
		Total economic cost of drought event	Socio-economic
Cultural ecosystem services	**Conservation values**	Native biodiversity	Environmental
		Macroinvertebrate community index (MCI)	Environmental
		Number of endangered native bird species	Environmental
		Number of important ecological landscapes	Environmental
	Educational values	Number of studies on Opihi River ecology	Socio-economic
		Number of publications about Opuha Dam	Socio-economic
	Aesthetic values	Annual periphyton cover	Environmental
		Clarity	Environmental
		Willingness to pay for property	Socio-economic
	Spiritual values	Cultural health index	Socio-economic
		Native biodiversity	Environmental
		Macroinvertebrate community index (MCI)	Environmental
	Recreational values	Annual periphyton cover	Environmental
		Number of swimmers in river	Socio-economic
		E. coli levels	Environmental
		Clarity	Environmental
		Total angler days per season	Socio-economic
		Number of salmon caught	Environmental

An evident problem for the construction of an ecosystem services index observed in *Table 3a* and *Table 3b* is that many indicators represent multiple ecosystem services (Boyd & Banzhaf, 2007). For example, the indicator *Macroinvertebrate Community Index* can capture components of various ecosystem services including **Water**

Purification, **Conservation Values** and **Spiritual Values**. This problem occurs because of the interrelatedness of ecosystem services (De Groot *et al*., 2002; Capistrano *et al*., 2006; Rodriguez *et al*., 2006). This is because the use of an indicator for representing a multitude of ecosystem services will result in double counting. Despite double counting being a critical problem for ecosystem services index construction, it is rarely considered. In fact, only one of 34 studies on ecosystem services has discussed the problem of double counting explicitly (Fisher *et al*., 2009).

In an effort to resolve the problem of double counting there is a need to standardize the various indicators to represent only a single ecosystem service. To undertake this standardization an evaluation of the indicators was performed. Recently, Layke (2009) suggested various criteria to evaluate indicators based on work developed by Boswell (1999). These criteria were 'the availability of data for the indicator' and 'the ability the indicator has in communicating information about the ecosystem service'. Specifically, the criterion of data availability considers the degree of data available at appropriate spatial and temporal scales for the indicator considered. The criterion of the ability to communicate information considers the degree to which the indicator can communicate information about the ecosystem service in a way that is intuitive and limits ambiguity. In addition to these two criteria, an additional cost consideration is also maintained in the evaluation of indicators.

Preliminary results for this indicator evaluation are shown in *Table 4* where these criteria were scored on a one-to-five scale by two expert scientists. The average scores for the two criteria were summed and divided by the average cost score given providing a measure of the cost-effectiveness of the indicator for each ecosystem service where an indicator represented multiple ecosystem services. Accordingly, where one indicator is initially considered capable of representing two or more ecosystem services, the ecosystem service that provided the highest cost-effectiveness for that indicator was preferred to represent that ecosystem service. However, in some cases, it was more appropriate to assign an indicator not to that ecosystem service with the highest cost-effectiveness where that ecosystem service was well-represented with other indicators.

Table 4: Evaluation of indicators that represent multiple ecosystem services. Boxes highlighted indicate preferred ecosystem services to be represented by the indicator.

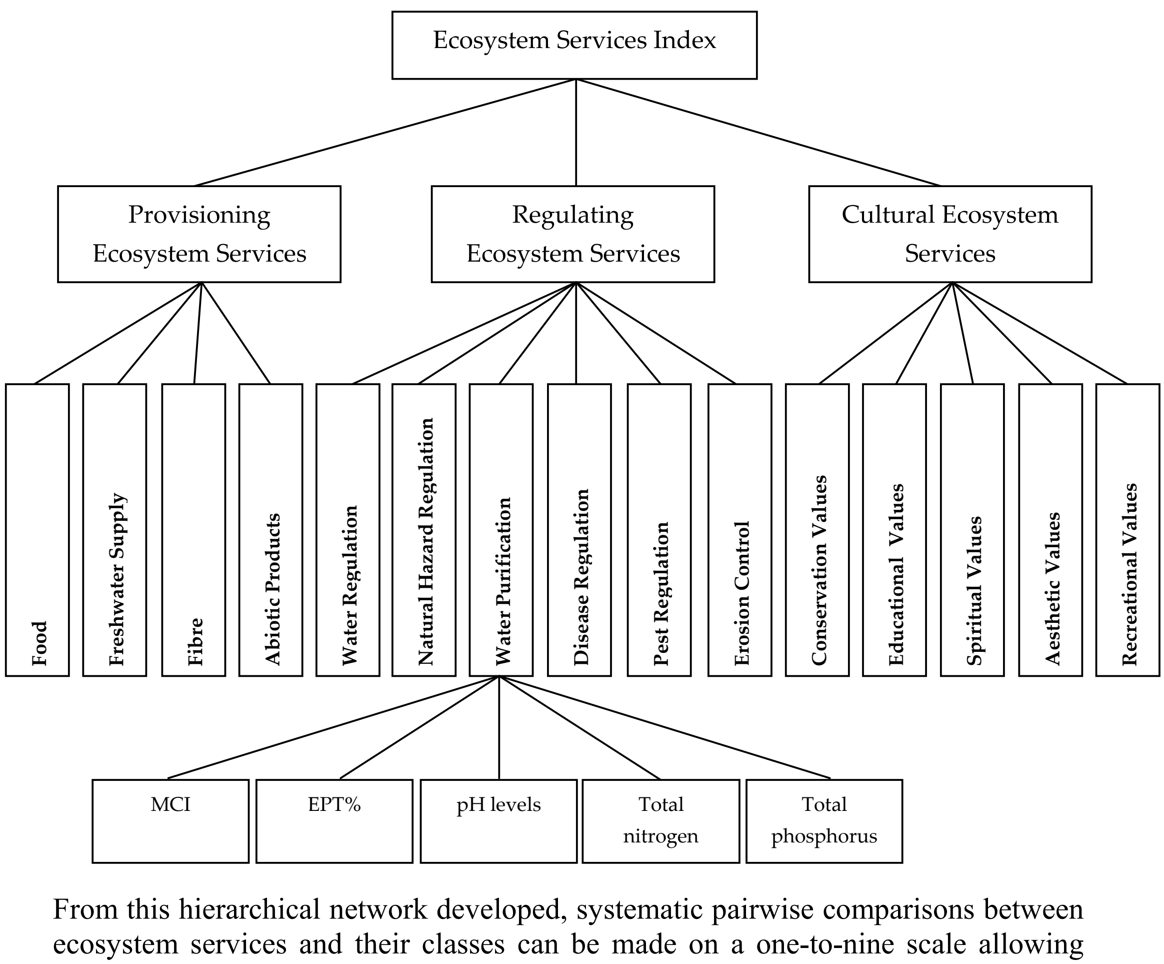
Indicator	Ecosystem service	Ability to communicate information (1-5 scale)	Data availability (1-5 scale)	Annual cost (1-5 scale)	Indicator cost-effectiveness (0-10 scale)
Annual Periphyton Cover	**Food**	2.5	3.5	2	3
	Water Purification	2.5			3
	Disease Regulation	3			3.25
	Aesthetic Values	4.5			4
	Recreational Values	5.5			4.5
Clarity	**Aesthetic Values**	4	3.5	1.5	5
	Recreational Values	4.5			5.33
Cultural Health Index	**Food**	4	2	3.5	1.71
	Spiritual Values	4.5			1.86
E. coli Levels	**Freshwater Supply**	5	5	3	3.33
	Recreational Values	3.5			2.83
Irrigated Area	**Freshwater Supply**	4.5	5	1.5	6.33
	Natural Hazard Regulation	2			4.67
	Water Purification	3			5.33
Macroinvertebrate Community Index	**Water Purification**	4	4	3.5	2.29
	Spiritual Values	2.5			1.86
Native Biodiversity	**Conservation Values**	5	3	4	2
	Spiritual Values	3			1.5
Number of Days River Mouth Closed	**Food**	4.5	3.5	2	4
	Water Regulation	4			3.75
Number of Flood Flows	**Water Regulation**	4	5	3	3
	Natural Hazard Regulation	5			3.33
Number of Salmon Caught	**Food**	5	4	3	3
	Recreational Values	5			3
Total Suspended Sediment	**Erosion Control**	3	4	3.5	2
	Aesthetic Values	3			2
Turbidity	**Water Purification**	3.5	3.5	3	2.33
	Erosion Control	3			2.17

4.2 Preferences of Ecosystem Services

In order to construct an ecosystem services index from the indicators available that represent the set of ecosystem services provided for the Opihi River there is a need to find an appropriate multi-criteria analytical method that can determine preferential weights of ecosystem services from cardinal measurements. One method that can determine preferential weights and allow the construction of utility indices is the analytical hierarchy process (Saaty, 1980; 1995). Recently, Zhang and Liu (2009) have employed the analytical hierarchy process for the determination of preferential weights for a limited number of ecosystem services provided by the Ruoergi Plateau marshland ecosystem in China. Hearnshaw (2009) has also successfully applied this analytical method to Te Waihora/Lake Ellesmere system for the determination of preferential weights for all ecosystem services provided by the lake.

The analytical hierarchy process, specifically, is a multi-criteria analytical method of expert elicitation, which attempts to decompose evaluations of preference into a hierarchical network (Saaty, 1980). In decomposing the evaluation of preferences for ecosystem services, water resource managers and policy makers tasked with the determination of preferential weights can do so without excessive information overload. In this work a three-level hierarchical network is developed. At its pinnacle is the ecosystem services index. The next level contains the classes of ecosystem services (*e.g.* provisioning ecosystem services). The bottom level contains the set of ecosystem services provided, which when aggregated with indicators that represent that ecosystem service allow for the construction of an ecosystem services index (*Figure 5*).

Figure 5: The hierarchical network for constructing preferences of ecosystem services.



From this hierarchical network developed, systematic pairwise comparisons between ecosystem services and their classes can be made on a one-to-nine scale allowing preferential weights to be mapped and estimated (*Table 5*). The pairwise comparisons that are elicited represent the cardinal intensity of preference between each ecosystem service pairing.

Table 5: Qualitative-numeric elicitation of preferential weights (adapted from Saaty, 1980).

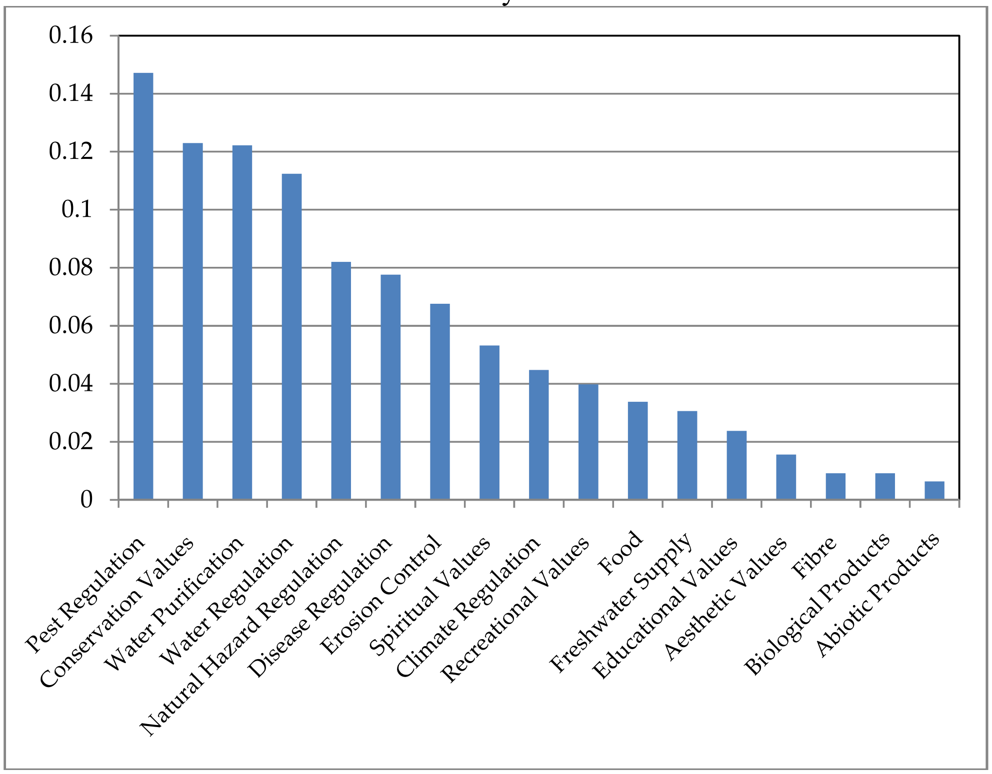
Linguistic Preference	Numeric preferences
Indifference	1
Weak preference	3
Medium preference	5
Strong preference	7
Overwhelming preference	9

These pairwise comparisons *w* as ratios between each ecosystem service pairing can be expressed in a ratio matrix *A* (*Equation 3*). It is in a ratio matrix that analysis can be undertaken and preferential weights estimated for the set of ecosystem services. While this ratio matrix for the many ecosystem services would be computationally demanding to solve, there are many programmes (*e.g. Expert Choice*) dedicated to undertaking evaluations by the analytical hierarchy process.

$$A = \begin{array}{c} 1 \\ 2 \\ \vdots \\ n \end{array} \begin{pmatrix} w_1/w_1 & w_1/w_2 & \cdots & w_1/w_n \\ w_2/w_1 & w_2/w_2 & \cdots & w_2/w_n \\ \vdots & \vdots & & \vdots \\ w_n/w_1 & w_n/w_2 & \cdots & w_n/w_n \end{pmatrix} \tag{3}$$

At this stage preferences were collected from six water resource managers and policy makers who represent various important stakeholder groups within the Canterbury region. This number of representatives is insufficient for a complete evaluation, as preferences need to be obtained from more representatives including those local people who are intimately connected with the Opihi River and its catchment. All people that provided preferences of ecosystem services were requested to elicit preferences from the perspective of the present needs of the stakeholders that they represent. Despite that preferences obtained are preliminary in nature, these preferences of ecosystem services obtained were analysed by the computational programme *Expert Choice* to further demonstrate the method advocated. From this analysis, preliminary results from the six water resource managers and policy makers indicate the importance of the ecosystem services **Pest Regulation**, **Conservation Values**, **Water Purification** and **Water Regulation** (*Figure 6*). Surprisingly, **Freshwater Supply** was not a highly preferred ecosystem service, despite the perceived lack of this ecosystem service being a critical driver for the construction of water storage projects in the Canterbury region (*e.g.* Opuha Dam scheme). Moreover, from these preliminary results it is evident that regulatory ecosystem services are more highly preferred than either provisioning or cultural ecosystem services.

Figure 6: Preferential weights for ecosystem services provided by Canterbury Rivers.



Despite the interesting preferential weights elicited, it is reiterated that these results are preliminary and only capture the preferential weights of six water resource managers and policy makers. Moreover, from the analysis undertaken it is evident that there are some inconsistencies (or intransitivities) in the preferences elicited, as indicated by a consistency index that is calculated by the *Expert Choice* computational programme. In fact, the average inconsistency, as calculated by a consistency index, was approximately 19 per cent. This is greater than the recommended level of inconsistency of ten per cent. Saaty (1980; 1995) suggests that where the consistency index for preferences is greater than ten per cent then preferential weights should be considered for some revision, where the intransitivity of ecosystem service pairings is particularly high. This revision process has not yet been undertaken in the preliminary results shown, further indicating caution in attempting to observe any meaningful extrapolation of the results thus far obtained.

4.3 Opihi River Sustainability

In order to evaluate the sustainability of the Opihi River impacted by the Opuha Dam scheme, two definitions of sustainability (*i.e.* weak and strong sustainability) should be evaluated. At this stage, there are insufficient results available to construct an ecosystem services index. Hence, the determination as to the weak sustainability of the Opihi River impacted by the Opuha Dam scheme is not investigated further, until such time that further data is acquired allowing for the construction of an ecosystem services index. However, with regards to strong sustainability, some preliminary results have been estimated allowing for the demonstration of how strong sustainability can be made operational. Specifically, for the purposes of demonstrating the analysis of strong sustainability three ecosystem services are investigated. These three ecosystem services, which each derive from different classes of ecosystem services, are the ecosystem services **Food**, **Water Purification** and **Recreational Values**. For the purposes of the analysis of strong sustainability of the Opihi River, *Table 6* indicates the average safe minimum standards elicited by two expert scientists for each indicator that represents the three ecosystem services investigated.

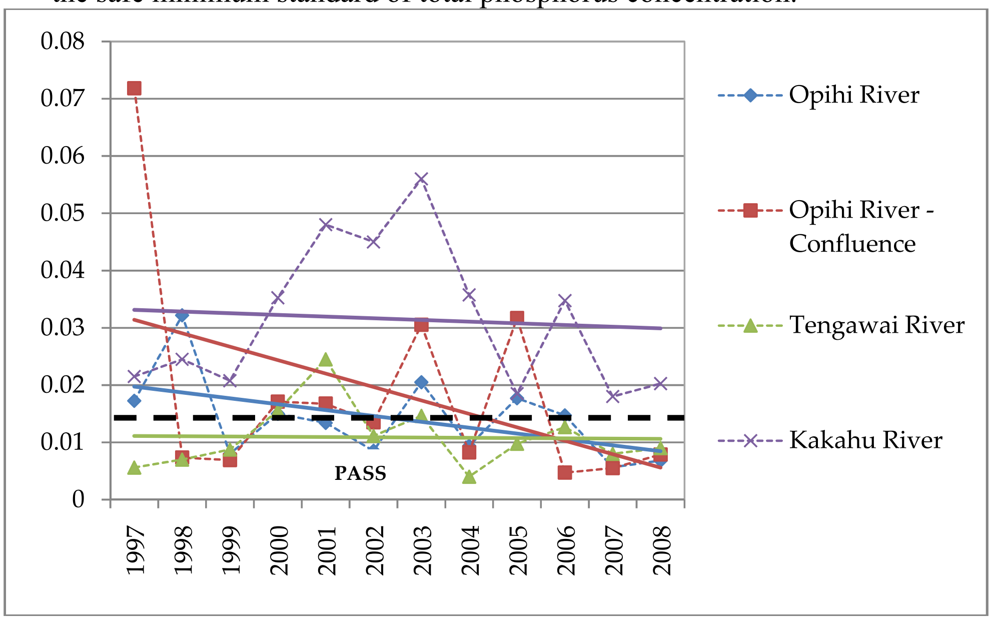
Table 6: Safe minimum standards for the indicators that represent the three ecosystem services investigated for the Opihi River and its tributaries.

Ecosystem service	Indicator	Unit	Safe minimum standard	
			Threshold	Length of time
Water purification	Macroinvertebrate community index (MCI)	Index	Minimum 100	Yearly
	EPT%	Percentage	Minimum 50%	Yearly
	pH levels	pH scale	Minimum 7 & maximum 8	---
	Total phosphorus concentration	mg per l	Maximum 0.015 per l	Yearly
	Total nitrogen concentration	mg per l	Maximum 0.4 per l	Yearly
Recreational Values	*E. coli* levels	Percentage	Minimum 85% of samples above 550 *E. coli* units per 100ml	---
	Total angler days per season	Days per season	No negative trend	---
	Number of salmon caught	Count per year	No negative trend	---
Food	Water temperature	Celsius	Minimum 4C & Maximum 20C	Daily
	Dissolved oxygen levels	ml per l	Minimum 8ml/l	Daily
	Number of days river mouth closed	Days per year	Maximum 5 days	---
	Spawning numbers	Count per year	No negative trend	---

The application and analysis of strong sustainability in its most complete form would be observed by all indicators representing the set of ecosystem services provided having been met. Previously, this was indicated to be unlikely. As a result, the appropriate analytical method to investigate strong sustainability, as argued previously, is the characteristic filtering rule. In employing the characteristic filtering rule the three ecosystem services indicated above to analyse strong sustainability, are ranked according to their preferential weights established previously (*Figure 6*). Hence, the ecosystem service **Water Purification** is ranked first, then the ecosystem service **Recreational Values** and finally the ecosystem service **Food**. This ranking indicates that if all indicators for the ecosystem service **Water Purification** are passed for the Opihi River either after or before the Opuha Dam scheme, then strong sustainability can be implied. Accordingly, we now analyse the various indicators that represent the ecosystem service **Water Purification**.

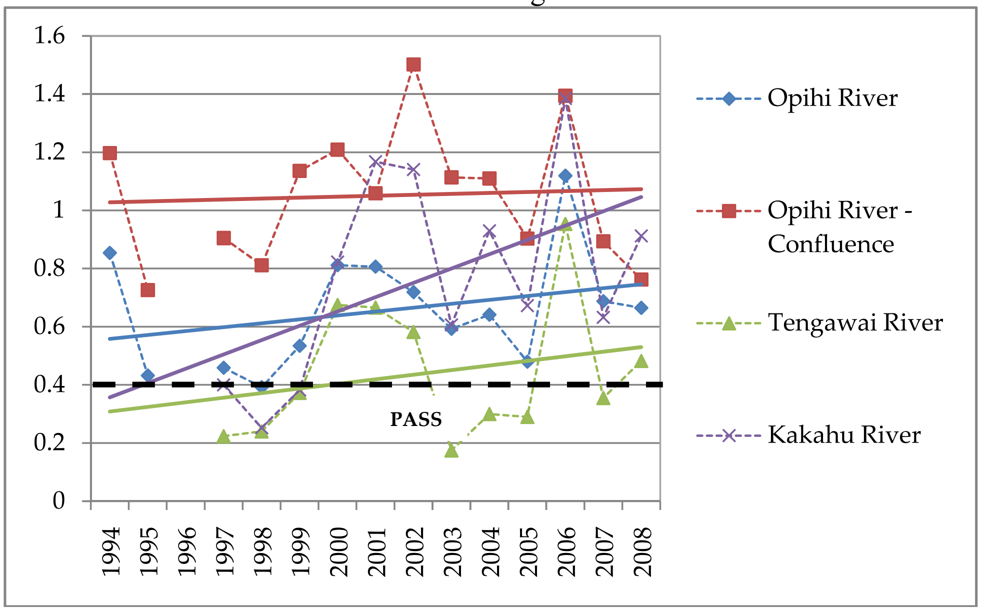
In *Figure 7* the indicator total phosphorus concentration is depicted. It is indicated that since the construction of the Opuha Dam scheme the safe minimum standard (*i.e.* maximum of 0.015 milligrams per litre) for the Opihi River is increasingly being passed (though fails standard in 2003 and 2005) as opposed to prior its construction where it had never passed the safe minimum standard. However, it is noted that only a small amount of data is available prior to the complete construction of the Opuha Dam scheme.

Figure 7: Trends and actual data for the average annual total phosphorus concentration in the Opihi River and its tributaries between 1997 and 2008 (adapted from Environment Canterbury, 2009). *Here* ▬ ▬ ▬ is the safe minimum standard of total phosphorus concentration.



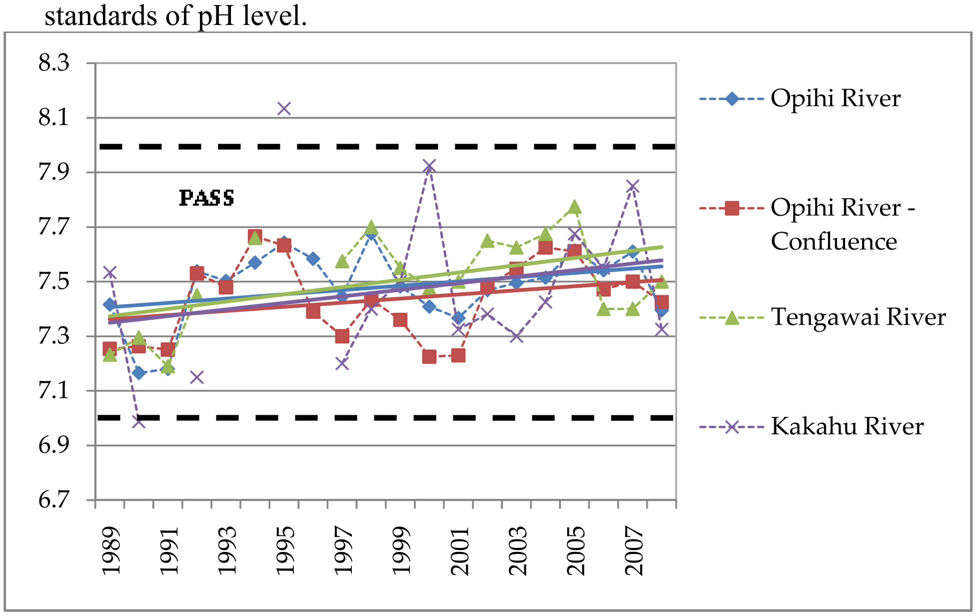
In *Figure 8* the indicator total nitrogen concentration is depicted. It is indicated that the safe minimum standard (*i.e.* maximum of 0.4 milligrams per litre) is failed for the Opihi River both before and after the construction of the Opuha Dam scheme. Worse still it appears that the total nitrogen concentration is increasing. This increase in total nitrogen concentration is presumably from land use intensification by way of increased fertilizer application and land use change since the construction of the Opuha Dam scheme.

Figure 8: Trends and actual data for the average annual total nitrogen concentration in the Opihi River and its tributaries between 1994 and 2008 (adapted from Environment Canterbury, 2009). *Here* — — — is the safe minimum standard of total nitrogen concentration.



In *Figure 9* the indicator pH levels is depicted. It is indicated that the safe minimum standards (*i.e.* minimum pH of 7 and maximum pH of 8) are passed for the Opihi River both before and after the construction of the Opuha Dam scheme.

Figure 9: Trends and actual data for the average annual pH levels in the Opihi River and its tributaries between 1989 and 2008 (adapted from Environment Canterbury, 2009). *Here* — — — is the safe minimum standards of pH level.



Finally, in *Table 7* the indicators macroinvertebrate community index (MCI) and EPT% are depicted. It is indicated with regards to the EPT% indicator that the safe minimum standard (*i.e.* minimum EPT of 50 per cent) is passed for the Opihi River after the construction of the Opuha Dam. However, with regards to the macroinvertebrate index the safe minimum standard (*i.e.* minimum MCI of 100) is failed for the Opihi River after the construction of the Opuha Dam.

Table 7: Trends and actual data in MCI and EPT% for the Opihi River and its tributaries in 2007 (adapted from the Ministry for the Environment, 2009).

River system / Environmental indicator	Opihi River: Waipopo site	Opihi River – Confluence: Rockwood site	Opuha River: Skipton Bridge site
Macroinvertebrate Community Index (MCI)	97	109	98
EPT%	73	83	10

From the analysis of the various indicators that represent the ecosystem service **Water Purification**, it is evident that not all indicators pass or fail for the Opihi River either before or after the construction of the Opuha Dam scheme. Accordingly, in order to evaluate the strong sustainability of the Opihi River the various indicators for the next ranked ecosystem service (*i.e.* **Recreational Values**) are analysed. This task is not performed in this paper, though will be undertaken for future research to fully determine the strong sustainability of the Opihi River impacted by the Opuha Dam scheme.

4.4 Opuha Dam Cost-Effectiveness

In order to evaluate the cost-effectiveness of the Opuha Dam, cost analysis was initially considered to determine the construction costs of the dam. From this simple economic analysis it was established that like other water storage projects the dam was underestimated in construction costs. In 1994 prior to the Opuha Dam construction it was estimated that it would cost NZ$28 million to build. However, after completion the resultant costs were NZ$34 million or approximately 21 per cent over initial cost estimates. The underestimation in costs was for three reasons: one, the belief that the hydroelectric production should be increased; two, the need to construct a weir immediately downstream of the dam to control river flows; and three, the impact of a dam breach resulting in extensive damage to the partially built dam (Worrall, 2007). With the construction of an ecosystem services index, cost utility analysis can be undertaken from the cost data available to indicate the cost-effectiveness of the Opuha Dam scheme relative to that delivered prior to its construction.

5.0 Discussion

From the few preliminary results obtained in this paper, it is certainly premature to indicate the cost-effectiveness of the Opuha Dam scheme and the sustainability of the Opihi River impacted by the dam. Accordingly, further data collection is required to fully determine the issues of cost-effectiveness and sustainability for the Opihi River case. Nevertheless, regardless that only a few preliminary results have been obtained, an ecosystem services approach using various analytical methods (*e.g.* cost utility analysis, analytical hierarchy process) has been demonstrated, which allows for the critical issues of cost-effectiveness and sustainability to be evaluated in a transparent and systematic way.

Despite the development in this paper of an ecosystem services approach to evaluate water storage projects, it is recognized that there is an increasing demand for water resources for the purposes of irrigation in the Canterbury region. In order to meet this increasing freshwater demand and resolve Canterbury's water allocation problem there are calls to evaluate proposed water storage projects to be constructed on various Canterbury river systems. For example, with regards to the Opihi River and its catchment, the demand for freshwater has led to the proposal of two 'feasible' management schemes (Canterbury Strategic Water Study, 2006). Both management schemes are projected to provide at least the same amount of potential irrigated area as the present Opuha Dam scheme (*i.e.* 16,000 hectares). One proposed management scheme, the Opihi Dam, is to construct another dam upstream from the Opihi Gorge. The other management scheme is to channel and transfer water from Lake Tekapo, found in neighbouring Waitaki catchment, through the Opuha and Opihi Rivers (*Table 8*).

Table 8: Proposed management schemes for increasing the supply of freshwater available for abstraction from the Opihi River (adapted from Canterbury Strategic Water Study, 2006).

Management scheme	Estimated cost (NZ$)	Irrigated area (ha)	Reliability	Active storage (Mm^3)
Opuha Dam (present)	---	16,000	28 (92%)	83
Opuha Dam and Opihi Dam	$33 million for Opihi Dam and $57 million for water distribution system	33,000	22 (93%)	240
Water from Lake Tekapo (10 m^3/s) with Opuha Dam	?	33,000	15 (96%)	83

In order to evaluate proposed water storage projects there is a need to undertake an *ex-ante* evaluation in a systematic way, presumably using the ecosystem services approach and analytical methods developed in this paper. The difficulty of *ex-ante* evaluations (as opposed to *ex-post* evaluations) is the 'uncertainty' in evaluating the possible future outcomes of water storage projects. Indeed, there are limits as to the ability water resource managers and policy makers can foresee changes over time that may redistribute and transform impacts on ecosystem services provided by river systems. Despite these uncertainties, it is evident that *ex-ante* evaluations, even if systematically undertaken, ignore these uncertainties. This leaves water storage projects being evaluated *ex-ante* as if all environmental and socio-economic conditions will remain under changed. Yet, it is most likely that over the life time of a water storage project that environmental and socio-economic drivers that determine and transform river systems will change significantly. These changes to drivers that transform river systems include: changes in land use, changes in water resource use (and non-use) priorities, changes in species composition, climatic changes, demographic changes, changes in local, regional and global markets, changes in institutions, governance and policy, technological changes and changes in consumptive behaviour and preferences (World Commission on Dams, 2000; Capistrano *et al*., 2006).

Given the limited ability to predict future outcomes of ecosystem services provided from river systems, it is inappropriate to employ forecasting methods for *ex-ante* evaluations of proposed water storage projects. Instead, the most appropriate analytical method for evaluating ecosystems, as complex systems, is scenario analysis (Petersen *et al*., 2003; World Resources Institute, 2008). Specifically, scenario analysis allows for a set of plausible future outcomes to be constructed in a systematic way that considers what might happen to a river system under various environmental and socio-economic conditions for a proposed water storage project. Accordingly, unlike forecasting, which assumes existing environmental and socio-economic conditions will prevail in the future, with scenario analysis different conditions resultant from changes to drivers can be explored at a range of spatio-temporal scales (Van der Heijden, 1996).

In order to construct a set of scenarios in a systematic way there is need to elicit and rank all relevant environmental and socio-economic drivers, consider their changes over time and associate drivers with ecological processes and the set of ecosystem services provided. The determination of environmental and socio-economic drivers can be obtained from scientists, water resource managers and policy makers, who have an expert understanding of the evaluated river system and its catchment (Peterson *et al*., 2003). With a set of environmental and socio-economic drivers elicited, they can be ranked using network analysis. This analytical method is appropriate because it recognizes the interrelatedness of drivers, ecological processes and ecosystem services. This ranking of drivers provides the basis for constructing multiple scenarios obtained from water resource managers and policy makers for the proposed water storage project. Moreover, ecological processes and the set of ecosystem services provided by the river system can be developed with the network analysis providing the potential for dynamic simulations to be performed allowing possible future trends in ecosystem services to be determined.

While controversial for the application of scenario analysis, uncertainty associated with *ex-ante* evaluations can be quantified through the subjective elicitation of probability weights obtained from water resource managers and policy makers. Where probability weights are applied, then they can be multiplied to the ecosystem services index for that future outcome to provide an uncertainty-adjusted index useful for cost utility analysis being performed for *ex-ante* evaluations of projects. Importantly, in order for the ecosystem services index to be constructed for *ex-ante* evaluations, it will be necessary for subjective elicitations of the normalized indicator scores to be performed by water resource managers and policy makers. With normalized indicator scores elicited for multiple scenarios, it allows safe minimum standards to be determined with future outcomes, trends and changes in mind. This is significant as the establishment of safe minimum standards from present conditions alone neglects to consider that future generations are likely to face different water resource problems than those faced today (Kontogianni *et al*., 2010). Hence, scenario analysis not only provides an appropriate analytical method for *ex-ante* evaluations of proposed water storage projects, but it also allows issues of cost-effectiveness and sustainability to be addressed in ways that would not be transparent otherwise.

While this paper has developed a systematic way to perform evaluations of water storage projects using an ecosystem services approach, it is apparent that the management of water resources had focused on meeting demand for freshwater by increasing its reliability and supply through the construction of water storage projects. Nevertheless, scientists and economists alike are beginning to voice concerns about the strictly supply-side orientation of the management of water resources (World Commission on Dams, 2000). Current research is increasingly focused on integrating supply-side and demand-side information into water resource management. In particular, research is investigating ways to improve the efficient allocation of water resources and limit freshwater demand by appropriately pricing the consumptive use of water resources and educating society about recirculating and rationing water resource usage (Renzetti, 2002). This demand-side research is significant as the increasing demand and lack of efficient use of water resources is largely the result of water use being unpriced (Grafton, 2009). More efficient use of water resources requires prices that more accurately reflect their actual value. One way to establish these prices is the development of water markets, such as those developed in Chile, which has resulted in the improved allocation of water resources and the decreased construction of expensive water storage projects (Brehm & Quiroz, 1995). However, even with the development of water markets it is recognized that the ecosystem services approach devised in this paper remains valid and necessary to ensure that all values are considered for the cost-effective management and sustainable use of water resources.

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7.0 References

Alvarez-Farizo, B. and Hanley, N. (2006). Improving the process of valuing non-market benefits: Combining citizens' juries with choice modelling. *Land Economics*, 82: 465-478.

Balmford, A., Bruner, A., Cooper, P., Costanza, R., Farber, S., Green, R. E., Jenkins, M., Jefferiss, P., Jessamy, V., Madden, J., Munro, K., Myers, N., Naeem, S., Paavola, J., Rayment, M., Rosendo, S., Roughgarden, J., Trumper, K. and Turner, R. K. (2002). Economic reasons for conserving wild nature. *Science*, 9: 950-953.

Barkmann, J., Glenk, K., Keil, A., Leemhuis, C., Dietrich, N., Gerold, G. and Marggraf, R. (2008). Confronting unfamiliarity with ecosystem functions: The case of ecosystem services approach to environmental valuation with stated preference methods. *Ecological Economics*, 65: 48-62.

Baskaran, R., Cullen, R. and Colombo, S. (2010). Testing different types of benefit transfer in valuation of ecosystem services: New Zealand winegrowing case studies. *Ecological Economics*, 69: 1010-1022.

Bossel, H. (1999). *Indicators for Sustainable Development*: *Theory, Methods, Applications*. International Institute for Sustainable Development: Winnipeg.

Bouwer, H., 2000. Integrated water management: Emerging issues and challenges. *Agricultural Water Management*, 45: 217-228.

Boyd, J. and Banzhaf, S. (2007). What are ecosystem services? The need for standardized environmental accounting units. *Ecological Economics*, 63: 616-626.

Brehm, M. R. and Quiroz, J. (1995). *The Market for Water Rights in Chile*. World Bank: Washington.

Canterbury Regional Council (1990). Opihi-Temuka River and Catchment: Issues and Options. Canterbury Regional Council: Christchurch.

Canterbury Regional Council (1995). Opihi River Augmentation. Canterbury Regional Council: Christchurch.

Capistrano, D., Samper, C. K., Lee, M. J. and Raudsepp-Hearne, C. (*eds*.). (2006). *Ecosystems and Human Well-bring. Findings of the Sub-Global Assessments Working Group for the Millennium Ecosystem Assessment*. Island Press: Washington.

Carpenter, S. R. and Brock, W. A. (2004). Spatial complexity, resilience and policy diversity: Fishing on lake-rich landscapes. *Ecology and Society*, 9: 8.

Carpenter, S. R., DeFries, R., Dietz, T., Mooney, H. A., Polasky, S., Reid, W. V. and Scholes, R. J. (2006). Millennium Ecosystem Assessment: Research needs. *Science*, 314: 257-258.

Chisholm, R. A. (2010). Trade-offs between ecosystem services: Water and carbon in a biodiversity hotspot. *Ecological Economics*, in press.

Ciriacy-Wantrup, S. V. (1952). *Resource Conservation*: *Economics and Policies*. University of California Press: Berkeley.

Colombo, S., Angus, A., Morris, J., Parsons, D. J., Brawn, M., Stacey, K. and Hanley, N. (2009). A comparison of citizen and expert preferences using an attribute-based approach to choice. *Ecological Economics*, 68: 2834-2841.

Cortner, H. and Moote, M. (1994). Trends and issues in land and water resources: Setting the agenda for change. *Environmental Management*, 18: 167-173.

Costanza, R. (*ed*.) (1991). *Ecological Economics: The Science and Management of Sustainability*. Columbia University Press: New York.

Costanza, R., d'Arge, R., de Groot, R., Farber, S., Grasso, M., Hannon, B., Limburg, K., Naeem, S., O'Neill, R., Paruelo, J., Raskin, R. G., Sutton, P. and van den Belt, M. (1997). The value of the world's ecosystem services and natural capital. *Nature*, 387: 253-260.

Costanza, R., Low, B. S., Ostrom, E. and Wilson, R. J. (2001). Ecosystems and human systems: A framework for exploring the linkages. *In* Costanza, R., Low, B.S., Ostrom, E. and Wilson, R. J. (*eds*.). *Institutions, Ecosystems and Sustainability*. CRC Press: New York.

Cullen, R., Fairburn, G. A. and Hughey, K. F. D. (2001). Measuring the productivity of threatened species programs. *Ecological Economics*, 39: 53-66.

Dacker, W. (1990). Belonging here/Toi Taketake. The People and the Place: Mahika Kai. New Zealand Commission: Wellington.

Daily, G. C. (*ed*.) (1997). *Nature's Services*: *Societal Dependence on Natural Ecosystems*. Island Press: Washington.

Daly, H. E. (1991). Elements of environmental macroeconomics. *In* Costanza, R. (*ed*.). *Ecological Economics*: *The Science and Management of Sustainability*. Columbia University Press: New York.

De Groot, R. S., Wilson, M. A. and Boumans, R. M. J. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological Economics*, 41: 393-408.

De Groot, R. S., 2006. Function-analysis and valuation as a tool to assess land use conflicts in planning for sustainable, multi-functional landscapes. *Landscape and Urban Planning*, 75: 175-186.

De Groot, R. S., Alkemade, R., Braat, L., Hein, L. and Willemen, L. (2009). Challenges in integrating the concept of ecosystem services and values in landscape planning management and decision making. *Ecological Complexity*, in press.

De Joux, R. T. (1982). The water resources of the Opihi and Temuka Rivers. South Canterbury Catchment Board and Regional Water Board: Timaru.

Dietz, S. and Neumayer, E. (2007). Weak and strong sustainability in the SEEA: Concepts and measurement. *Ecological Economics*, 61: 617-626.

Dominati, E., Patterson, M. and Mackay, A. (2010). A framework for classifying and quantifying the natural capital and ecosystem services of soil. *Ecological Economics*, 69: 1858-1868.

Dyson, M., Bergkamp, G. and Scanlon, J. (2003). *Flow*: *The Essentials of Environmental Flows*. IUCN: Gland.

Earl, P. E. (1986). *Lifestyle Economics*: *Consumer Behaviour in a Turbulent World*. Wheatsheaf Books: Brighton.

Ehrlich, P. R. (1996). Conservation in temperature forests: What do we need to know and do. *Forest Ecology and Management*, 85: 9-19.

Ekins, P., Simon, S., Deutsch, L., Folke, C. and de Groot, R. (2003). A framework for the practical application of the concepts of critical natural capital and strong sustainability. *Ecological Economics*, 44: 165-185.

Farber, S. C., Costanza, R., Childers, D. L., Erickson, J., Gross, K., Grove, M., Hopkinson, C. S., Kahn, J., Pincetl, S., Troy, A., Warren, P. and Wilson, M. (2006). Linking ecology and economics in ecosystem management. *BioScience*, 56: 121-132.

Faucheux, S. and O'Connor, M. (1998). Weak and strong sustainability. *In* Faucheux, S. and O'Connor, M. (*eds*.). *Valuation for Sustainable Development*: *Methods and Policy Indicators*. Edward Elgar: Cheltenham.

Fisher, B., Turner, R. K. and Morling, P. (2009). Defining and classifying ecosystem services for decision making. *Ecological Economics*, 68: 643-653.

Frame, B. and Russell, S. (2009). Water allocation: Canterbury's wicked problem. *In* Frame, B., Gordon, R. and Mortimer, C. (*eds*.). *Hatched*: *The Capacity for Sustainable Development*. Landcare Research: Lincoln.

Food and Agricultural Organization (2000). New Dimensions in Water Security – Water, Society and Ecosystem Services in the 21st Century. Land and Water Development Division. Food and Agriculture Organization of the United Nations: Rome.

Food and Agricultural Organization (2006). Water Monitoring: Mapping Existing Global Systems and Initiatives Background Document. Prepared by the FAO on behalf of the UN-Water Task Force on Monitoring Stockholm, 21 August 2006.

Functowicz, S. O., Martinez-Alier, J., Munda, G. and Ravetz, J. (2001). Multicriteria-based environmental policy. *In* Abaza, H. and Branazini, A. (*eds*.). *Implementing Sustainable Development*: *Integrated Assessment and Participatory Decision-Making Processes*. Edward Elgar: Cheltenham.

Goulder, L. H. and Kennedy, D. (1997). Valuing ecosystem services: Philosophical basis and empirical methods. *In* Daily, G. C. (*ed*.). *Nature's Services*: *Societal Dependence on Natural Ecosystems*. Island Press: Washington.

Gowan, C., Stephenson, K. and Shabman, L. (2006). The role of ecosystem valuation in environmental decision making: Hydropower relicensing and dam removal on the Elwha River. *Ecological Economics*, 56: 508-523.

Gowdy, J. M. (1997). The value of biodiversity: Markets, societies, and ecosystems. *Land Economics*, 73: 25-41.

Graf, W. L. (2006). Downstream hydrologic and geomorphic effects of large dams on American rivers. *Geomorphology*, 79: 336-360.

Grafton, R. Q. (2009). Introduction. *In* Grafton, R. Q. (*ed*.). *The Economics of Water Resources*. Edward Elgar: Cheltenham.

Harris Consulting (2006). The Opuha Dam: An *ex-post* study of its impacts on the provincial economy and community. Environment Canterbury: Christchurch.

Harris Consulting (2009). Modelling the Canterbury Water Management Strategy area: Draft report prepared for Environment Canterbury. Environment Canterbury: Christchurch.

Hearnshaw, E. J. S. (2009). A post-classical economic approach to ecosystem management. Unpublished PhD thesis, Lincoln University: Christchurch.

Hearnshaw, E. J. S., Cullen, R. and Hughey, K. F. D. (2010). Ecosystem services review of water storage projects in Canterbury: The Opihi River case. Faculty of Commerce Report, Lincoln University: Christchurch.

Hein, L., Koppen, K. V., de Groot, R. S. and Ireland, E. C. (2006). Spatial scales, stakeholders and the valuation of ecosystem services. *Ecological Economics*, 57: 209-228.

Hoeinghaus, D. J., Agostinho, A. G., Gomes, L. C., Pelicice, F. M., Okada, E. K., Latini, J. D., Kashiwaqui, E. A. L. and Winemiller, K. O. (2009). Effects of river impoundment on ecosystem services of large tropical rivers: Embodied energy and market value of artisanal fisheries. *Conservation Biology*, 23: 1222-1231.

Howarth, R. B. (2007). Towards an operational sustainability criterion. *Ecological Economics*, 63: 656-663.

Jewitt, G. (2002). Can integrated water resources management sustain the provision of ecosystem goods and services? *Physics and Chemistry of the Earth*, 27: 887-895.

Kuik, O. J. and Gilbert, A. J. (1999). Indicators of sustainable development. *In* Van den Bergh, J. C. J. M (*ed*.). *Handbook of Environmental and Resource Economics*. Edward Elgar: Northampton.

Kontogianni, A., Luck, G. W. and Skourtos, M. (2010). Valuing ecosystem services on the basis of service-providing units: A potential approach to address the ‘endpoint problem’ and improve stated preference methods. *Ecological Economics*, 69: 1479-1487.

Kosoy, N. and Corbera, E. (2009). Payments for ecosystem services as commodity fetishism. *Ecological Economics*, 69: 1228-1236.

Kremen, C. and Ostfeld, R. S. (2005). A call to ecologists: Measuring, analyzing and managing ecosystem services. *Frontiers in Ecology and Environment*, 3: 540-548.

Layke, C. (2009). Measuring nature’s benefits: A preliminary roadmap for improving ecosystem service indicators. World Resources Institute: Washington.

Lamb, E. G., Bayne, E., Holloway, G., Schieck, J., Boutin, S., Herbers, J. and Haughland, D. L. (2009). Indices for monitoring biodiversity change: Are some more effective than others? *Ecological Indicators*, 9: 432-444.

Lockwood, M. (1996). Non-compensatory preference structures in non-market valuation of natural area policy. *Australian Journal of Agricultural Economics*, 40: 85-101.

Loomis, J., Kent, P., Strange, L., Fausch, K. and Covich, A. (2000). Measuring the total economic value of restoring ecosystem services in an impaired river basin: Results from a contingent valuation study. *Ecological Economics*, 33: 103-117.

Losos, E., Hayes, J., Phillips, A., Wilcove, D. and Alkire, C. (1995). Taxpayer-subsidized resource extraction harms species: Double jeopardy. *BioScience*, 45: 446-45.

Mooney, H., Cropper, A. and Reid, W. (2005). Confronting the human dilemma: How can ecosystems provide sustainable services to benefit society? *Nature*, 434: 561-562.

Munda, G. Nijkamp, P. and Rietveld, P. (1994). Qualitative multicriteria evaluation for environmental management. *Ecological Economics*, 10: 97-112.

Naidoo, R., Balmford, A., Costanza, R., Fisher, B., Green, R. E., Lehner, B., Malcolm, T. R. and Ricketts, T. H. (2008). Global mapping of ecosystem services and conservation priorities. *Proceedings of the National Academy of Sciences*, 105: 9495-9500.

National Research Council (2005). *Valuing Ecosystem Services*: *Toward Better Environmental Decision-Making*. National Academy Press: Washington.

Navrud, S. (2001). Economic valuation of inland recreational fisheries: Empirical studies and their policy use in Norway. *Fisheries Management and Ecology*, 8: 369-382.

Neumayer, E. (2003). *Weak versus Strong Sustainability*: *Exploring the Limits of Two Opposing Paradigms*. Edward Elgar: Northampton.

Norgaard, R. B. (2010). Ecosystem services: From eye-opening metaphor to complexity blinder. *Ecological Economics*, 69: 1219-1227.

Parkyn, S. M. and Wilcock, R. J. (2004). Impacts of agricultural land use. *In* Harding, J. S., Mosley, M. P., Pearson, C. P. and Sorrell, B. K. (*eds*.). *Freshwaters of New Zealand*. New Zealand Hydrological Society and New Zealand Limnological Society: Christchurch.

Pearce, D. W., Barbier, E. and Markandya, A. (1990). *Sustainable Development*: *Economics and Environment in the Third World*. Edward Elgar: Aldershot.

Peterson, G. D., Beard, D., Beisner, B., Bennent, E., Carpenter, S., Cumming, G., Dent, L. and Havlicek, T. (2003). Assessing future ecosystem services: A case study of the northern highland lake district, Wisconsin. *Conservation Ecology*, 7: 1.

Prato, T. (1999). Multiple attribute evaluation of landscape management. *Journal of Environmental Management,* 60: 325-337.

Prato, T. (2007). Selection and evaluation of projects to conserve ecosystem services. *Ecological Modelling*, 203: 290-296.

Raymond, C. M., Bryan, B. A., Hatton MacDonald, D., Cast, A., Strathearn, S., Grandgirard, A. and Kalivas, T. (2009). Mapping community values for natural capital and ecosystem services. *Ecological Economics*, 68: 1301-1315.

Rayner, S. (2006). Wicked problems: Clumsy solutions. www.martininstitute.ox.ac.uk

Renzetti, S. (2002). *The Economics of Water Demands*. Kluwer Academic: Boston.

Rodríguez, J. P., Beard, T. D., Bennett, E. M., Cumming, G. S., Cork, S., Agard, J., Dobson, A. P. and Peterson, G. D. (2006). Trade-offs across space, time, and ecosystem services. *Ecology and Society*, 11: 28.

Saaty, T. L. (1980). *The Analytic Hierarchy Process*. McGraw-Hill: New York.

Saaty, T. L. (1995). *Decision Making for Leaders*: *The Analytic Hierarchy Process in a Complex World*. RWS Publications: New York.

Scarf, F., Waugh, J. R. and Swete, K. N. (1984). Opihi River Water Management Plan 1984-1990. South Canterbury Catchment and Regional Water Board: Timaru.

Scudder, T. (2005). *The Future of Large Dams*: *Dealing with Social, Environmental, Institutional and Political Costs*. Earthscan: London.

Spash, C. and Vatn, A. (2006). Transferring environmental value estimates: Issues and alternatives. *Ecological Economics*, 60: 379-388.

Stephenson, K. and Shabman, L. (2001). The role of nonmarket valuation in hydropower relicensing: An application of a pattern modelling approach. *Journal of Economic Issues*, 35: 497-504.

Straton, A. (2006). A complex systems approach to the value of ecological resources. *Ecological Economics*, 56: 402-411.

Swanson, T. M. (1995). *The Economics and Ecology of Biodiversity Decline*. Cambridge University Press: Cambridge.

Swinton, S. M., Lupi, F., Robertson, G. P. and Hamilton, S. K. (2007). Ecosystem services and agriculture: Cultivating agricultural ecosystems for diverse benefits. *Ecological Economics*, 64: 245-252.

Van der Heijden, K. (1996). *Scenarios*: *The Art of Strategic Conversation*. Wiley: New York.

Wainger, L. A., King, D. M., Mack, R. N., Price, E. W. and Maslin, T. (2010). Can the concept of ecosystem services be practically applied to improve natural resource management decisions? *Ecological Economics*, 69: 978-987.

Wilson, M., Boumans, R., Costanza, R. and Liu, S. (2005). Integrated assessment and valuation of ecosystem goods and services provided by coastal systems. *In* Wilson, J. G. (*ed*.). *The Intertidal Ecosystem*: *The Value of Ireland's Shores*. Royal Irish Academy: Dublin.

Wilson, M. and Liu, S. (2008). Non-market value of ecosystem services provided by coastal and nearshore marine systems. *In* Patterson, M. and Glavovic, B. (*eds*.). *Ecological Economics of the Oceans and Coasts*. Edward Elgar: Cheltenham.

Winkler, R. (2006). Valuation of ecosystem goods and services. Part 1: An integrated dynamic approach. *Ecological Economics*, 59: 82-93.

World Commission on Dams (2000). *Dams and Development*: *A New Framework for Decision-Making*. Earthscan: London.

World Resources Institute (2008). *Ecosystem Services*: *A Guide for Decision Makers*. World Resources Institute: Washington.

Worrall, J. (2007). *A Dream Fulfilled – The Story of the Opuha Dam*. Opuha Dam Limited: Timaru.

Young, R. A., Smart, G. and Harding, J. (2004). Impacts of hydro-dams, irrigation schemes and river control works. *In* Harding, J. S., Mosley P., Pearson, C. and Sorrell, B. (*ed*). *Freshwaters of New Zealand*. New Zealand Hydrological Society and New Zealand Limnological Society: Christchurch.

Young, R. A. (2005). Economic criteria for water allocation and valuation. *In* Brouwer, R. and Pearce, D. (*eds*.). *Cost-benefit Analysis and Water Resources Management*. Edward Elgar: Cheltenham.

Zhang, X. Y. and Liu, X. G. (2009). Multiple criteria evaluation of ecosystem services for the Ruoergai Plateau Marshes in southwest China, *Ecological Economics*, in press.