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Evaluating the sustainability of impounded river systems and the cost-effectiveness of dam projects: An ecosystem services approach

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Keywords: Cost utility analysis, dam projects, ecosystem services, impounded river systems, indicators and sustainability.

1.0 Introduction

In recent times, there has been increasing demand in the Canterbury region of New Zealand for the abstraction of water from rivers. Much of the abstracted water is used for irrigating agricultural land, which enables farmers to intensify their agricultural operations through increased stocking rates or a change toward more productive land uses (*e.g.* sheep farming to dairy farming). While much irrigation in Canterbury uses run-of-river water schemes, there is a realization that this water supply is scarce and, in some cases, has reached its maximum allocation limits while retaining acceptable minimum river flows needed to sustain aquatic health (Dyson *et al.*, 2003; Canterbury Mayoral Forum, 2009). This realization has led to the increased interest and subsequent investment in projects involving water storage. Water storage can be achieved either through river diversion to a nearby off-stream reservoir or river impoundment by way of dam construction. With regards to river impoundment, water is stored upstream of the dam through the creation of an artificial reservoir, while they also regulate, stabilize and augment minimum river flows downstream (Graf, 2006). This capacity of dams to increase the supply and reliability of water makes them particularly beneficial to farmers, as they are able to plan the irrigation of their farms with confidence allowing them to intensify their land use in an attempt to maximize profits.

While river impoundment can result in significant benefits to farmers, it also can come at a 'cost', especially to river ecology. For example, Losos *et al.* (1995) found that dams have resulted in more degradation of threatened species and their habitats than any other activity involving environmental resources. Arguably the impetus for investing in dams has been with (short-term) financial returns and not in sustaining the aquatic health of river systems (Dyson *et al.*, 2003). Scientists have long recognized the negative impact of land use intensification on rivers. Land use intensification often leads to a substantial increase in nutrients (*e.g.* nitrates) from the increased application of fertilizers (Harris *et al.*, 2006). The nutrients applied with intensified agricultural practices can, through surface runoff, degrade the ecology of rivers by the excessive proliferation of algae amongst other things.

Given the potential benefits and costs from water abstraction and river impoundment, it is essential that the overall impact of dams on river systems, and all their associated values, is evaluated. The evaluation of dams provides information for learning in accordance with adaptive water resource management and improved understanding as to whether the original motivations for dam investment remains valid (World Wildlife Fund, 2003). Despite its need, such evaluations are rarely performed; even though dams are costly investments that have significant impact on river systems that provide much human well-being.

In 2000 the World Commission on Dams was established to examine appropriate means of evaluation. A cost analysis of various dam projects revealed that the actual cost of dams often exceeds their projected costs. Additionally, non-consumptive uses of water were frequently ignored and the benefits from river impoundments often exaggerated (World Commission on Dams, 2000). Several authors have thought it inappropriate to only consider those tangible use values that relate to the consumptive use of water resources (Cortner & Moote, 1994; Jewitt, 2002; Frame & Russell, 2009). The exaggeration of some benefits coupled with the ignorance of others signals a critical failure in water resource management with regards to discerning the actual return on investment of a dam and its impact on the ecology and sustainability of the impounded river system.

1.1 Ecosystem Services

The need to evaluate the multiple values provided by river systems has led to the consideration of using an ecosystem services approach. This approach has been popularized by some notable studies (*e.g.* Costanza *et al.*, 1997), including the landmark *Millennium Ecosystem Assessment* (Capistrano *et al.*, 2005). Specifically, ecosystem services are the collection of goods and services provided by ecosystems (*e.g.* rivers) that benefit the well-being of humans (Daily, 1997; National Research Council, 2005). Ecosystem services are the connection between ecological processes and humans that value them for their well-being.

To date, while numerous researchers have recognized the potential of the ecosystem services approach for considering the many values provided by ecosystems, including river systems, the relevant literature reveals that only some ecosystem services are regularly considered (*e.g.* Water Supply, Recreational Values) (De Groot *et al.*, 2009). Moreover, there are few studies that have examined the change in ecosystem services provided by river systems when evaluating the impacts of impoundment (Hoeinghaus *et al.*, 2009). An underlying reason for the uneven distribution of research into ecosystem services is that there is still much debate on how to apply and implement the approach. In particular, researchers have conflicting views as to the inclusion of intermediate ecosystem services in addition to final ecosystem services that provide direct benefits to humans (Boyd & Banzhaf, 2007; Fisher *et al.*, 2009). This debate is complex in that the determination as to whether an ecosystem service is final or intermediate is context-dependent (Turner *et al.*, 2010). For example, the ecosystem service Water Supply may be considered final to a hydroelectric company, but only an intermediate ecosystem service to an angler.

Despite a number of classifications devised, the set of ecosystem services established in the *Millennium Ecosystem Assessment* remains the most recognizable and well-developed (Raymond *et al.*, 2009). This classification incorporates four classes of ecosystem services including provisioning, regulating, cultural and supporting. This paper adopts this classification, however, does not utilise supporting services as it was determined that they more resembled ecosystem processes than ecosystem services within the context of this study. Hence, the three classes examined are: provisioning ecosystem services (*e.g.* Water Supply), which provide use benefits through goods that are obtained from the ecosystem; cultural ecosystem services (*e.g.* Spiritual Values), which provide less tangible non-material benefits including non-use benefits; and regulating ecosystem services (*e.g.* Erosion Control), which provide benefits through controlling and regulating various ecological processes.

Given that the ecosystem services approach allows for the consideration of multiple values this paper applies the approach to evaluate the cost-effectiveness of a dam and the sustainability of the impounded river system. Specifically, the Opuha Dam and the Opihi River are evaluated, which are located in the particularly drought-prone region of South Canterbury. The remainder of the paper is structured as follows. In *Section 2* various evaluation methods are discussed to determine the cost-effectiveness of dam projects and the sustainability of impounded river systems. This leads to the promotion of cost utility analysis and the need to construct an ecosystem services index, which is formed through the aggregation of indicators that represent the set of ecosystem services provided. With appropriate methods of evaluation determined, *Section 3* overviews the Opihi River and the Opuha Dam. Then, in *Section 4* the two components required for the apt construction of an ecosystem services index is considered; that is, indicators that capture each ecosystem service and

preferential weights of each ecosystem service that depict the value of the set of ecosystem services to various stakeholder groups. *Section 5* presents the evaluation results, which reveal the sustainability of the impounded Opihi River and the cost-effectiveness of the Opuha Dam. Lastly, in *Section 6* conclusions are stated and limitations are discussed.

2.0 Evaluation Methods

While cost-benefit analysis is put forward as the most appropriate method for evaluating investments such as dam projects, it is unlikely to be capable of considering all ecosystem services affected by a dam. Frequently cost-benefit analysis accounts only for tangible use values that are conspicuous. This deficiency arises because only tangible use values are easily captured and quantified in monetary terms through the availability of market prices (Young *et al*., 2004; Farber *et al*., 2006). Because many ecosystem services are less tangible and not currently captured by markets they are often undervalued or erroneously given an implicit value of zero (Loomis *et al*., 2000; Navrud, 2001; Dyson *et al*., 2003; National Research Council, 2005; Troy & Wilson, 2006; Barkmann *et al*., 2008).

In order to tackle this economic problem, environmental economists have devised a number of non-market valuation methods (*e.g*. contingent valuation, choice modelling) in the absence of actual functioning 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 and analysing information from a large sample population of affected stakeholders. This can make the undertaking of non-market valuation costly, labour intensive and time-consuming (Gowdy, 1997; National Research Council, 2005; Baskaran *et al*., 2010).

In order to reduce these impracticalities of non-market valuation, the benefit transfer method is often promoted and employed (*e.g*. Costanza *et al*., 1997; Baskaran *et al*., 2010). This method uses monetary values obtained from previous non-market valuations of river systems worldwide and transfers these values to monetize less tangible values for the particular river system examined. However, numerous errors (*e.g*. measurement errors, transfer errors) can accrue with the benefit transfer method (Rosenberger & Stanley, 2006; Turner *et al*., 2010). Transfer errors are particularly problematic and arise from benefits being less transferable than the method would imply. Indeed, those that transfer benefits must assume that all ecosystems of a certain type (*e.g*. rivers) can be treated as if they are much alike. But, empirical evidence indicates that ecosystems are complex with dynamics that are path dependent and site-specific (*e.g*. Jordano *et al*., 2003). Hence, assuming correspondence between ecosystems can lead to the estimation of wildly inaccurate monetary values (Carpenter & Brock, 2004; National Research Council, 2005; Spash & Vatn, 2006; Naidoo *et al*., 2009).

Fortunately, while not well-known, there are alternative evaluation methods to cost-benefit and benefit-transfer analyses that are practical and useful. One appropriate method, which remains underutilized by economists, is cost utility analysis. This evaluation method maintains the cost function in monetary terms, but measures the outcome (or benefit) function through the construction of a single non-monetary metric that aggregates multiple values into a 'utility' index. The development of an index makes cost utility analysis synonymous with multi-criteria analysis.

Multi-criteria analysis is an overarching term depicting a set of methods many of which are capable of weighting and aggregating multiple values together (Munda *et al*., 1994). The capacity of using an index is significant as it allows all (or almost all) values to be evaluated without having to monetize less tangible values that are more appropriately left in their own terms (Wainger *et al*., 2010).

2.1 Indices & Indicators

In order to establish an index that considers the impacts on ecosystem services from river impoundment, there is a need to ascertain changes of ecosystem services over time in quantitative terms. While the understanding and measurement of ecosystem services is a critical area of ongoing research (Carpenter *et al*., 2006; De Groot *et al*., 2009), a suitable means of revealing ecosystem service changes in quantitative terms is through the use of indicators. It is with indicators that ecosystem services can be captured, which is significant as most ecosystem services are difficult to measure directly because of the complexity of ecosystems (World Resources Institute, 2008). Indeed, indicators are able to "summarize complex information of value to the observer. They condense … complexity to a manageable amount of meaningful information …" (Bossel, 1999; p. 8). Despite the usefulness of indicators for representing ecosystem services, they remain underdeveloped. There are no indicators that are fully agreed upon for the monitoring of each ecosystem service for a particular ecosystem type, no matter the classification employed. However, De Groot *et al*. (2009) recently highlighted a number of indicators that could partially capture various ecosystem services.

One reason why no well-defined list has been established is that each ecosystem service is difficult to adequately capture by a single indicator that is depicted strictly in environmental terms. A means of more comprehensively capturing an ecosystem service is the use of multiple indicators from both environmental and socio-economic perspectives. In using multiple indicators an ecosystem service is represented through the collective communication of all appropriate indicators for that ecosystem service. While the methodological development of using environmental and socio-economic indicators together opposes the position held by Boyd and Banzhaf (2007), it supports the widely promoted view in the sustainability literature of extending evaluations beyond environmental perspectives alone (Hacking & Guthrie, 2008). Indeed, by capturing both environmental and socio-economic indicators, the objective ecosystem dimension and subjective human dimension of an ecosystem service is measured. This is significant as ecosystem services captured solely by a single monetary metric fail to reveal information about the actual status of the impounded river system (Rosenberger & Stanley, 2006). Similarly, when conducting an evaluation with environmental indicators alone, as is typical of impact analyses (Lamb *et al*., 2009), socio-economic realities are inherently ignored (Straton, 2006; Winkler, 2006). Therefore, in this paper an effort is made to avoid this one-sidedness through endeavouring to represent each ecosystem service by both environmental and socio-economic indicators wherever possible.

The indicators used to represent the set of ecosystem services provided, when aggregated, allow for the construction of an index, which is henceforth termed the 'ecosystem services index'. The use of indices is not new for evaluating rivers (*e.g*. Qualitative Macro invertebrate Community Index) nor is the idea of an index of ecosystem services; as this concept was originally proposed by Banzhaf and Boyd (2005). While indices are widely employed and have been considered an efficient instrument for evaluating running waters (Hering *et al*., 2006), there remains little consensus as to how best to construct indices (Saisana & Saltelli, 2008) including those focused on ecosystem services.

This paper constructs an ecosystem services index by selecting a set of indicators that represent each ecosystem service. These indicators are subsequently normalized onto a standardized scale, and preferential weights for each ecosystem service that reflect the preferences of all stakeholders are applied. The determination of preferential weights is necessary as stakeholder groups have differing priorities as to their importance. Once preferential weights are quantified, an index can be estimated by multiplying the weight of each ecosystem service with the average normalized score from the selected indicators that represent that ecosystem service. These products are then summed together to form an aggregated set of indicators or ecosystem services index of the river system evaluated (see *Equation 1*).

$$\mathrm{ESI} = \sum w_n s_{in} \qquad \textit{Equation 1}$$

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

2.2 Cost-Effectiveness & Sustainability

Cost utility analysis has been promoted for the evaluation of the cost-effectiveness of dam projects. In order to perform cost utility analysis an ecosystem services index must be constructed and monetary costs of river management and the dam project itself (*e.g.* dam maintenance costs, dam running costs) are required. These monetary costs coupled with the ecosystem services index allow cost-utility ratios to be calculated and in turn the cost-effectiveness of the dam project inferred. One indication of the cost-effectiveness of the dam evaluated would be if the cost-utility ratio post-dam was less than that calculated pre-dam.

The ecosystem services index also provides a means to evaluate the sustainability of the impounded river system, where sustainability is defined as aggregated welfare that is non-declining over the long-term (Pearce *et al.*, 1990; Neumayer, 2003). Non-declining aggregated welfare ensures that future generations are provided with at least the same well-being from ecosystem services as present generations. Hence, if the ecosystem services index is non-declining over the long-term then the river system can be considered 'sustainable' or at least progressing towards sustainability. This determination of sustainability reflects the 'weak' sustainability criterion because it assumes that all ecosystem services are compensatory and, therefore, commensurable and reducible to a single metric; in this case an ecosystem services index (see *Figure 1*). For example, an ecosystem services index implies that a high scoring Recreational Values ecosystem service can compensate a low scoring Water Regulation ecosystem service.

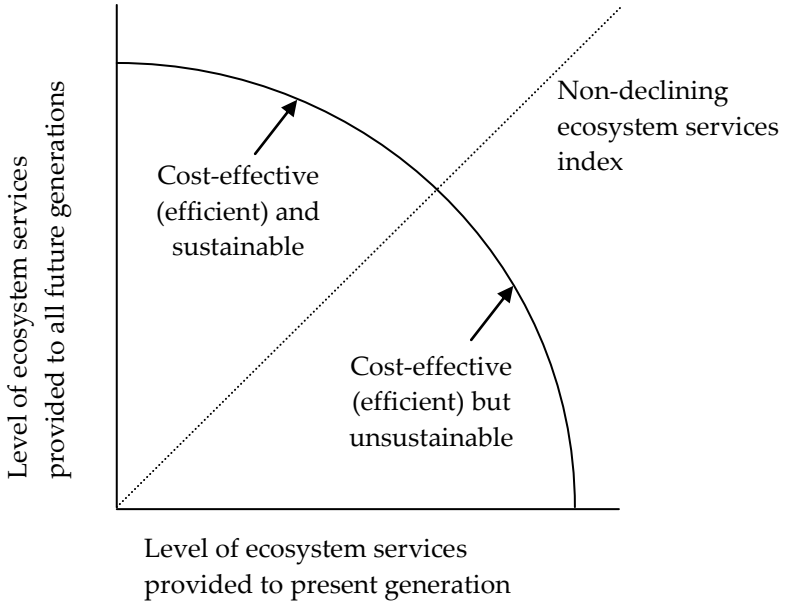

***Figure 1*:** A non-declining index indicates weak sustainability as future generations are provided with at least the same well-being as present generations (adapted from Norgaard, 2010).

In allowing for compensation, the ecosystem services index neither is able to consider who explicitly gains and losses from the dam project nor consider the criterion of 'strong' sustainability. Unlike weak sustainability, strong sustainability considers welfare in non-compensatory terms, so that the evaluation of strong sustainability by an ecosystem services index is inappropriate (Faucheux & O'Connor, 1998). For this reason, strong sustainability recognizes that measuring the actual delivery of an ecosystem service through indicators does not necessarily indicate whether the ecological processes that produce ecosystem services are sustainable (Mooney *et al*., 2005). Ideally then, both the weak and strong criteria of sustainability should be accounted for when evaluating an impounded river system , as claims of sustainability are mistaken unless both the well-being of future generations and the ecological processes that produce the set of ecosystem services are maintained.

The difficulty with strong sustainability has been in making the criterion practicable (Prato, 2007). (Turner *et al*., 2010). However, Costanza (1991) has maintained that strong sustainability can be made operational by translating it in terms of a safe minimum standard, a concept first introduced by Ciriacy-Wantrup (1952). Specifically, a safe minimum standard indicates a threshold where an ecosystem service represented by a set of indicators should not be breached in order to sustain its supply. Beyond the threshold (*e.g*. below the acceptable minimum river flow) the supply of an ecosystem service may change abruptly leading to potential 'irreversible' losses (Costanza *et al*., 2001; Norgaard, 2010). This concept of a safe minimum standard (or threshold) should not be foreign to water resource managers as it is widely applied through the determination of acceptable minimum river flows for sustaining aquatic health. Despite this, a critical reason for poor water resource management in New Zealand is that thresholds are neither adequately acknowledged nor managed for (Land and Water Forum, 2010). Yet, when incorporating the strong sustainability criterion into an evaluation thresholds are acknowledged, so that they can be managed for and not breached in order to sustain benefits for human well-being.

In applying the safe minimum standard, strong sustainability in its most complete form would be observed where safe minimum standards for the set of indicators representing all ecosystem services have been met and are therefore not breached. It is, of course, unlikely that strong sustainability in this most complete form will be demonstrated, especially when one considers that approximately two thirds of all ecosystem services provided worldwide are presently degraded (Capistrano *et al*., 2005). Where a safe minimum standard is breached, there are several evaluation methods for the determination as to whether the strong sustainability of an impounded river system has progressed or regressed since dam construction. One simple method is the checklist approach where a count is determined of the number of safe minimum standards breached pre-dam compared with the number of safe minimum standards breached post-dam. One difficulty with this method is it assumes that all ecosystem services are equally preferred. However, other evaluation methods can be applied including refinements of this checklist approach outlined above.

An alternative evaluation method is the lexicographic-based characteristic filtering rule. This evaluation method does not assume that all ecosystem services are equally preferred. Rather, in using the preferential weights for the set of ecosystem services, a hierarchy of ecosystem services can be established from most to least preferred. In establishing a hierarchical ranking of ecosystem services the characteristic filtering rule can be applied to assess the strong sustainability of an impounded river system. This is possible because the characteristic filtering rule uses the hierarchical ranking of ecosystem services to filter each ecosystem service, so that ecosystem services are evaluated from most to least preferred. The characteristic filtering rule is applied first to the most preferred ecosystem service and establishes whether the safe minimum standards for all indicators that represent that ecosystem service have been breached or not, both pre-dam and post-dam (Earl, 1986; Lockwood, 1996). Where in both periods (*i.e.* pre-dam or post-dam) all safe minimum standards are passed or at least one safe minimum standard for an indicator is breached in both periods in any one year, then the safe minimum standards for the next most preferred ecosystem service are subsequently evaluated. This process continues until all safe minimum standards have been passed for one period, but not the other. When this happens it indicates which period has provided the greater progress towards (strong) sustainability.

Where one or more safe minimum standards are breached, it is possible to determine the degree of degradation of the ecosystem service by applying the concept of the (strong) sustainability gap (Ekins *et al*., 2003). The sustainability gap refers to the proximity-to-threshold or, more precisely, the normalized difference between the present breached score of one or more indicators and their corresponding safe minimum standard. The differences between all indicators that represent the degraded ecosystem service can be aggregated to indicate the overall degree of degradation of that ecosystem service when compared with other degraded ecosystem services.

The assumption, thus far, has been that safe minimum standards can be determined in a straightforward manner. But, there is considerable uncertainty about the delivery of ecosystem services, let alone ecological processes, given the complexity of ecosystems. This uncertainty makes it difficult to determine thresholds precisely, until these thresholds have been already breached with potentially irreversible consequences. And indeed, this inability to set thresholds for the sustainable management of water resources in New Zealand has been recently acknowledged at length (see Land and Water Forum, 2010). Accordingly, the determination of safe minimum standards, where they are not already defined formally, requires the careful judgement of experts (Turner *et al*., 2010). Where a threshold remains difficult to determine with any certainty, it may be

stated by a fuzzy number, so as to depict the uncertainty and imprecision in the safe minimum standard given. Alternatively, where a safe minimum standard remains difficult to determine, regress towards unsustainable states could be indicated simply by undesirable trends in the set of indicators that represent the ecosystem service (Ekins *et al*., 2003).

3.0 The Opihi River Case Study

The Opihi River is located in South Canterbury. Its headwaters 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 to the coast. The Opihi River has three main tributaries that feed into it including the Opuha River, the Tengawai River and the Temuka River. The total area of the Opihi Catchment is approximately 181,000 hectares and 245,000 hectares when the Opuha Catchment is also included (*Figure 2*). Within these catchments a range of land uses take place including grazing, dairy farming and cropping.

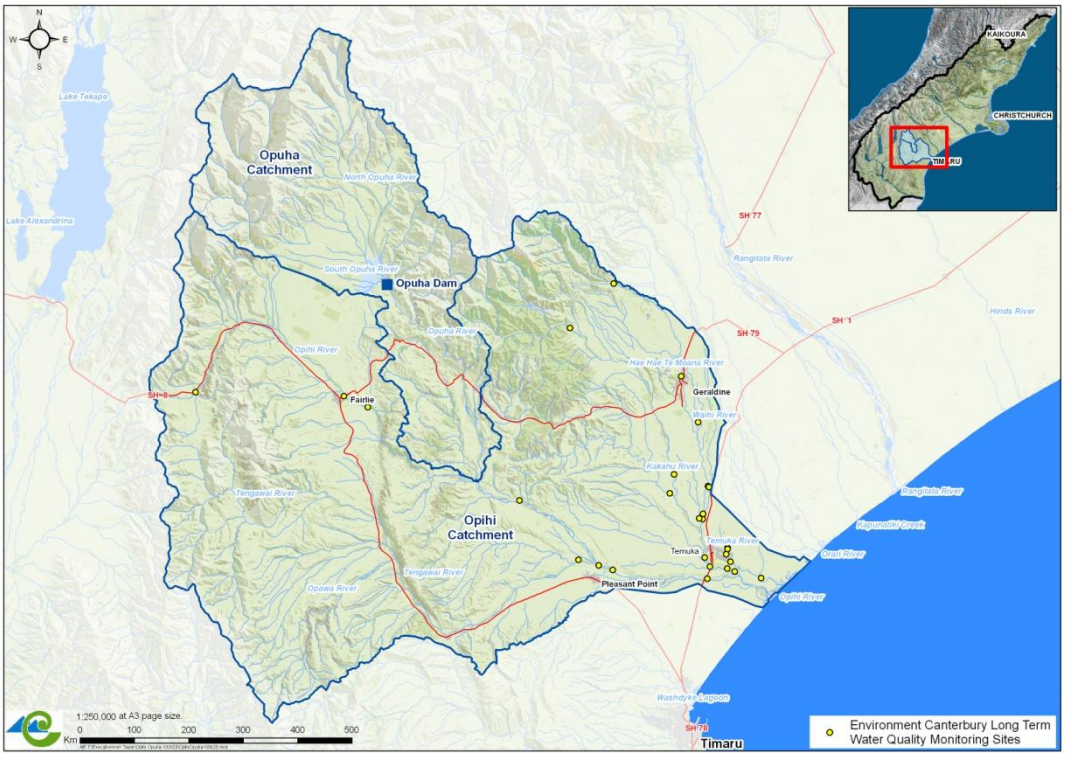

***Figure 2*:** The Opihi Catchment, the Opuha Catchment and the Opuha Dam. Canterbury region in the inset.

The Opihi River provides many ecosystem services. Examples of the set of ecosystem services provided by the Opihi River are given in *Table 1*. This set was determined through the systematic consideration of each ecosystem service classified in the *Millennium Ecosystem Assessment*. All ecosystem services except Biological Products and Climate Regulation were found to be relevant to the Opihi River.

***Table 1*:** The set of ecosystem services provided by the Opihi River.

Class	Ecosystem service	Examples of ecosystem service
Provisioning ecosystem services	Abiotic Products	Gravel extraction for road chip and concrete
	Biological Products	*Not applicable*
	Fibre	Flax, driftwood
	Food	Game fisheries (*e.g.* salmon, trout), native fisheries (*e.g.* eel, whitebait, flounder) and other mahinga kai
	Water Supply	Irrigation, hydroelectric production, municipal water use, industrial water use, stock water use
Regulating ecosystem services	Climate Regulation	*Not applicable*
	Disease Regulation	Parasite and toxic algae regulation
	Erosion Control	Stabilization of river banks
	Natural Hazard Regulation	Flood and drought protection
	Pest Regulation	Invasive non-native species (*e.g.* algae, willows, gorse)
	Water Purification	Removal of pollutants
	Water Regulation	River flow regulation (*e.g.* minimum river flows)
Cultural ecosystem services	Aesthetic Values	Perceived beauty
	Conservation Values	Native biodiversity and habitat, endangered native species (*e.g.* black-billed gull), significant landscapes (*e.g.* Opihi Lagoon)
	Educational Values	Historical/archaeological values & knowledge systems
	Recreational Values	Sailing, rowing, kayaking, fishing, duck hunting, picnicking, swimming, walking
	Spiritual Values	Māori values (*e.g.* mauri)

All rivers in the Opihi Catchment are rain-fed with peak river flows normally occurring during the winter months. The average rainfall during the irrigation season (*i.e.* September to April) is approximately 700 millimetres in the foothills and only 420 millimetres along the coast. This low rainfall coupled with strong winds during the summer months, results in the Opihi Catchment being severely drought-prone, which impacts negatively on agricultural productivity. In an effort to overcome soil water deficiencies, the Levels Plains Irrigation Scheme was established in the catchment in 1936. However, the demand for water in the catchment often exceeded its supply, which resulted in water used for irrigation purposes being over-allocated. The excessive abstraction of water resources often resulted in unacceptable minimum river flows in the Opihi River, which coupled with land use intensification caused the decline in water quality. Furthermore, the low river flows were unable to keep the coastal river mouth open, which prevented fish migration out to sea (Dacker, 1990). The limited fish passage and poor water quality degraded many ecosystem services that were once provided in abundance by the river. In reaction to the degradation of ecosystem services provided, a local initiative emerged that lead to the construction of the Opuha Dam, which was conceived to be a means to augment minimum river flows, enhance water quality and in turn improve the delivery of ecosystem services provided by the Opihi River.

3.1 The Opuha Dam

The Opuha Dam sited on the Opuha River was made fully operational in 1998 and cost NZ$34 million to construct. This was 21 per cent over projected construction costs (Worrall, 2007). Actual operating costs have also been higher than those projected (Harris *et al.*, 2006). The river impoundment artificially created the 700 hectare Lake Opuha upstream of the dam and augmented minimum river flows downstream of the dam to levels considered acceptable to sustain aquatic

health. The increased supply of water, in turn, has increased the capacity to irrigate agricultural land in the Opihi Catchment from 4000 hectares to approximately 16,000 hectares. This increased capacity to irrigate has granted farmers with an estimated NZ$12,000,000 surplus per year when compared to dryland farms, an estimated NZ$123,000,000 per year in spillover benefits to surrounding rural communities and has created an estimated 480 jobs (Harris *et al*., 2006).

Despite positive signs about the performance of the Opuha Dam, which has been officially recognized by way of the Opuha Dam Limited receiving the coveted first prize for Environment Canterbury's (Canterbury Regional Council) biennial resource management awards in 2008, there still remain questions about its cost-effectiveness and the sustainability of the impounded Opihi River (Hearnshaw *et al*., 2010). Indeed, Bryan Jenkins, the chief executive of Environment Canterbury, has stated that "… there are issues that need to be looked at [with dam construction], such as the possible spread of [algae], the mixing of the waters, sustainability and cost" (Worrall, 2007; p. 107). These unanswered questions have arisen largely because less tangible values have been ignored and the impact of land use intensification over time on the aquatic health of the Opihi River remains unclear. Hence, given that a significant period of time has elapsed since the construction of the Opuha Dam, it is appropriate to systematically evaluate its cost-effectiveness and the sustainability of the hydrologically modified reach of the Opihi River.

4.0 Indicator Selection

With an established set of ecosystem services provided by the Opihi River (see *Table 1*), available environmental and socio-economic indicators that have been used to measure particular characteristics of the Opihi River were compiled in an effort to represent each ecosystem service. This initial compilation was surveyed and verified by experts and amended where necessary, which resulted in the removal of some indicators and the inclusion of other previously unidentified indicators. Only those indicators that are applicable to river systems regardless of dam construction were retained in the compilation.

A critical problem in the construction of an ecosystem services index is the issue of double counting, which arises primarily from the complexity of ecosystems and the resulting interdependence of ecosystem services (De Groot *et al*., 2002; Rodriguez *et al*., 2006). In particular, double counting may be evident when both intermediate and final ecosystem services are valued in that final ecosystem services valued would, at least partially, be accounted for in intermediate ecosystem services valued (Turner *et al*., 2010). Double counting is also evident when the indicators used to construct an ecosystem services index are used to represent multiple ecosystem services. For example, the indicator Clarity has been compiled to capture multiple ecosystem services including Aesthetic Values, Recreational Values and Water Purification.

Despite double counting being a critical concern to the legitimate evaluation of ecosystem services, it is rarely considered. In fact, Fisher *et al*. (2009) found that only one of 34 recent studies surveyed on ecosystem services raised this issue. Consequently, in this paper an effort is made to adequately deal with, , the issue of double counting through the alignment of an indicator to only one ecosystem service by a process of indicator selection. By assigning each indicator to only one ecosystem service, the problem of summing ecosystem services rather than applying a more complicated aggregative function (*i.e*. multiplicative or non-linear) is considered to be adequately dealt with, in spite of ecosystem services being interdependent phenomena.

To select the most appropriate fit between indicators and ecosystem services, indicators assigned to multiple ecosystem services were evaluated against various scientific criteria previously considered in the literature for indicator selection. These criteria include measurability, credibility, sensitivity, simplicity, relevance, timeliness, data availability and communicability, which refers to the ability of an indicator to convey relevant information about the state of the ecosystem service (Noss, 1990; Lorenz *et al*., 2001; Keeney & Gregory, 2005; Hak, Moldan & Dahl, 2007). Layke (2009) applied the criteria of data availability and communicability for the selection of indicators to represent the relevant set of ecosystem services. These two criteria were applied here as well. However, the additional criterion of indicator cost was also employed in recognition of the scarce funds available to monitor ecosystems cost-effectively (Garcia, 1996; Cantarello & Newton 2008). Hence the selection of indicators was based on their cost-effectiveness.

The results for the selection of indicators are presented in *Table 3* where the criteria of data availability, communicability and indicator cost were scored on a one-to-nine scale (where one is low and nine is high) by six experts knowledgeable of indicators employed in the monitoring of Canterbury river systems. The average scores for the data availability and communicability criteria were summed and divided by the average cost score, which provided an account of each indicator's cost-effectiveness. Accordingly, where one indicator was initially compiled to represent 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, an indicator was subsequently assigned not to that ecosystem service with the highest cost-effectiveness if that ecosystem service was already well-represented by other indicators and the competing ecosystem service lacked an adequate set of indicators.

The final compilation of indicators that represent each ecosystem service provided by the Opihi River is depicted in *Appendix 1*. While there was an effort to represent each ecosystem service by multiple indicators from both environmental and socio-economic perspectives, it is apparent that many ecosystem services remain inadequately captured. This limited representation of ecosystem services from indicators is especially evident with regards to socio-economic indicators. A similar conclusion was reported by Layke (2009), who investigated the use of indicators to capture changes in the delivery of ecosystem services.

4.1 Preferential Weights

It has been established previously that the construction of an ecosystem services index requires two components: a set of selected indicators for the representation of each ecosystem service, and preferential weights for each ecosystem service. With regards to the latter component, the determination of preferential weights requires a suitable multi-criteria analytical method. One method that can determine preferential weights, and has proven to be useful for the construction of indices, is the analytical hierarchy process (Saaty, 1995). While the capacity of the analytical hierarchy process to aid in the construction of indices is recognized in the literature, this is the first effort where the analytical hierarchy process is used to construct a comprehensive ecosystem services index.

The analytical hierarchy process is a method that decomposes evaluations of preference for values (or other criteria) into a hierarchical network (Saaty, 1995). From the hierarchical network constructed, pairwise comparisons between values as ecosystem services and their classes can be made on a one-to-nine scale, where one represents neutrality or indifference between the pairing and nine represents an overwhelming preference for one value over the other. Each pairwise comparison on this scale captures the cardinal intensity of preference between the ecosystem service pairing examined. Thus, in using pairwise comparisons to indicate preference intensity, the inevitable 'trade-offs' between each ecosystem service pairings can be mapped. The pairwise comparisons of all pairings depict ratios, which can be expressed in a ratio matrix (*Equation 2*). It is in this form that the analytical hierarchy process allows the estimation of preferential weights for the set of ecosystem services. While this ratio matrix is computationally demanding to solve, there are a number of programmes (*e.g. Expert Choice*) dedicated to undertaking such evaluations.

$$A = \begin{array}{c} 1 \\ 2 \\ \vdots \\ n \end{array} \begin{pmatrix} w_1/w_1 & w_1/w_2 & \cdots & w_1/w_n \\ w_2/w_1 & w_2/w_2 & \cdots & w_2/w_n \\ \vdots & \vdots & & \vdots \\ w_n/w_1 & w_n/w_2 & \cdots & w_n/w_n \end{pmatrix} \qquad \textbf{(Equation 2)}$$

Here w is the ratios of pairwise comparisons between ecosystem services; and
A is the determination of preferences from the ratio matrix.

In *Figure 3* the constructed hierarchical network is depicted. At its pinnacle is the ecosystem services index. The next level contains the classes of ecosystem services and the lower level the set of ecosystem services. Extensive use of even lower levels could have been developed that attempted to decompose each ecosystem service into further component parts. However, lower levels were not employed here, except for the ecosystem service Water Supply, which was decomposed further in order to decipher preferential weights for Irrigation over Other Water Supply Uses (*e.g.* hydroelectric production, municipal water supply).

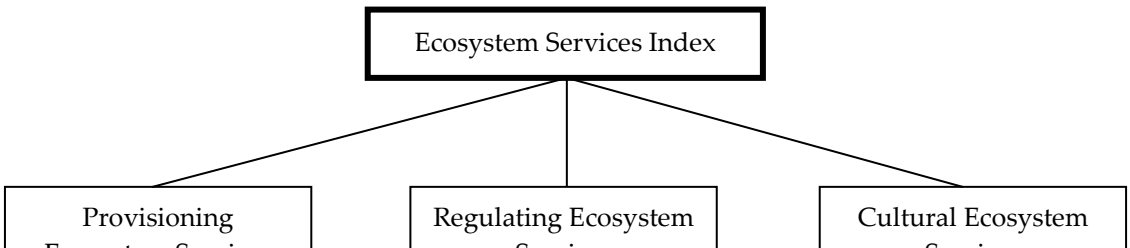

***Figure 3*:** The hierarchical network for constructing preferences of ecosystem services.

In an effort to limit the impracticalities found when applying non-market valuation methods, the determination of preferential weights is considered to be best carried out by a small number of stakeholder representatives. The use of stakeholder representatives as 'overseers' of a stakeholder group rather than a large stakeholder sample is less costly, conducive with the method of cost utility analysis and has been shown to provide a reasonable approximation of preferences obtained from larger sampling frames (Colombo *et al*., 2009). In fact, the surveying of stakeholders may lead to less than satisfactory results as stakeholders may neither possess sufficient understanding of the ecosystem services provided, nor an adequate grasp of the evaluation methods used (Alvarez-Farizo & Hanley, 2006; Barkmann *et al*., 2008). However, it is recognized that the use of stakeholder representatives makes absolute claims of representation difficult (Spash, 2007). Hence, the selection of stakeholder representatives is critical and must attempt to represent a reasonably equitable and proportional microcosm of all affected stakeholder groups (Turner *et al*., 2010). Moreover, the selection of stakeholder representatives needs to account for the fact that preferences of ecosystem services may vary at local and regional scales of analysis. Hence, stakeholder representatives selected for the determination of preferential weights of ecosystem services included those with a local appreciation of the impounded river system evaluated, as well as those with a broader perspective that we able to consider preferences for ecosystem services provided by river in a regional context (*e.g*. Canterbury).

Preferences of ecosystem services were collected from 15 regional stakeholder representatives and six local stakeholder representatives, who, in each case, represent (or have represented) in a formal capacity various important stakeholder groups that are pertinent to water resource management in

the Canterbury region or the Opihi Catchment in particular. The difficult task of selecting stakeholder representatives was made easier as many regional stakeholder representatives used were already a part of a formal regional group established specifically to address water resource issues. While no formalized local group existed, such a group was in the process of development. Accordingly, additional stakeholder representatives were chosen that represented many of the stakeholder groups that have been of historical importance both regionally and locally.

All stakeholder representatives were informed to provide their preferences from the perspective of the present needs of the stakeholders that they represent. The preferences obtained were analysed in the computational programme *Expert Choice*, which provided each stakeholder representative a set of preferential weights for each ecosystem service. In addition, an inconsistency measure that gauged the degree of intransitivity between preferences throughout all ecosystem service pairings was also calculated. This inconsistency measure is significant as it recognizes the presence of bounded rationality in preference formation in that people do not always possess consistent or transitive preferences as assumed in normative theories of rational choice. The initial average inconsistency for stakeholder representatives was 13 per cent. The recommended level of inconsistency is about ten per cent. Saaty (1995) recommends that where preferences are higher than ten per cent then they should be revised where possible. Hence, some stakeholder representatives were asked if they wish to revise highly inconsistent preferences according to various computationally-devised remedies. Those stakeholder representatives that choose to revise their preferences resulted in a reduction in the average inconsistency to a satisfactory level of 11 per cent.

In *Figure 4* the revised average preferential weights for ecosystem services from 15 regional stakeholder representatives and six local stakeholder representatives are shown. The ecosystem service Water Supply is the most preferred for both regional and local stakeholder representatives. This preference supports previous research for ecosystem services provided by rivers within Canterbury (MacDonald & Patterson, 2008), and might be expected given the increased demand for abstracting water in the region. Of interest the average ratio of Irrigation to Other Water Supply Uses was significantly less than one for both local (0.48) and regional (0.70) stakeholder representatives. While the ecosystem service Water Supply was given the greatest preference, other ecosystem services were found to have relatively high preferential weights. These findings reiterate that water resources are valued for many reasons. The preferences held for ecosystem services other than Water Supply differ considerably, however, between local and regional stakeholder representatives. Regional stakeholder representatives placed greater emphasis on regulating ecosystem services. Conversely, local stakeholder representatives placed greater emphasis on provisioning and cultural ecosystem services.

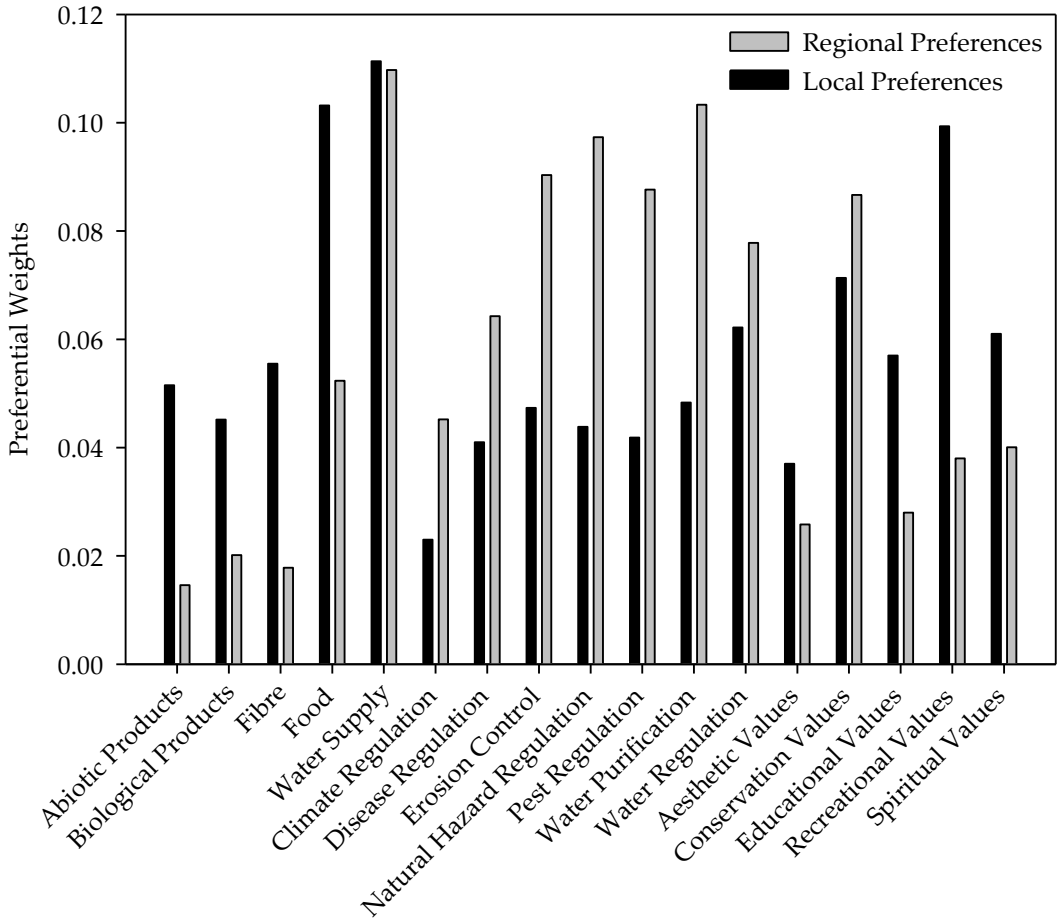

***Figure 4*:** Preferential weights for ecosystem services from regional (*i.e*. Canterbury river systems) and local (*i.e*. Opihi River) stakeholder representatives. Ecosystem services from left to right progress from Provisioning services through Regulating to Cultural.

5.0 The Sustainability of the Opihi River

The evaluation of the sustainability of the Opihi River both pre-dam and post-dam was analysed through weak and strong criteria. The evaluation period was between the years 1989 to 2008, which provided a timeframe with a number of years both before and after the Opuha Dam construction in 1997. Indicators were averaged if they were collected from multiple periods throughout the year and from multiple monitoring sites along the Opihi River. Ideally, the evaluation period would have been extended before the Opihi River was first hydrologically modified by the Levels Plain Irrigation Scheme. However, establishing such a 'natural' baseline was not possible as many indicators had little or no data available before 1989. Conclusions drawn from this paper should therefore acknowledge the state of the river under study pre-dam before making assumptions on similar findings being applicable to the impacts of dams on unmodified or pristine river systems.

The strong sustainability criterion was operationalized by amassing safe minimum standards for each indicator by collated 'formal' pronouncements, such as those indicated in the Opihi River Regional Plan (Environment Canterbury, 2000) and the proposed Canterbury Natural Resources Regional Plan (Environment Canterbury, 2010). Where safe minimum standards were not pronounced, they were elicited by expert judgement. In total three experts were used to elicit safe minimum standards. Where safe minimum standards elicited from experts differed then they were averaged. The final thresholds for the set of indicators, as safe minimum standards, are detailed in *Appendix 3*. Of note, not all indicators (*e.g.* Volume of Gravel Extracted) were given safe minimum standards. This was because either the elicitation of safe minimum standards was not appropriate for these indicators or no such threshold could yet be confidently given. In these cases, a 'no undesirable trend' in the indicator was used for its evaluation.

The evaluation of strong sustainability in its most complete form would be observed where the safe minimum standards for all indicators have not been breached. Strong sustainability in its most complete form was not indicated as safe minimum standards were breached for many indicators both pre-dam and post-dam on the Opihi River. Accordingly, a checklist approach was first applied to evaluate strong sustainability both pre-dam and post-dam by the determination of the number of ecosystem services that failed their respective safe minimum standards. Failure was indicated by any indicator that represented an ecosystem service breaching its safe minimum standard in any of the years that were evaluated. It was determined that nine ecosystem services failed their safe minimum standards pre-dam, while eight ecosystem services failed their safe minimum standards post-dam. This indicates marginal progress towards (strong) sustainability on the Opihi River since the construction of the Opuha Dam. However, as previously indicated, this checklist approach is a simple, somewhat unsatisfactory, method of evaluation.

A more refined checklist approach was also applied that considered the percentage of years safe minimum standards failed for the set of indicators used to represent each ecosystem service. *Table 4* indicates the percentage of years failed for each ecosystem service provided both pre-dam and post-dam by the Opihi River and for the 'Opihi River - Upstream', which is the part of the Opihi River between its headwaters and the confluence where the Opihi River meets the Opuha River. Consequently, this upstream part is not directly impacted by the Opuha Dam and thus acts as a 'control'. However, the lack of available data for many indicators along the upstream part of the Opihi River limited the number of ecosystem services that could be evaluated adequately.

***Table 4*:** Percentage of years failed for ecosystem services provided by the Opihi River and the Opihi River - Upstream both pre-dam and post-dam.

	Percentage of Years Failed			
	Opihi River		Opihi River - Upstream	
Ecosystem Service	Pre-dam	Post-dam	Pre-dam	Post-dam
Abiotic Products	0	0	---	---
Fibre	0	0	---	---
Food	3.3	20.5	0	25
Water Supply	37.5	0	---	---
Disease Regulation	0	18.2	12.5	9.1
Erosion Control	50	36.4	62.5	27.3
Natural Hazard Regulation	11.1	0	---	---
Pest Regulation	0	0	---	---
Water Purification	27.3	26.7	50.8	35.4
Water Regulation	69.6	18.2	---	---
Aesthetic Values	50	54.5	62.5	59.1
Conservation Values	3.3	2.6	---	---
Educational Values	0	0	---	---
Recreational Values	41.7	33.3	---	---
Spiritual Values	0	0	---	---
Total Percentage of Years Failed	19.6	14.0	*	*

*Not calculated due to insufficient data

Table 4 indicates the percentage of years failed decreases on the Opihi River post-dam when compared with the Opihi River pre-dam. Of the set of ecosystem services provided, the ecosystem services Water Supply and Water Regulation improved markedly post-dam. These findings provide further evidence that since the construction of the Opuha Dam the ecosystem services provided by the Opihi River have progressed towards strong sustainability. However, three cautionary observations are noted. First, the percentage of years failed increases post-dam for the ecosystem services Food, Disease Regulation and Aesthetic Values. Secondly, while not revealed in *Table 4*, the percentage of years failed rose considerably from 25 per cent pre-dam to 70 per cent and 90 per cent post-dam for the indicators Number of Anglers and Number of Salmonids Caught, respectively. Finally, the indicator Nitrate Concentration failed 45 per cent of years post-dam, yet did not fail at all pre-dam. This noticeable change in the percentage of years failed for this indicator suggests, despite a diluting effect from augmented minimum river flows, that nitrates have increasingly entered the river presumably from intensified land use in the Opihi Catchment.

Previously, the application of the characteristic filtering rule was proposed for evaluating the strong sustainability criterion. This method was applied for evaluating the Opihi River pre-dam and post-dam according to the hierarchical ranking of ecosystem services determined by the preferential weights determined in *Figure 4*. In applying the characteristic filtering rule, the most preferred ecosystem service Water Supply was evaluated. In evaluating Water Supply, it was observed that the indicators that represent this ecosystem service passed all safe minimum standards post-dam, but breached some safe minimum standards pre-dam (see *Appendix 4a*). This finding further indicates that the Opihi River post-dam has progressed towards strong sustainability.

Further analysis using the characteristic filtering rule was developed to indicate the strong sustainability pre-dam and post-dam for the various classes of ecosystem services provided. This

level of analysis is significant, as it allows for 'triple bottom-line' type of evaluations often applied to environmental, social and economic dimensions of sustainability (Hacking & Guthrie, 2008). Hence, similar to a triple bottom-line evaluation, but in line with the ecosystem services approach undertaken in this paper, each class of ecosystem services were evaluated independently. Provisioning ecosystem services were not evaluated as this analysis is inferred from the evaluation of the ecosystem service Water Supply performed previously. With regards to regional preferences of regulating ecosystem services the ecosystem service Water Purification was the most preferred. The evaluation of this ecosystem service indicated that safe minimum standards were breached both pre-dam and post-dam. Hence, given that both periods failed, the next most preferred ecosystem service Natural Hazard Regulation was evaluated. For this ecosystem service it was found that all safe minimum standards passed post-dam, yet one safe minimum standard failed pre-dam (see *Appendix 4b*). With regards to local preferences, the most preferred regulating ecosystem service was Natural Hazard Regulation. Hence, the same finding for the analysis of regional preferences is observed with local preferences. The analysis of cultural ecosystem services applying the characteristic filtering rule was inconclusive. This was because all ecosystem services both pre-dam and post-dam had some failures in safe minimum standards.

The sustainability gap, which is applied to ecosystem services that have breached their safe minimum standard, is a measure of the degree of degradation of each ecosystem service. The sustainability gap was established in three steps. First, the quantitative output of each indicator is normalized on a 0-to-100 scale. For some indicators (*e.g.* Irrigated Area), zero and 100 represent the historical minimum point and the historical maximum point, respectively, where the historical maximum point is the optimal state. However, not all indicators follow this positive progression. Other indicators (*e.g.* Total Nitrogen Concentration) follow a negative progression where the optimal state is found at the historical minimum point. While still other indicators (*e.g.* pH Levels and Water Temperature) have an optimal state, not at the extremes of their historical quantitative output, but between their minimum and maximum thresholds given as safe minimum standards. The second step establishes the present state and safe minimum standard of the indicator on the 0-to-100 scale, which allows the determination of the sustainability gap by calculating the difference between the present normalized score and the normalized safe minimum standard. The final step calculates the sustainability gap for each degraded ecosystem service by aggregating the normalized degree of degradation from all indicators that represent that ecosystem service.

Figure 5 indicates the sustainability gap or degree of degradation for the various ecosystem services that breached their safe minimum standards both pre-dam and post-dam. It is shown that the sustainability gap of degraded ecosystem services were more degraded pre-dam than post-dam. In fact, for some ecosystem services there has been a significant decrease in their degree of degradation post-dam. For example, the degree of degradation for the ecosystem service Water Regulation has more than halved since the construction of the Opuha Dam.

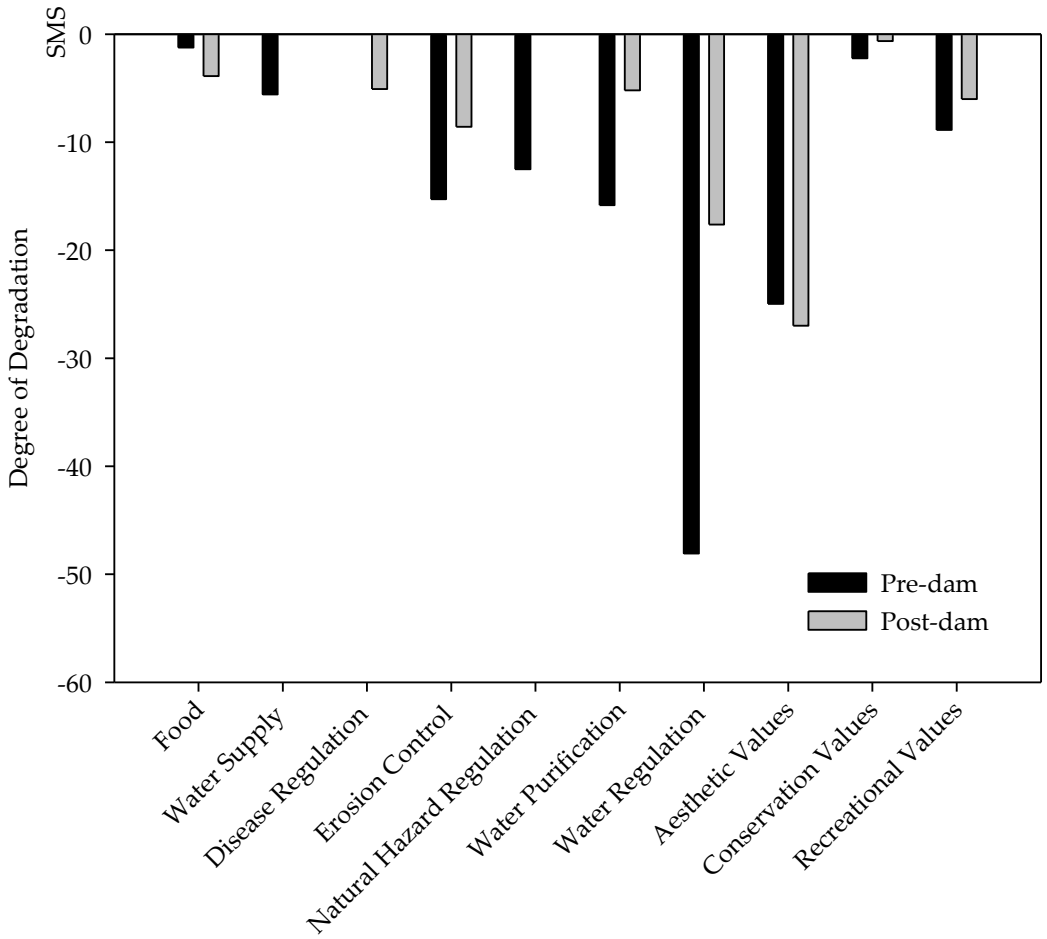

***Figure 5*:** The sustainability gap or degree of degradation of ecosystem services that breached their safe minimum standard (SMS) (*i.e.* 0 degree of degradation).

5.1 Weak Sustainability and the ESI

Weak sustainability of the impounded Opihi River was evaluated using the ecosystem services index, which has the form indicated previously in *Equation 1*. However, the evaluation of weak sustainability was analysed by dividing the ecosystem services index by the number of indicators used to construct the index in that year. This ensured that missing data for indicators would not impact the inferences obtained from the ecosystem services index. In addition, while preferential weights for ecosystem services were determined, it was assumed that indicators that capture each ecosystem service were of equal weight. However, indicator weightings could have been potentially given for the construction of the ecosystem services index according to their cost-effectiveness.

Sensitivity analysis was performed prior to the application of the ecosystem services index. For the sensitivity analysis, each ecosystem service was removed from the index to determine the percentage change and dominance that ecosystem service had on the constructed ecosystem service index. An indication of the dominance of any one ecosystem service in an index is important, as a well-constructed index should not be dominated by the quantitative output of only a single or a few of its indicators (Saisana & Saltelli, 2008). From the sensitivity analysis it was observed that the ecosystem service Water Supply had the greatest impact on the ecosystem service index, though the

percentage change after its removal was less than 20 per cent. This percentage change was considered sufficiently small to conclude that the ecosystem services index constructed was multi-dimensional. However, closer inspection of the indicators of each ecosystem service revealed that the indicators Irrigated Area and Economic Impact from Irrigation that partially represent the ecosystem service Water Supply were completely positively correlated. Given this correlation the environmental indicator Irrigated Area was removed from the ecosystem services index.

Figure 6 depicts the average ecosystem services index per indicator over the evaluation period. It is shown that whether the ecosystem services index is constructed using local or regional preferences that the aggregated set of ecosystem services provided by the Opihi River increases over the evaluation period. This positive long-term trend, therefore, indicates evidence of the weak sustainability of the impounded Opihi River. In fact, all index points for local preferences are greater post-dam than pre-dam indicating a 'step' change in the aggregated set of ecosystem services provided by the Opihi River since dam construction.

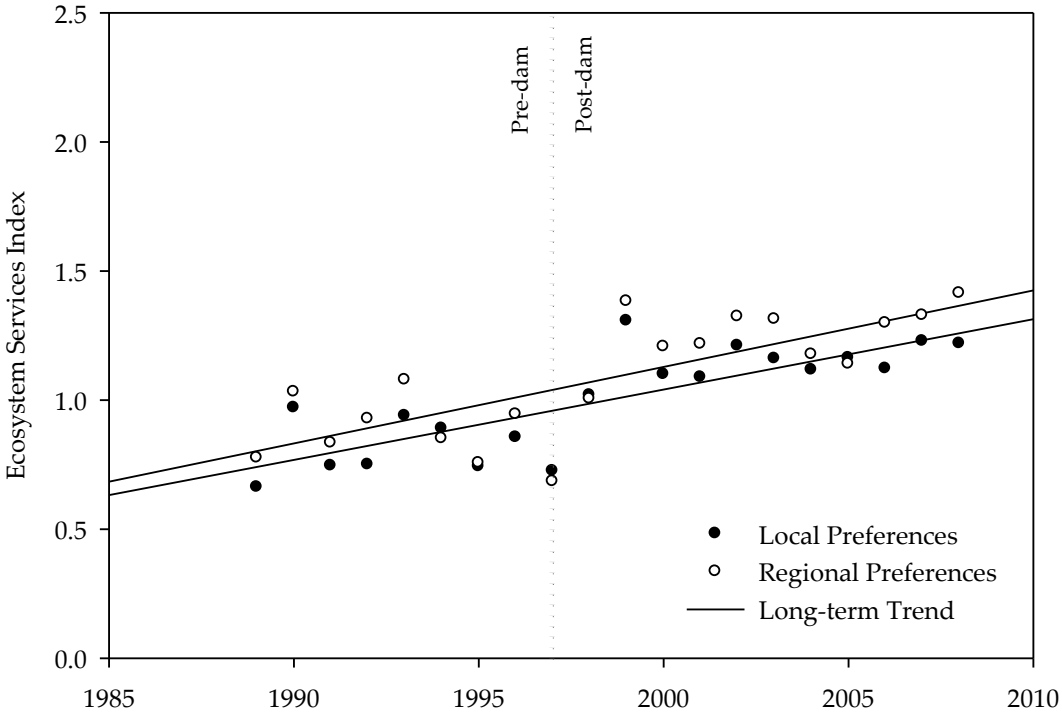

***Figure 6*:** Average ecosystem services index per indicator for the ecosystem services provided by the Opihi River.

In keeping with the ecosystem services approach to triple-bottom-line evaluations, each class of ecosystem service was evaluated independently by aggregating only those ecosystem services relevant to that class into an ecosystem services index. *Figures 7a-c* depict the average ecosystem services index per indicator for provisioning ecosystem services, regulating ecosystem services and cultural ecosystem services, respectively. In each case, it is demonstrated that the aggregated class set of ecosystem services increases over the evaluation period. This, therefore, further indicates evidence of the weak sustainability of the impounded Opihi River. However, while all classes of ecosystem services were increasing, the local preferences for regulating ecosystem services were

only marginally increasing. This suggests that this class of ecosystem service is vulnerable to a transition towards an unsustainable state indicated by a declining ecosystem services index.

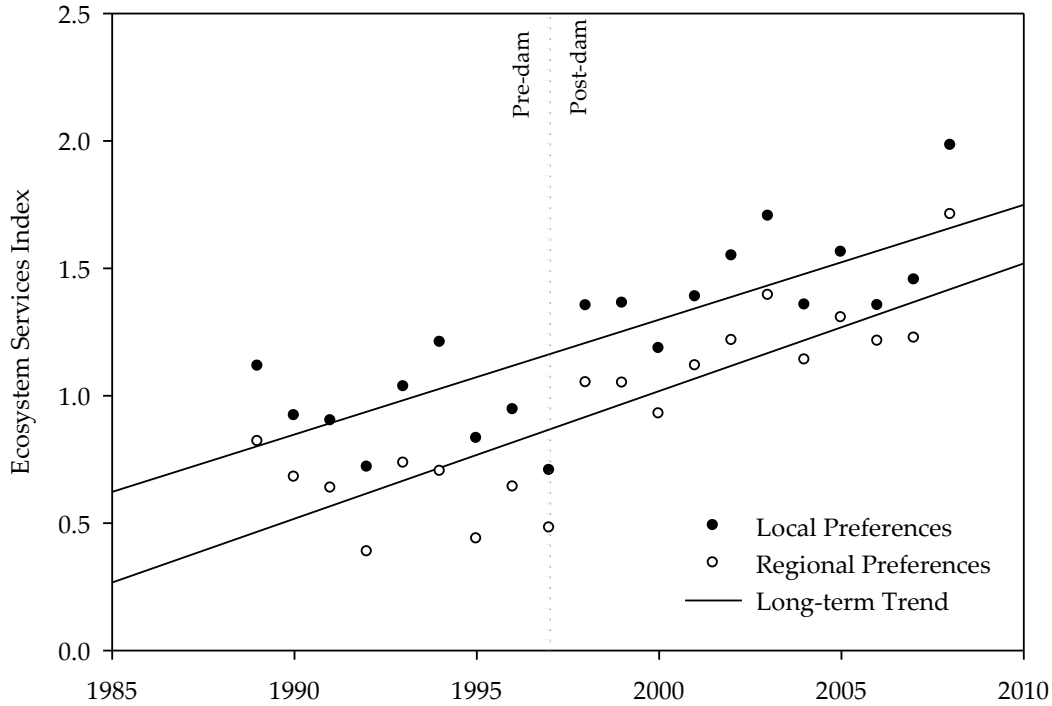

***Figure 7a*:** Average ecosystem services index per indicator for the provisioning ecosystem services provided by the Opihi River.

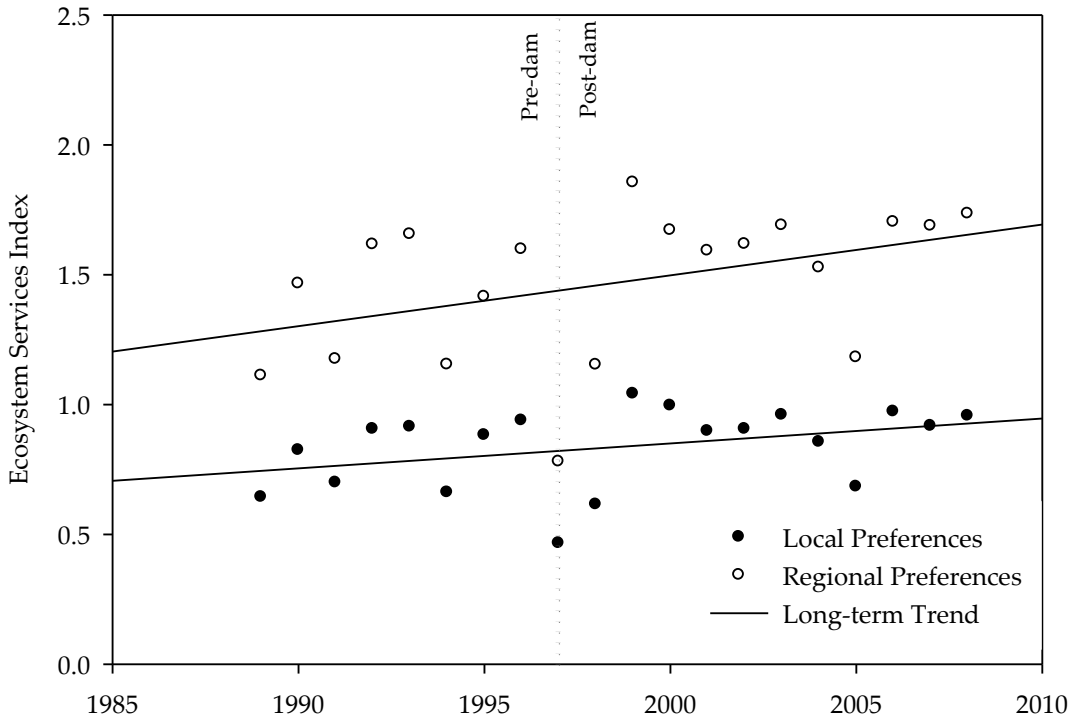

***Figure 7b*:** Average ecosystem services index per indicator for the regulating ecosystem services provided by the Opihi River.

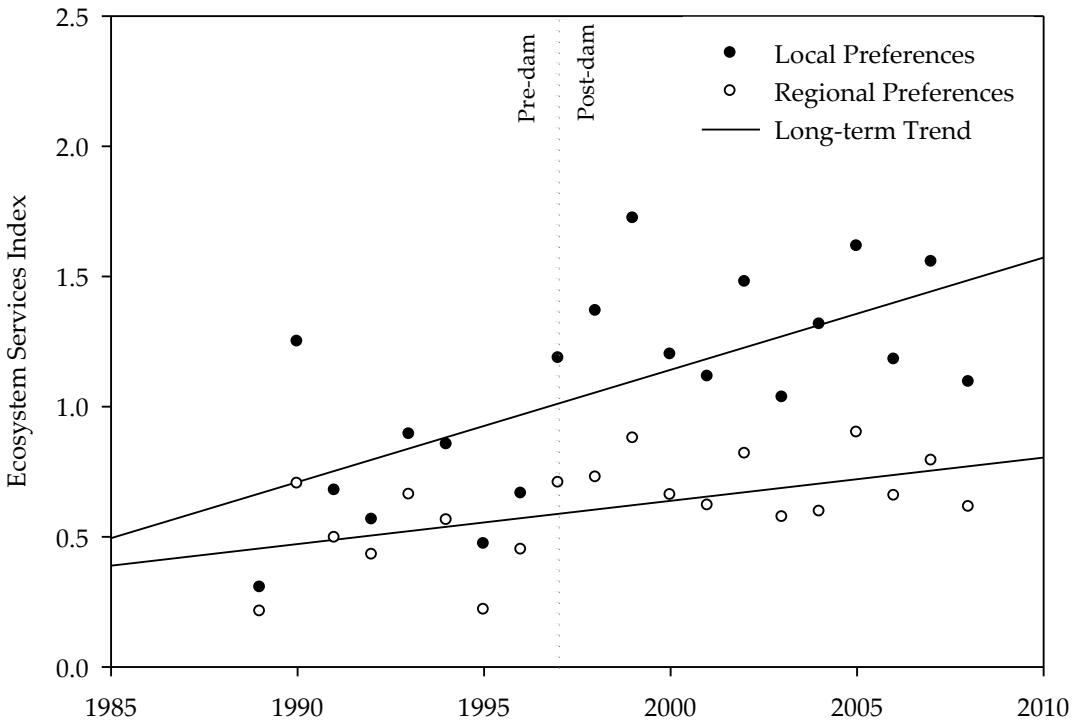

***Figure 7c*:** Average ecosystem services index per indicator for the cultural ecosystem services provided by the Opihi River.

From the series of analysis undertaken it is contended that the impounded Opihi River has progressed towards weak and strong sustainability since the construction of the Opuha Dam. This conclusion supports the belief reported in interviews with many experts and stakeholder representatives that since dam construction the overall aquatic health of the Opihi River has improved. Three cautionary observations, however, are noted. First, the various evaluation methods applied, while able to evaluate the sustainability of Opihi River, are not able to infer the causal mechanism that has led to this progression towards sustainability. While it may be hypothesized that the Opuha Dam is the underlying causal mechanism for this progress, this reasonable hypothesis has not been scientifically validated. Secondly, the evaluation of the ecosystem services Fibre, Pest Regulation and Spiritual Values were inadequately captured, as each of these ecosystem services were only represented by a single indicator that did not vary throughout the period of evaluation. Finally, as stated previously, the evaluation period adopted in this paper may be too short a timeframe to infer with confidence the sustainable state of the Opihi River. Moreover, there may be lag effects from the impact of land use intensification on the aquatic health of the river that are yet to be revealed, which highlights the ongoing need to continue to evaluate the river in the future. With regards to future evaluations, it may be necessary to update preferential weights from stakeholder representatives and normalized scores from indicators as new historical quantitative output extends beyond the extremities of the data previously analysed. Where this is so, then all previous index output should also be readjusted accordingly so that previous evaluations are measured against current weighting and scoring parameters.

5.2 The Cost-Effectiveness of the Opuha Dam

The evaluation of the cost-effectiveness of the Opuha Dam was performed by cost utility analysis. Annual operational cost data relating to the Opuha Dam (*i.e.* dam running costs, dam maintenance costs) and the Opihi River (*i.e.* river management costs) were aggregated for each year of the evaluation period. River management costs included only the management cost of opening the river mouth of the Opihi River to the sea by bulldozer. Additional river management costs that are observed on the Opihi River, but were not practically available, include pest management costs, riparian management costs and flood control management costs. Monitoring costs were ignored as these costs do not produce or modify the delivery of ecosystem services provided by the river system. Aggregated costs coupled with the ecosystem services index determined in each year provided the basis for the calculation of cost-utility ratios per year. *Figure 8* shows these cost-utility ratios for the Opihi River over the evaluation period.

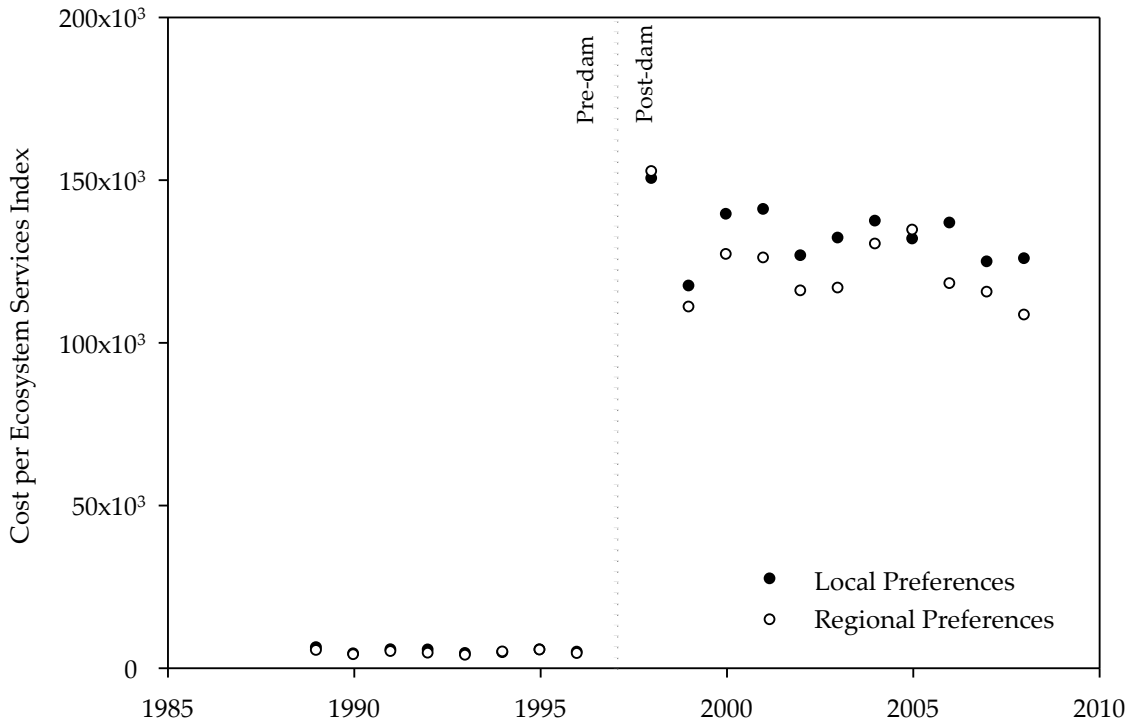

***Figure 8*:** Cost per ecosystem service index for the Opihi River.

From *Figure 8* it is evident that the cost per ecosystem service index is many orders of magnitude less pre-dam than post-dam for the Opihi River. The lower cost per ecosystem services index pre-dam might be expected given that ecosystem services provided by 'natural' river systems are produced by nature for free, yet obviously river systems that are hydrologically modified by way of dams or other artificial means are not.

Despite the poor cost-effectiveness of the Opuha Dam, it is noteworthy that since dam construction progress towards sustainability has been indicated for the Opihi River. Hence, without the construction of the Opuha Dam it is plausible that many degraded ecosystem services once provided by nature for free would be lost, which if retrievable in the future would presumably be at an even greater cost than that evidenced with the construction of the Opuha Dam. Moreover, despite the poor cost-effectiveness of the Opuha Dam, it may be a cost-effective investment relative

to other dams also constructed for the purposes of water storage and irrigation in the Canterbury region (*e.g.* Highbank Dam). Such analysis might reveal reasons as to why one dam is more cost-effective than another.

6.0 Conclusion

In this paper various evaluation methods were applied in accordance with the ecosystem services approach in order to evaluate the cost-effectiveness of the Opuha Dam and the sustainability of the impounded Opihi River. From the analysis it was observed that since the construction of the Opuha Dam the Opihi River has progressed towards weak and strong sustainability. However, the cost-effectiveness of the Opuha Dam was poor relative to that established pre-dam.

In order to determine with greater precision the cost-effectiveness of dam projects and the sustainability of impounded river systems using the various evaluation methods applied that incorporate indicators, there is a need to establish a more comprehensive and potentially standardized set of indicators. Improving the use of indicators to better represent ecosystem services provided by rivers requires the development of: one, more indicators, especially socio-economic indicators, for many ecosystem services; two, cost-effective monitoring practices that bring about indicators with long and uninterrupted data series collected at multiple spatio-temporal scales; and three, scientifically defensible indicators that track the delivery of the ecosystem service more closely, rather than acting only as a 'proxy' that is poorly correlated with its associated ecosystem service. One can foresee that progress in developing indicators that more closely correlate with the ecosystem service that they are intended to represent will occur as the ecosystem services approach adopted in this paper becomes more widely integrated in water resource management. Alternatively, rather than developing scientifically defensible indicators that more closely correlate with ecosystem services, it might be beneficial to decompose ecosystem services further so as to diminish the abstraction of many ecosystem services to more tangible phenomena. For example, the ecosystem service Recreational Values could be decomposed into each recreational activity (*e.g.* swimming) that makes up this value, which in turn allows indicators (*e.g.* Number of Swimmers in River) to better correlate with this ecosystem service.

While further work remains to refine the ecosystem service approach demonstrated here, namely to develop more effective indicators of river ecosystem services, the work does introduce a novel method to evaluate the impacts of dams on river systems which may significantly aid future river management decisions.

6.0 Acknowledgements

The authors gratefully acknowledge the financial support of Environment Canterbury. The authors also acknowledge the time of experts and stakeholder representatives that were interviewed for this paper, who represent the following organizations: the Canterbury Chamber of Commerce, Canterbury Water Management Steering Group, Christchurch City Council, Department of Conservation, Environment Canterbury, Fish and Game, Irrigation NZ, Lincoln University, Ministry of Agriculture and Forestry, National Institute of Water and Atmospheric Research, Ngāi Tahu, Opuha Water Limited, Royal Forest and Bird Protection Society, Timaru District Council and the Water Rights Trust.

7.0 Appendices

***Appendix 1a*:** Indicators compiled to represent provisioning ecosystem services provided by the Opihi River.

Class	Ecosystem service	Environmental indicators	Socio-economic indicators
Provisioning ecosystem services	Abiotic Products	Mean River Bed Level (m)	Profitability of Gravel Resource ($)
		Volume of Gravel Extracted (m^3)	
	Fibre	Number of Fibrous Species	Number of People Actively Collecting Fibrous Materials
		Total Biomass of Fibrous Species (kg)	
	Food	Annual Periphyton Cover (%)	Commercial Fishery Employment
		Average Weight of Fish Caught (kg)	Cultural Health Index
		Benthic Community Metabolism (R^2)	Fish Taste
		Biochemical Oxygen Demand (mg/l)	Number of People Actively Collecting Food
		Days River Mouth Closed	
		Dissolved Oxygen Level (ml/l)	
		Number of Mahinga Kai Species	
		Number of Salmonids Caught	
		pH Level	
		Presence of Riparian Vegetation	
		Spawning Numbers	
		Turbidity (NTU)	
		Water Temperature (^0C)	
	Water Supply	Irrigated Area (ha)	Economic Impact from Irrigation ($)
		River Flow Variability (σ^2)	
		Total Volume of Water Takes (m^3)	

Note: (1) Lightly shaded sections reflect known available indicators. (2) Darkly shaded sections reflect indicators selected to represent each ecosystem service.

***Appendix 1b*:** Indicators compiled to represent regulating ecosystem services provided by the Opihi River.

Class	Ecosystem service	Environmental indicators	Socio-economic indicators
Regulating ecosystem services	Disease Regulation	Annual Periphyton Cover (%)	
		Number of Fish Kills	
	Erosion Control	Area of Riparian Vegetation (km^2)	
		Turbidity (NTU)	
	Natural Hazard Regulation	Number of Flood Flows	Number of Flood Event Fatalities
		Irrigated Area (ha)	Total Cost of Flood Event ($)
	Pest Regulation	Area Covered by Invasive Species (km^2)	
		Number of Pest Species	
	Water Purification	Annual Periphyton Cover (%)	Total Cost of Water Treatment ($)
		Benthic Community Metabolism (R^2)	
		Clarity (m)	
		Conductivity (mS/m)	
		Cryptosporidium Levels	
		Dissolved Organic Carbon (mg/l)	
		Dissolved Reactive Phosphorus (mg/l)	
		E.coli Levels (unit/100ml)	
		Ephemeroptera, Plecoptera and *Trichoptera* Taxa Percentage (%)	
		Faecal Coliforms (unit/100ml)	
		Qualitative Macroinvertebrate Community Index	
		Macrophyte Maximum Cover (%)	
		Nitrate Concentration (mg/l)	
		pH Level	
		Area of Riparian Vegetation (km^2)	
		Total Nitrogen Concentration (mg/l)	
		Total Phosphorus Concentration (mg/l)	
		Toxicant Level (ug/l)	
	Water Regulation	Days River Mouth Closed	
		Minimum River Flows (m^3/s)	
		Number of Flood Flows	
		Number of Flushing Flows	

***Appendix 1c*:** Indicators compiled to represent cultural ecosystem services provided by the Opihi River.

Class	Ecosystem service	Environmental indicators	Socio-economic indicators
Cultural ecosystem services	Aesthetic Values	Algal Mat Colour (0-100 scale*)	Willingness to Pay for Riverside Property ($)
		Annual Periphyton Cover (%)	
		Clarity (m)	
		Macrophyte Maximum Cover (%)	
		Area of Riparian Vegetation (km^2)	
		Turbidity (NTU)	
	Conservation Values	Area Covered by Invasive Species (km^2)	
		Area Covered of Native Vegetation (km^2)	
		Native Biodiversity (Shannon Diversity Index)	
		Number of Threatened Native Bird Species	
		Number of Native Fish Species Present	
		Number of Significant Landscapes	
		Number of Braided River Islands	
		Qualitative Macroinvertebrate Community Index	
		Width of Braided River Channels (m)	
	Educational Values		Number of Studies about River
			Number of School Visits
			Number of Signs with Educational Content
	Recreational Values	Annual Periphyton Cover (%)	Number of Swimmers in River
		Clarity (m)	Number of Anglers
		E. coli Levels (units/100ml)	Number of Recreational Facilities Constructed
		Minimum River Flows (m^3/s)	Number of Duck Hunters
		Number of Salmonids Caught	Number of White Water Kayakers
		Turbidity (NTU)	Willingness to Pay for Recreational Activities
	Spiritual Values	Native Biodiversity (Shannon Diversity Index)	Cultural Health Index
		Number of Mahinga Kai Species	

*Algal Mat Colour was represented on a 0-100 numeric scale where 0 represents Black, 50 represents Brown, 100 represents Green.

***Appendix 2*:** The selection of indicators for representing each ecosystem service was deciphered by their cost-effectiveness. Note that shaded sections indicate preferred ecosystem services to be represented by the indicator. Also note that * indicates where an indicator was allocated to an ecosystem service for which it is not the most cost-effective. This was done due to the paucity of indicators for representing that ecosystem service.

Indicator	Ecosystem service	Communicability (1-9 scale)	Data availability (1-9 scale)	Annual cost (1-9 scale)	Indicator cost-effectiveness
Annual Periphyton Cover	Aesthetic Values	7	5.67	3	4.22
	Disease Regulation*	5			3.56
	Food	4			3.22
	Recreational Values	8			4.56
	Water Purification	4.3			3.32
Clarity	Aesthetic Values	7	6.33	2.3	5.80
	Recreational Values	7			5.80
	Water Purification	1			3.19
Days River Mouth Closed	Food	7	6.33	3	4.44
	Water Regulation	7			4.44
Dissolved Oxygen Level	Food	7	7	3	4.67
	Water Purification	7			4.67
E. coli Level	Recreational Values	6.3	7.67	5	2.79
	Water Purification	7			2.93
Irrigated Area	Water Supply	8	9	2	8.5
	Natural Hazard Regulation	3			6
Minimum River Flows	Water Regulation	5	7	3	4
	Recreational Values	5			4
Number of Flood Flows	Natural Hazard Regulation	9	8.33	4.33	4
	Water Regulation	6.33			3.39
Number of Mahinga Kai Species	Food	9	5	3	4.67
	Spiritual Values*	5			3.33
Number of Salmonids Caught	Food	9	7	5	3.2
	Recreational Values	9			3.2
Qualitative Macroinvertebrate Community Index	Conservation Values	7	7	6.33	2.21
	Water Purification	6.33			2.11
pH Level	Water Purification	7	7	3	4.67
	Food	5			4
Turbidity	Erosion Control*	4	7	4.33	2.31
	Food	2			1.85
	Aesthetic Values	5			2.54
	Recreational Values	4			2.31

***Appendix 3*:** Safe minimum standards for indicators sourced from formal pronouncement or expert judgement. SMS were set at 1996 levels (pre-dam).

Ecosystem Service	Indicator	Safe Minimum Standard	
		Threshold	Source
Abiotic Products	Mean River Bed Level	40.93m	Boyle & Surman, 2007
Fibre	Number of Fibrous Species	No decline	Expert
Food	Biochemical Oxygen Demand	Maximum 1mg/l	Expert
	Dissolved Oxygen Levels	Daily minimum 8ml/l	Expert
	Number of Salmonids Caught	500 caught	Expert
	Spawning Numbers	No undesirable trend	Expert
	Water Temperature	Daily minimum 4C & maximum 20C	ECan, 2010; Expert
Water Supply	Economic Impact from Irrigation	No decline	CMF, 2010
	River Flow Variability	No increase	CMF, 2010
	Total Volume of Water Takes	No undesirable trend	CMF, 2010
Disease Regulation	Annual Periphyton Cover	Maximum 30%	Expert
Erosion Control	Turbidity	Maximum 2	Expert
Natural Hazard Regulation	Number of Flood Event Fatalities	No fatalities	Expert
	Number of Flood Flows	No floods	Expert
Pest Regulation	Number of Pest Species	No increase	Expert
Water Purification	Conductivity	No decline	Expert
	Dissolved Reactive Phosphorus	Maximum 0.006mg/l	ECan, 2010
	E.coli Level	Maximum 550 *E. coli* units/100ml	ECan, 2010
	Ephemeroptera, *Plecoptera* and *Trichoptera* Taxa Percentage	Minimum 50%	Expert
	Faecal Coliform Level	Maximum 550 coliform units/100ml	Expert
	Nitrate Concentration	Maximum 0.47mg/l	ECan, 2010
	pH Level	Minimum 6.5pH & maximum 8.5pH	ECan, 2010
	Total Phosphorus Concentration	Maximum 0.015mg/l	Expert
	Total Nitrogen Concentration	Maximum 0.47mg/ l	ECan, 2010
Water Regulation	Minimum River Flows	Monthly minimum at $2.5m^3/s$	ECan, 2000
	Days River Mouth Closed	Maximum 5 days	Expert
	Number of Flushing Flows	5 flushes	Expert
Aesthetic Values	Algal Mat Colour	No Black Mats	Expert
	Clarity	Minimum 5m	Expert
Conservation Values	Qualitative Macroinvertebrate Community Index	Minimum 5	ECan, 2010
	Number of Threatened Native Species	No decline	Expert
	Number of Significant Landscapes	No decline	CMF, 2010
	Number of Native Fish Species	No decline	CMF, 2010
Educational Values	Number of Publications about River	No decline	Expert
Recreational Values	Number of Recreational Facilities Constructed	No decline	CMF, 2010
	Number of Anglers	500 anglers	Expert
Spiritual Values	Number of Mahinga Kai Species	No decline	Expert

***Appendix 4a*:** The characteristic filtering rule used to evaluate the ecosystem service Water Supply.

Year	Indicator	Economic Impact from Irrigation ($)	River Flow Variability (σ^2)	Total Volume of Water Takes (m^3)
SMS		No decline (>2,280,000)	No increase (<230.9)	No undesirable trend
Pre-dam	1989	2,280,000	137.6	---
	1990	2,280,000	153.5	---
	1991	2,280,000	244.8	---
	1992	2,280,000	702.7	---
	1993	2,280,000	21.7	---
	1994	2,280,000	61.1	1,337,266
	1995	2,280,000	833.8	1,747,373
	1996	2,280,000	230.9	2,149,885
	1997	Construction of Opuha Dam		
Post-dam	1998	9,120,000	37.8	---
	1999	9,120,000	56.8	---
	2000	9,120,000	201.9	---
	2001	9,120,000	93.7	2,585,578
	2002	9,120,000	82.1	2,192,418
	2003	9,120,000	143.6	2,709,809
	2004	9,120,000	73.1	2,752,716
	2005	9,120,000	125.0	3,117,856
	2006	9,120,000	142.3	2,652,294
	2007	9,120,000	37.0	2,402,120
	2008	9,120,000	75.8	3,364,796

*Darkly shaded sections denote years where the safe minimum standard (SMS) is breached.

***Appendix 4b*:** The characteristic filtering rule used to evaluate the ecosystem service Natural Hazard Regulation.

Year	Indicator	Number of Flood Event Fatalities	Number of Flood Flows
SMS		No fatalities (= 0)	No floods (= 0)
Pre-dam	1989	0	0
	1990	0	0
	1991	0	0
	1992	0	0
	1993	0	0
	1994	0	1
	1995	0	0
	1996	0	0
	1997	Construction of Opuha Dam	
Post-dam	1998	0	0
	1999	0	0
	2000	0	0
	2001	0	0
	2002	0	0
	2003	0	0
	2004	0	0
	2005	0	0
	2006	0	0
	2007	0	0
	2008	0	0

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