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Addressing the wicked problem of water resource management: An ecosystem services approach

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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*:**

This paper develops a systematic assessment of the sustainability of ecosystem services provided by rivers impacted by water storage projects. Given the conflicting preferences amongst stakeholders and the incomplete, uncertain and contradictory understanding about river ecology it is recognized that managing water resources sustainably is a wicked problem. In order to address this wicked problem, the methods of multi-criteria analysis and graph analysis are applied, in accordance with integrated water resource management, to assess the potential of investing in water storage projects and explore for sustainable solutions through the construction of an ecosystem services index.

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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). The effects of irrigation in Canterbury are evident as much land use intensification has occurred over the past 20 years (Parkyn & Wilcox, 2004). Today irrigation is viewed as a vital component of the region's land-based economy with approximately 500,000 hectares of agricultural land irrigated. However, a desired target has been set to increase irrigation to 850,000 hectares by 2040 (Canterbury Mayoral Forum, 2009).

This desired target is considered necessary to meet demand, especially given that the high levels of evaporation during the summer months experienced in Canterbury are projected to become more severe with climate change. Despite this, there is a realization that a reliable supply of water from run-of-river water schemes is scarce, as many rivers have reached their maximum allocation limits while maintaining acceptable minimum river flows needed to sustain aquatic health (Dyson *et al*., 2003). Hence, to meet this target, yet manage water resources sustainably, considerable interest has developed in the 'solution' of investing in water storage projects (Frame & Russell, 2009). Indeed, there have been a series of water storage projects proposed for various rivers (*e.g.* Hurunui River, Opihi River) in Canterbury, by way of either river impoundment through the construction of dams or river diversion through the transfer of water to or from a reservoir (Canterbury Mayoral Forum, 2009). While water storage projects are costly, they provide the potential to store water, thus increasing water supply and its reliability. This reliable water supply allows farmers to increase irrigated area and further intensify their land use in an attempt to maximize profits.

Despite the perceived gains from water storage to farmers, these gains are often exaggerated. For example, generally dams designed for irrigation purposes provide only 65 to 85 per cent of projected gains (World Commission on Dams, 2000). Furthermore, other less tangible, but still highly important, non-use (or in-stream) values not associated with the consumptive use of water resources are typically ignored. Yet, scarce water resources from rivers have the character of public 'goods' and are valued for a multitude of reasons by a diverse variety of stakeholder groups (Canterbury Mayoral Forum, 2009). Thus, to adequately value and assess changes to rivers, there is a need to avoid the marginalization of some stakeholders and ensure that the all stakeholder value systems are accounted for.

The need to consider the multiplicity of values from water resources is critical, as while water storage can result in significant gains for farmers, it can also generate significant losses, especially to those non-use values highly dependent on the ecology of the river being functional and healthy. Losos *et al*. (1995), for example, found that water storage projects have resulted in more degradation to threatened species and their habitats than any other activity utilizing environmental resources. Moreover, scientists have long recognized the negative impact on rivers from land use intensification, where the substantial increase in nutrients (*e.g.* nitrates) from the increased application of fertilizers can, through surface runoff, degrade river ecology by way of excessive algal proliferation. Hence, given the potential gains and losses from water storage projects, the potentiality for their investment requires a systematic assessment to ascertain whether these solutions are sustainable.

In order to assess the gains and losses for the many values provided by rivers, the ecosystem services approach is often recommended. 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, such as rivers, that provide well-being to humans (Daily, 1997; National Research Council, 2005). Ecosystem services derive from internal ecological processes through "complex interactions between biotic and abiotic [factors] of ecosystems" (De Groot *et al.*, 2002; p. 394). Significantly, these complex interactions are also evident amongst various ecosystem services (Rodriguez *et al.*, 2006). Hence, ecosystem services, internal ecological processes, agricultural operations, water resource management and external environmental processes that impact rivers (*e.g.* climatic conditions), all interact in complex ways. *Figure 1* indicates conceptually the interactions between the generalized set of factors outlined and the pivotal role ecosystem services perform in connecting the 'subjective value dimension' of human well-being (*i.e.* human system) with the 'objective ecosystem dimension' of the river system itself.

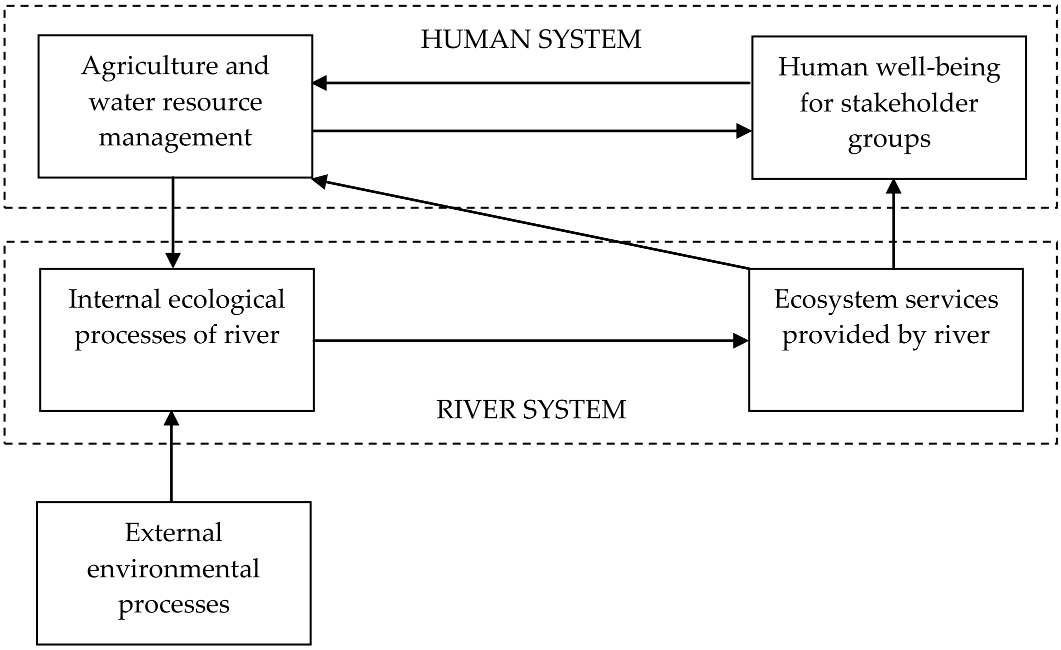

Figure 1: Conceptual diagram of the interactions between the generalized set of factors critical to assessing ecosystem services provided by rivers and their manipulation from water resource management (adapted from Wilson *et al*., 2005).

To date, while numerous researchers have recognized the potential of the ecosystem services approach for considering the many values provided by ecosystems, including rivers, the relevant literature reveals that only some of the more tangible ecosystem services are regularly considered (*e.g.* Water Supply) (Foley *et al.*, 2005; De Groot *et al.*, 2009). Moreover, there are few studies that have systematically assessed the gains and losses to ecosystem services provided by rivers from the impacts of impoundment or diversion (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. A critical debate that is yet to be resolved is how to define and classify the set of all ecosystem services. 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). It is therefore applied herein.

After consideration of the classification of ecosystem services from the *Millennium Ecosystem Assessment* (Capistrano *et al*., 2005), 15 ecosystem services were compiled to be provided from rivers in Canterbury. *Table 1* indicates the set of ecosystem services provided by rivers in Canterbury and examples of each ecosystem service. The only two ecosystem services not included in this set were the ecosystem services Biological Products and Climate Regulation. The class of supporting ecosystem services (*e.g*. Primary Production) were not considered in this compilation as they are better viewed as internal ecological processes that support the production of ecosystem services, rather than ecosystem services *per se*.

Table 1: The set of ecosystem services provided by rivers in Canterbury.

Class	Ecosystem service	Description of ecosystem service	Examples of ecosystem service
Provisioning ecosystem services	Food	Ecosystem supplies food produce	Sport fish, mahinga kai
	Fibre	Ecosystem supplies extractable renewable raw materials for fuel and fibre	Flax, driftwood
	Water Supply	Ecosystem supplies freshwater for use and storage	Irrigation, hydroelectricity, municipal, industrial and stock water supply
	Abiotic Products	Ecosystem supplies extractable non-renewable raw materials for commercial use	Gravel extraction for road chip and concrete
Regulating ecosystem Services	Disease Regulation	Ecosystem regulates the abundance of pathogens	Parasite and toxic algae regulation
	Water Regulation	Ecosystem regulates hydrological flows (*i.e*. surface water runoff)	River flow regulation (*e.g*. minimum river flows)
	Water Purification	Ecosystem purifies and breaks down excess nutrients in water	Removal of pollutants
	Pest Regulation	Ecosystem regulates abundance of invasive or pest species	Stabilization of river banks
	Erosion Control	Ecosystem controls biological catastrophes and stabilizes against erosion, thus, retaining soils	Invasive non-native species (*e.g*. algae, willows, gorse)
	Natural Hazard Regulation	Ecosystem regulates and protects against extreme natural events (*i.e*. floods or droughts)	Flood and drought protection
Cultural ecosystem services	Educational Values	Ecosystem provides opportunities for non-commercial uses (*e.g*. knowledge systems).	Historical/archaeological values & knowledge systems
	Conservation Values	Ecosystem provides existence values for species including important values relating to biodiversity	Native biodiversity/habitat, endangered native species, significant ecological landscapes
	Aesthetic Values	Ecosystem provides aesthetic qualities	Perceived beauty
	Spiritual Values	Ecosystem provides spiritual and inspirational qualities	Tranquillity, Māori values (*e.g*. mauri)
	Recreational Values	Ecosystem provides opportunities for recreational uses	Fishing, hunting, kayaking, picnicking, swimming, walking

The many ecosystem services coupled with the scarcity of water resources have lead to fragmentation and disputes amongst stakeholders (Canterbury Mayoral Forum, 2009; Land and Water Forum, 2010). These disputes are intensified by the conflicting value systems or preferences held amongst an increasingly heterogeneous set of stakeholders (Giordano *et al*., 2007). Indeed, a recreational fisherman might value a river, above all, by its abundance of salmonid fish (*i.e.* ecosystem services Food and Recreational Value), a Ngāi Tahu member by its aquatic health and abundance of mahinga kai/native fish (*e.g.* eel, flounder) (*i.e.* ecosystem services Food and Spiritual Values), a farmer by its capacity to abstract a reliable supply of water for irrigation purposes (*i.e.* ecosystem services Water Supply and Water Regulation), an environmentalist by its biodiversity and presence of threatened bird species that inhabit its banks (*i.e.* ecosystem service Conservation Value) and a water treatment firm by its water quality and the treatment costs required to produce safe drinking water (*i.e.* ecosystem services Water Supply and Water Purification).

These disputes between stakeholder groups, which have become antagonistic especially between those stakeholders that want to consume water (*e.g.* farmers) and those stakeholders that want to conserve water instream (*e.g.* environmentalists, recreationalists), renders trade-offs between ecosystem services as inescapable (Rodriguez *et al*., 2006). Any attempt to maximize the gains of a single ecosystem service will invariably lead to losses to other ecosystem services (Holling & Meffe 1996; Jackson *et al*., 2001). Hence, the problem of managing water resources sustainably is challenging, as it is unfair to ignore or marginalize the preferences of some stakeholder groups. After all, "[w]ater storage is only one of the things that need to be considered ... in Canterbury. Other issues that [also] need to be considered include land use intensification, water quality, cultural values, tangata whenua objectives and recreation uses" (Whitehouse *et al*., 2008; *p*. 4). This paper, therefore, attempts a systematic assessment for: one, the determination of the sustainability of ecosystem services provided by rivers impacted by water storage projects; two, the identification of the conflicting preferences between stakeholder groups; and three, the analysis of gains and losses in ecosystem services provided and, therefore, an understanding of the trade-offs implicated from investment in a water storage project (Grimble & Wellard, 1997; Reed *et al*., 2009).

The remainder of the paper is structured as follows. In *Section 2* the problem of managing water resources is identified as a 'wicked problem', which requires the adoption of a post-normal approach to science to explore for sustainable solutions. In *Section 3* the methods of systematic assessment is outlined, which is, in part, determined by an ecosystem services index. Specific details of the construction of an ecosystem services index are indicated, which include the use of multi-criteria analysis and graph analysis. These methods once applied, will indicate the conflicts and trade-offs of investing in water storage projects, and whether such projects are sustainable for rivers in Canterbury. Finally, in *Section 4* conclusions are offered.

1.1 Wicked Problems & Integrated Water Resource Management

The problem of water resource management in Canterbury is increasingly identified as a wicked problem (Frame & Russell, 2009). This is especially so, as solutions proposed are not easily undone. For example, the 'solution' of investing in a water storage project is, for all intents and purposes, one-off and irreversible given the large costs required to construct and decommission such projects and their long life expectancy often being over 100 years (Wieland, 2010). Wicked problems, however, arise largely from the interplay of a multiplicity of stakeholders with conflicting preferences and the incomplete, uncertain and contradictory understanding emphasized in the numerous interdependent factors that comprise the problematic system (*i.e.* rivers) assessed (Rittel & Webber, 1973; Turnpenny *et al*., 2009). Hence, with wicked problems complexity is twofold. Complexity is apparent in the interdependent factors constituting the river system and it is apparent in the heterogeneous preferences of stakeholders. This twofold complexity renders wicked problems irreducible. As a consequence, any effort to solve a wicked problem by 'compressing' it into a singular objective using naive 'optimal' solutions derived from a 'rigidly' structured problem statement (*e.g.* constrained optimization methods) invariably generates solutions that may be successful in the short-term, but generate losses that outweigh gains in the long-term when these unsustainable solutions are put into practice (Pahl-Wostl, 2007). These unsustainable solutions result, because rigidly structured problems, while elegant, can hide or over-simplify the complexity (*i.e.* removing the many interactions including feedback) involved in wicked problems. As such, wicked problems are problems that are ill-structured and best formulated in more 'flexible' ways.

In order to explore for sustainable solutions, the management of water resources should adopt a 'post-normal' approach to science (Functowicz & Ravetz, 1993; 1997). Post-normal science acknowledges the high investment stakes resultant from the conflicting, seemingly irresolvable, value system commitments held by a multiplicity of stakeholders and the high, seemingly irreducible, uncertainties resultant from the incomplete, uncertain and contradictory understanding of many resources (*Figure 2*). Much of this incomplete, uncertain and contradictory understanding arises from the complexity of managing water resources from river systems. However, different understanding also is derived from the varied perspectives held, which are adopted in relation to our values. In this way, perspectives are value-laden understandings. For example, a recreational fisherman values the recreational activity of fishing. Their perspective, therefore, encompasses an understanding of the river system that supports their values for fishing. Hence, post-normal science is an approach that recognizes the impossibility of transcending perspectives developed from values to a value-free objectivity wherever derivatives of wicked problems remain present. This position of post-normal science differs fundamentally from 'normal science' (Kuhn, 1962), which assumes the detached conditions of value neutrality (or understanding free of perspective) and a scientific progression toward certainty (Allison & Hobbs, 2006; Frame & Brown, 2007).

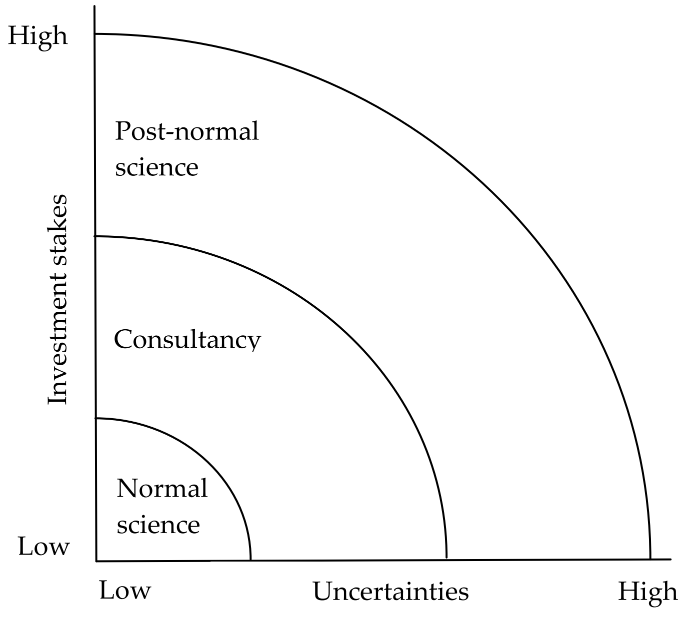

Figure 2: A typology of approaches to science (adapted from Functowicz & Ravetz, 1992).

By post-normal science acknowledging the high uncertainties and high investment stakes involved with wicked problems, it upholds the legitimacy of a co-existence of perspectives (Functowicz & Ravetz, 1997; O'Connor, 1999; Lovell *et al.*, 2002; Kolkman *et al.*, 2005). Progress towards objectivity through various perspectives can be found through the integration of these perspectives, as each perspective is likely to have some elements of 'truth' within it, which are potentially missing from the perspectives of others (Verweij *et al*., 2006). The philosopher Nietzsche (1887; III, p. 12), who originated the epistemological position of perspectivism, would have supported the post-normal approach to science as: "the more affects we allow to speak about one thing, the more eyes, different eyes, we can use to observe one thing, the more complete will our concept of the thing, our objectivity, be." The greatest level of objectivity is, therefore, indicated in a varied set of perspectives that are integrated with one another, providing hopefully a common 'factual' perspective that is without contradictions (Anderson, 1998).

Despite the legitimacy of various perspectives, Nietzsche (1887) recognized that not all perspectives are equal. Rather, better perspectives are evident in those that have had their understanding appraised rigorously with regards to their logical consistency and empirical adequacy. In this sense, two aspects provide progress towards objectivity in a post-normal science. The first is the incorporation of a variety of 'scientists' with a thoroughgoing understanding of river systems and water resources within the region, rather than those with an understanding that remains undeveloped or too heavily entangled within the management of water resources. The second is the need to make perspectives of scientists explicit (Haas, 2004; Turnpenny *et al*., 2009). This allows contradictions amongst scientists to be more easily identified and, in turn, compromising solutions to be more easily reached (Von Winterfeldt, 1992; Antunes *et al*., 2006; Pahl-Wostl, 2007).

There is much agreement that the management of water resources that has previously sought control, a singular perspective and the preferences of the 'dominant' stakeholder group have lead to fragmentation and disputes. To manage water resources sustainably, integration is required between the varied perspectives of scientists, between the multiplicity of stakeholders and their conflicting preferences and also between scientists and stakeholders to ensure both of their 'voices' are heard (Fischer, 2000; Paton *et al*., 2004). This challenging integration has been formally recognized for the management of water resources through the term 'integrated water resource management' (Food and Agricultural Organization, 2000; Kolkman *et al*., 2005; Ferreya *et al*., 2009), which is considered to provide the greatest promise for sustaining the delivery of ecosystem services provided by rivers (DeReynier *et al*., 2010).

Despite the importance of integrated water resource management, there appears to be a lack of conviction to make the methodological shift towards a post-normal science that acknowledges conflicts and contradictions, aids integration and allows, in turn, for the exploration of sustainable solutions. Rather, methods used, despite claims to the contrary, continue to support normal science and rigidly structured problem statements (Kolkman *et al*., 2005). For example, while Weng *et al*. (2010) recently applied a number of methods to allegedly aid integrated water resource management, they did so with methods typically associated with normal science. Yet, post-normal science and integrated water resource management are palpably better practiced with methods that accommodate the varied perspectives of scientists and the multiplicity of preferences held by stakeholders (Fischer, 2000; Carlsson & Berkes, 2005; Kolkman *et al*., 2005; Kodihara *et al*., 2010). Indeed, it is methods that 'respect' conflict and contradictions within the form of an ill-structured problem statement, yet continue to uphold quantitative analysis rather than lengthy qualitative narrative, which will genuinely aid integrated water resource management. Accordingly, in this paper a systematic assessment of the potential of investing in water storage projects on rivers in Canterbury is undertaken by methods conducive with a post-normal approach to science.

2.0 Non-Market Valuation & Indices

The value of ecosystem services provided by rivers are typically assessed and measured in monetary terms. However, more than 80 per cent of ecosystem services lack functioning markets. Their value, therefore, cannot be inferred from market prices (De Groot *et al*., 2002; Swinton *et al*., 2007). As a consequence, 'missing' markets for many ecosystem services (*e.g*. Erosion Control, Spiritual Values) leave them either undervalued or erroneously given an implicit value of zero (Loomis *et al*., 2000; Dyson *et al*., 2003; National Research Council, 2005; Barkmann *et al*., 2008). However, the mismanagement of scarce water resources results when the problem of missing markets is not adequately addressed. In order to tackle this problem, 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 methods are methodologically advanced, they require a painstaking amount of effort in gathering and analysing information from a large sample of stakeholders. This can make the undertaking of non-market valuation costly and time-consuming (National Research Council, 2005; Baskaran *et al*., 2010).

It is apparent that the monetization of many ecosystem services is not necessarily appropriate (Failing *et al*., 2003). As a result, water resource managers may resign themselves to assessment, which is unsystematic (Prato, 1999). However, while expressing the value of ecosystem services in monetary terms remains difficult and possibly inappropriate, the economists Boyd and Banzhaf (2007; p. 617) highlight the importance of adopting "standardized units of account to measure the value of ecosystem services." Fortunately, the value of ecosystem services can be assessed without their monetization through the construction of a single metric that aggregates the value of ecosystem services provided from rivers into a 'utility' index as a measure of human well-being. However, despite that indices are widely employed in assessing aspects of running waters (*e.g*. Macroinvertebrate Community Index) (Hering *et al*., 2006), there remains little consensus as to how best to construct them (Saisana & Saltelli, 2008), let alone those that aggregate ecosystem services in the form of an ecosystem services index.

Despite this lack of consensus, in this paper it is recognized that the adequate construction of an ecosystem services index of a river system requires two components: utility scores that represent the amount of each ecosystem service delivered by the river, and preferential weights for each ecosystem service provided that reflect stakeholder preferences. The appropriate construction of an index through the aggregation of utility scores and preferential weights is attained by multi-criteria analysis. This is because multi-criteria analysis is an overarching term depicting a set of methods capable of weighting and aggregating multiple values together (Munda *et al*., 1994). With preferential weights assessed by stakeholders and utility scores estimated for each ecosystem service delivered either with or without the proposed water storage project, then an ecosystem services index can be formed in accordance with *Equation 1*.

$$\mathrm{ESI} = \sum w_n s_{in}$$

Equation 1: The ecosystem services index (adapted from Banzhaf & Boyd, 2005).

Here ESI is the ecosystem services index;

w_n is the preferential weight w for ecosystem service n; and

s_{in} is the change in the ecosystem service n delivered for the proposed water storage project i.

Previously, it had been recognized that the complexity of ecosystems results in the interdependence of many ecosystem services, not the independence of ecosystem services as implied by the summation function in *Equation 1* (De Groot *et al*., 2002; Rodriguez *et al*., 2006). Therefore, a problem exists with the construction of the ecosystem services index herein in that it would lead to double counting. This problem is addressed at a later section of this paper through the estimation of utility scores of each ecosystem service by graph analysis, a method that accounts for the complex interactions between factors. Hence, double counting is addressed in this paper, which is significant, as Fisher *et al*. (2009) has noted that only one of 34 recent ecosystem services studies surveyed have raised, let alone aptly addressed, this problematic issue.

2.1 Preferential Weights & Multi-Criteria Analysis

It has been established that the construction of an ecosystem services index requires two components: utility scores that represent the delivery of the set of ecosystem service and preferential weights for each ecosystem service. With regards to the latter component, the determination of preferential weights requires a suitable method of multi-criteria analysis. One method that can determine preferential weights, has strong axiomatic foundations, is proven to be useful for the construction of indices and is particularly applicable for "preference analysis in complex, multi-attribute problems" (Herath, 2004; p. 264) is the analytical hierarchy process (Saaty, 1995; Petkov *et al*., 2007). In fact, this method has been applied to determine preferences for ecosystem services (*e.g*. Zhang & Liu, 2009) and has been successfully applied to construct an ecosystem services index (Hearnshaw *et al*., 2011).

The analytical hierarchy process is a method that decomposes assessments of preferences for values (or other criteria) into a hierarchical network (Saaty, 1995). From the hierarchical network constructed, pairwise comparisons between ecosystem services and their classes can be made on a one-to-nine scale, where one represents neutrality between the pairing and nine represents an overwhelming preference for one ecosystem service over the other. Each pairwise comparison on this scale captures the cardinal intensity of preference between the pairing assessed. Thus, in using pairwise comparisons to indicate preference intensity, the 'trade-offs' between ecosystem services are mapped. The pairwise comparisons of all pairings depict ratios, which can be expressed in a ratio matrix A (*Equation 2*). It is in this form that the strong axiomatic foundations of analytical hierarchy process are highlighted (*e.g*. reciprocal comparison between pairings) (Strager & Rosenberger, 2006). While the ratio matrix will be computationally demanding to solve, there are a number of programmes (*e.g*. *Expert Choice*) dedicated to undertaking such computations.

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

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. A lower level again contains 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).

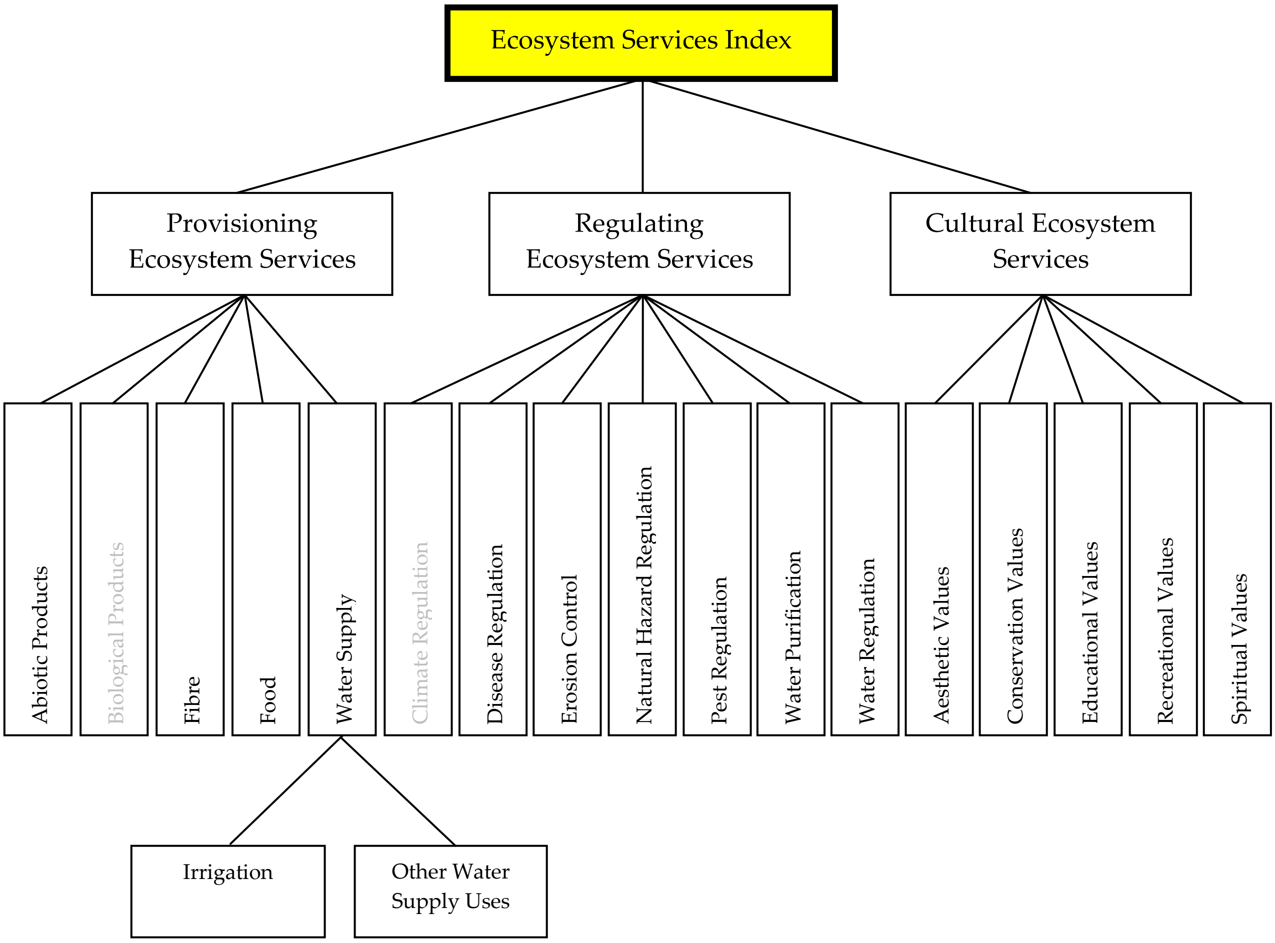

Figure 3: The hierarchical network for constructing preferences of ecosystem services.

In an effort to limit the impracticalities of sampling a large number of stakeholders, the determination of preferential weights were performed by representatives of stakeholder groups (Failing *et al.*, 2007; van Vliet *et al.*, 2010). The use of representatives as 'overseers' of a stakeholder group rather than a large sample of the population of stakeholders is less costly 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 have excessively inconsistent and unstable preferences and neither possess sufficient understanding of the ecosystem services provided, nor an adequate grasp of the methods applied (Alvarez-Farizo & Hanley, 2006; Barkmann *et al.*, 2008; Carlsson, 2010). However, it is recognized that the selection of representatives makes claims of 'representation' difficult (Spash, 2007). Hence, the appropriate selection of representatives must attempt to represent a reasonably equitable and proportional microcosm of all affected stakeholder groups (Turner *et al.*, 2010). This difficult task, which is typically *ad hoc* in water resource management (Reed *et al.*, 2009), was made somewhat easier as a number of representatives were already selected in a Steering Group specifically established to aid in the management of water resources in Canterbury.

In total 21 representatives were selected that represented many stakeholder groups important to water resource management. Representatives were placed into four distinguishable stakeholder groups judged by their affiliations. These groups were the Steering Group, the Government Group, the Water Consumers Group and the Water Conservators Group. The Steering Group consisted of many representatives formally selected to participate in this group organized by Environment Canterbury, the regional government authority. The Government Group consisted of representatives as employees to local, regional and national government authorities. The Water Consumers Group consisted of representatives from development organizations, irrigation advocacy organizations and water storage firms. Finally, the Water Conservators Group consisted of representatives from non-governmental organizations that advocate the conservation of water resources. Group membership was exclusive, except for the Steering Group, where representatives belonged to this group and another group.

The representatives' preferences of ecosystem services were collected and subsequently analysed in the computational programme *Expert Choice*, which provided each representative a set of preferential weights for the set of ecosystem services provided by rivers in Canterbury. In addition, an inconsistency measure was calculated that gauged the degree of intransitivity between preferences throughout all ecosystem service pairings. This inconsistency measure is significant as it recognizes the presence of bounded rationality and inconsistency in preferences held. The initial average inconsistency for representatives was 13 per cent. The recommended level of inconsistency is about ten per cent (Saaty, 1995). Hence, some representatives were asked if they wish to revise 'inconsistent' preferences according to various computationally-devised remedies. Those 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 all representatives are indicated. The ecosystem service Water Supply is the most preferred. This preference supports previous research (MacDonald & Patterson, 2008). However, despite this expectation, the average ratio of Irrigation to Other Water Supply Uses was significantly less than one. Moreover, while the ecosystem service Water Supply was given the greatest preference, other ecosystem services were found to have relatively high weights. These findings iterate that water resources are valued for many reasons. However, according to Rodriguez *et al*. (2006) the preferences for ecosystem services tend towards provisioning ecosystem services, then regulating ecosystem services and then finally cultural ecosystem services. However, in the preferences obtained, there is evidence to suggest that regulating ecosystem services overall are the most preferred class of ecosystem services for rivers in Canterbury.

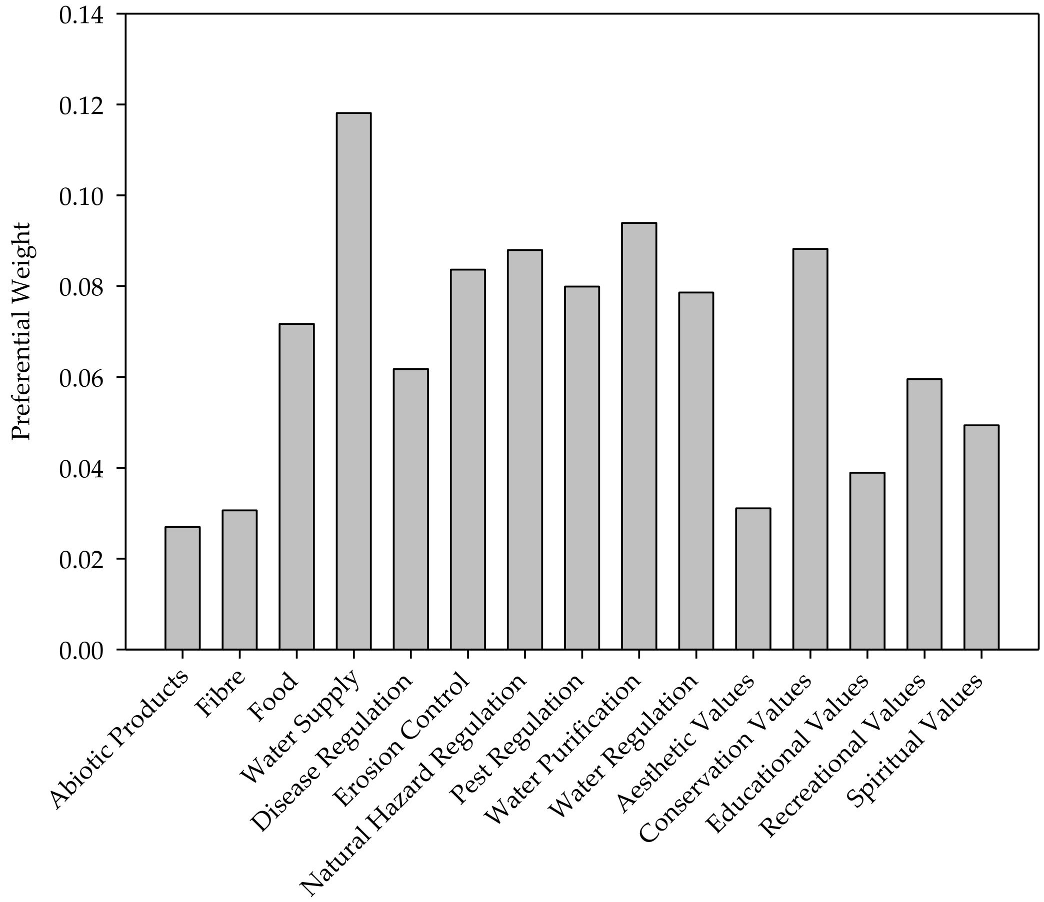

Figure 4: Preferential weights for ecosystem services provided from rivers in Canterbury.

With preferential weights for ecosystem services determined, statistical tests will be performed to analyse the intra-group and inter-group differences in preferential weights elicited according to the method outlined by Strager and Rosenberger (2006). The initial test will indicate whether the intra-group preferential weights are compatible to be grouped together by their affiliations. If intra-group variation is too large, then the average preferential weight for the group will not be used to represent the group. For groups where variation is found to be satisfactory, they will be analysed against the preferential weights of ecosystem services with other groups. To determine whether preferential weights both within groups and between groups are homogeneous, Friedman's Q statistic will be applied. This statistical test is a non-parametric, two-way analysis of variance by ranks statistic. However, this statistic does not indicate which ecosystem service representatives within groups or between groups may differ on. To indicate this, the non-parametric Mann-Whitney U test will be applied (McGlave & Benson, 1991). By ascertaining where commonalities and conflicts lie within and between groups as to their preferences for ecosystem services, it will aid the establishment of compromising solutions. As an indication of these statistical tests, *Table 2* reports Friedman's Q statistic to assess intra-group differences in preferential weights for the Water Consumers Group. It was found that Water Consumers rejected the null hypothesis of similar preferential weights for ecosystem services. This suggests that there are possible conflicts in preferences in this group, which reiterates that water resource management is a wicked problem.

Table 2: Summary of Friedman's *Q* statistic for the Water Consumers Group.

Friedman's Q statistic	37.622
Significance	<0.001
N	6

Preferences of representatives for ecosystem services also need not be assumed to be stable and unchanging. This is important because ecosystem services are likely to be far less stable than services tradeable in markets because of their limited exchange properties and poor understanding about them (Boyd & Banzhaf, 2007). Accordingly, preferences of representatives will be elicited under the hypothetical scenario that climatic conditions have changed to that projected for the region. That is, climatic conditions that have higher average temperatures and lower average rainfall (Canterbury Mayoral Forum, 2009).

2.2 Utility Scores & Graph Analysis

The estimation of utility scores that account for the change in delivery of ecosystem services provided with a water storage project is a difficult undertaking. This difficulty lies in that poor understanding exists over ecosystem services and that water storage projects are one-off and irreversible, which impact ecosystem services provided from river systems through a large number of factors that interact in complex ways. A suitable method that provides quantitative outputs, yet necessarily accounts for the interdependencies between factors is by way of graph analysis. These attributes of graphs are ideal for ascertaining utility scores for the delivery of ecosystem services as accounting for the complex interactions between factors including ecosystem services minimizes the problem of double counting. Graph analysis is also appropriate for integrated water resource analysis as it allows the varied perspectives of scientists to be easily integrated, whereby common 'factual' perspectives and contradictions in understanding can be readily identified (Allison & Hobbs, 2006). The suitability of graphs is further established in that they can be represented mathematically in terms of an adjacency matrix without restrictive assumptions (*e.g.* no feedback), but also they can be represented diagrammatically, which provides an accessible format for stakeholders to understand river ecology without the need for lengthy qualitative narrative (Coyle, 2000; Liu *et al*., 2008). This double aspect of graphs is significant for ensuring that favourable water storage projects do indeed attract investment. This is because water storage projects may fail to attract investment because of a lack of quantitative analysis, or even where analysis is undertaken and appears favourable for investment may continue to fail because stakeholders cannot comprehend the analysis and remain sceptical about its outputs (DeReynier *et al*., 2010).

In addition, to the 15 ecosystem services supplied by rivers in Canterbury, other factors were also compiled, as internal ecological processes, external environmental processes, agricultural operations and water resource management attributes in accordance with *Figure 1*, from relevant literature and the perspective of the analysts. With all factors considered, a graph was developed by the analysts indicating interactions as, causal interpretations, between factors. Factors, other than ecosystem services, that only received one interaction were removed from the graph. The final compilation of factors became the base set of factors used amongst scientists to identify interactions between factors in order for them to devise their own graph (see *Appendix 1*). Thus far, a total of five regionally-based scientists with extensive understanding of various aspects of river ecology or water resources have developed graphs that, in effect, represent their own perspective of the functioning of a generic river system in Canterbury.

The elicitation of the interactions to construct a graph of a generic river in Canterbury began by each scientist indicating the direction and weight of causation between factors that were within their understanding. Each interaction identified was indicated with an arrow and a weight given on the positive-negative interval scale [-1, 1]. A weight of -1 would indicate an overwhelming negative causation in the direction of the arrow between factors. Similarly, a weight of 1 would indicate an overwhelming positive causation in the direction of the arrow between factors. Weights between -1 and 1 indicate lower degrees of causality in the interaction between factors. With all interactions indentified within the scientist's perspective, each scientist was able to produce a graph, which they understood how a generic river system in Canterbury functions.

Once all scientists' graphs are complete, they will be aggregated together by additively superimposing them to form a 'complete' graph of a generic river in Canterbury. In aggregating graphs together, a common 'factual' perspective held will emerge and contradictions between scientists can be identified. The identification of contradictions (*e.g*. negative calculus given by one scientist and a positive calculus given by another for the same interaction) can be subsequently evaluated to find a compromising solution. However, in addition to identifying contradictions the complete graph, upon its construction, can be validated for its logical consistency and empirical adequacy by scientists and stakeholders alike (Jakeman & Letcher, 2003). This validation by scientists and stakeholders provides a scientifically sound and defensible understanding that makes outputs from the analysis of the graph more difficult to reject or ignore (Voinov & Bousquet, 2010).

The aggregated graph, once validated, can be transformed to the mathematical form of an adjacency matrix. By transforming the graph into an adjacency matrix, it becomes possible to undertake static and dynamic graph analysis. Static graph analysis is performed by using various graph indices based in matrix algebra. Relevant graph indices include indegree, outdegree and centrality (Harary *et al*., 1965). Indegree refers to the cumulative weights of interactions coming into a particular factor, while outdegree refers to the cumulative weights of interactions leaving a factor. With the determination of indegree and outdegree indices, the centrality index for each factor can be determined by summing outdegree and indegree indices together. Centrality then, indicates how connected in terms of cumulative weight a factor is relative to other factors in the graph. *Table 2* summarizes the equations to be used to determine these graph indices discussed. Utility scores for the delivery of ecosystem services are obtained from the centrality indicated for each of the ecosystem services. Hence, the change in the delivery of ecosystem services, and therefore utility scores, with water storage projects is indicated with the addition of factors specific to water storage to the aggregated graph.

Table 2: Equations of graph indices.

Index	Equation
Outdegree $od(C_i)$	$od(C_i) = \sum_{k=1}^{N} \bar{e}_{ik}$
Indegree $id(C_i)$	$id(C_i) = \sum_{k=1}^{N} \bar{e}_{ki}$
Centrality L	$L_i = od(C_i) + id(C_i)$

Where N is the total number of factors; and
e is the weights on interactions.

The structure of the graph can also be analysed through the types of factors, which can be determined by classing factors as transmitter, receiver and ordinary factors in the graph. Transmitter factors are those factors that have a non-zero outdegree and zero indegree. Receiver factors are those factors that have a non-zero indegree and zero outdegree. Ordinary factors are all other factors that have both a non-zero indegree and outdegree.

In addition to static graph analysis, dynamic graph analysis can also be performed. The mathematics of dynamic graph analysis is that each factor sums its inputs and yields an output according to a particular mathematical function. Specifically, the factor is activated to whatever degree the inputs dictate, and this value will pass a signal to its neighbouring factors that interact with this factor. The signal arriving at the neighbouring factor is determined by both the weight of the interaction e_n and the activation input a_n of the transmitting factor. Accordingly, the signal downstream to a factor is determined by the weighted sum of the individual products of the activation input and the interaction weight, which is then transformed by a non-linear activation squashing function f into the interval space [0, 1] at each computational step (*Figure 5*) (Kosko, 1992). Given the large number of factors and the numerous interactions between factors expected in the graph, dynamic graph analysis will be performed using the spreadsheet computational programme *FCM Mapper*, which is specifically designed to undertake dynamic graph analysis.

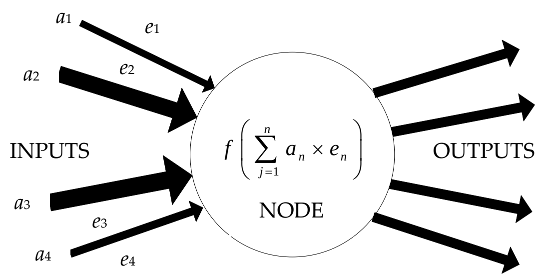

Figure 5: A simulated factor in the graph using dynamic graph analysis. Briefly, in dynamic graph analysis activation inputs *a* are multiplied by the interaction weights *e* and these products are summed. The sum is input into an activation squashing function *f*. The result is output as an activation input to neighbouring factors.

Utility scores for the delivery of ecosystem services are indicated by dynamic graph analysis after the simulation, as a series of computational steps, reaches a point where the activation outputs for all factors settle on to a fixed value. Once settled, the activation outputs for each ecosystem service correspond to utility scores. Similarly to static graph analysis, the change in the delivery of ecosystem services, and therefore utility scores, with water storage projects will be indicated with the addition of factors specific to water storage. Hence, with the completion of static and dynamic graph analysis, utility scores before and after a water storage project will be obtained. These utility scores will provide evidence of the gains and losses to the set of ecosystem services from a water storage project. Moreover, utility scores, whether obtained from static or dynamic graph analysis, will be multiplied with corresponding preferential weights in order to construct an ecosystem services index.

2.3 Weak & Strong Sustainability

The ecosystem services index developed herein provides the basis in which to assess the sustainability of ecosystem services provided from rivers, where sustainability is defined as human well-being that is non-declining over the long-term (Neumayer, 2003). Human well-being ensures that future generations are provided with at least the same welfare from ecosystem services as present generations. Hence, if the ecosystem services index, as an aggregation of ecosystem services that provide well-being to humans, is non-declining, then a sustainable solution can be inferred. Thus, to ensure the sustainability of ecosystem services provided by rivers the ecosystem services index must be equal or greater than the ecosystem services index prior to the water storage project. It is through this non-declining account of the ecosystem services index that it is possible to determine the criterion of 'weak sustainability', because it assumes that all ecosystem services are compensatory and, therefore, commensurable and reducible to a single metric (*i.e.* an ecosystem services index). For example, such an index implies that a high utility score for the ecosystem service Recreational Values can compensate a low utility score for the ecosystem service Water Regulation.

In allowing for compensation, the ecosystem services index neither is able to consider who explicitly gains and losses from a proposed water storage project nor is it able to consider the criterion of 'strong sustainability'. Those that adhere to and advocate the use of the strong sustainability criterion assert that there is a "... minimum quantity of [internal] ecosystem ... processes... required to maintain a well-functioning ecosystem capable of supplying [ecosystem] services" (Fisher *et al*., 2009; p. 2053). Accordingly, unlike weak sustainability, strong sustainability considers welfare in non-compensatory terms, so that the assessment of strong sustainability by an ecosystem services index is inappropriate (Faucheux & O'Connor, 1998). For this reason, strong sustainability recognizes that measuring the potential delivery of an ecosystem service does not necessarily indicate whether the internal ecological processes that produce ecosystem services are sustainable (Mooney *et al*., 2005). Ideally then, weak and strong sustainability should be assessed when exploring for sustainable solutions, as genuine claims of sustainability are mistaken unless both the well-being of future generations (*i.e.* human system) and the internal ecological processes that produce the set of ecosystem services are maintained (*i.e.* river system).

The difficulty with strong sustainability has been in making the criterion operational and practicable (Prato, 2007; Turner *et al*., 2010). However, strong sustainability can be made operational by defining targets for internal ecological processes and ecosystem services that should be reached. Despite the need for targets, a critical reason for poor water resource management is that such targets (or limits) are often not adequately defined or managed for (Land and Water Forum, 2010). However, recently, the Canterbury Mayoral Forum (2010) proposed various targets for internal ecological processes and ecosystem services from river systems as a strategy to manage the region's water resources more sustainably. The targets that have been proposed are, for the most part, defined by trends. For example, a target devised is 'an upward trend in the abundance of native fish populations on rivers in Canterbury'. Hence, this target could be assessed by whether the factor or internal ecological process Native Fish Production in the graph assembled increases or decreases in its centrality (*i.e.* static graph analysis) or activation output (*i.e.* dynamic graph analysis) with the addition of factors specific to water storage.

In applying these targets to assess strong sustainability, sustainability would be observed in its most complete form where all targets have been reached. It is, of course, unlikely that strong sustainability in this most complete of forms will be demonstrated; this is especially so when one considers that approximately two thirds of all ecosystem services provided worldwide are degraded (Capistrano *et al*., 2005). Where a target is not reached, there are several methods for the determination as to whether strong sustainability has progressed or regressed. One simple method is the checklist approach where a comparison is made between the number of targets not reached without the water storage project and the number of targets not reached with the water storage project. One obvious criticism with this assessment is that it assumes that all ecosystem services are equally preferred.

An alternative method is the lexicographic-based characteristic filtering rule. This method does not assume that all ecosystem services are equally preferred. Rather, in using the preferential weights for the set of ecosystem services provided, 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 strong sustainability. 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 assessed from most to least preferred. The characteristic filtering rule is applied first to the most preferred ecosystem service and establishes whether its targets have been reached or not with and without the proposed water storage project (Earl, 1986; Lockwood, 1996). Where targets are reached, then the targets for the next most preferred ecosystem service is subsequently assessed. This process continues until a target has been reached for one project, but not for the other. When this happens it indicates, which project provides greater progress towards strong sustainability.

3.0 Conclusion

In this paper, the wicked problem of water resource management is addressed through a post-normal approach to science. This novel approach applies multi-criteria analysis and graph analysis as a means to determine the sustainability of ecosystem services provided by rivers. Moreover, these methods provide an indication of where conflicts, contradictions and trade-offs are when assessing the potential of investing in proposed water storage projects. These methods and the analysis derived from them are important for managing water resources sustainably and are considered essential to progress integrated water resource management towards a legitimate methodological approach.

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